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Technical Support Manual: Waterbody Surveys and Assessments for Conducting Use Attainability Analyses

Foreword

The Technical Support Manual: Water Body Surveys and Assessments for Conducting Use Attainability Analyses contains technical guidance prepared by EPA to assist States in implementing the revised Water Quality Standards Regulation (48 FR 51400, November 8, 1983). EPA prepared this document in response to requests by several States for additional guidance and detail on conducting use attainability analyses beyond that which is contained in Chapter 3 of the Water Quality Standards Handbook (December, 1983).

Consideration of the suitability of a water body for attaining a given use is an integral part of the water quality standards review and revision process. This guidance is intended to assist States in answering three central questions:

(1) What are the aquatic protection uses currently being achieved in the water body?
(2) What are the potential uses that can be attained based on the physical, chemical and biological characteristics of the water body?; and,
(3) What are the causes of any impairment of the uses?

EPA will continue providing guidance and technical assistance to the States in order to improve the scientific and technical basis of water quality standards decisions. States are encouraged to consult with EPA at the beginning of any standards revision project to agree on appropriate methods before the analyses are initiated, and frequently as they are conducted.

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

Elliot Lomnitz
Criteria and Standards Division (WH-585)
401 M Street, S.W.
Washington, D.C. 20460

Steven Schatzow, Director
Office of Water Regulations and Standards

TECHNICAL SUPPORT MANUAL:
WATER BODY SURVEYS AND ASSESSMENTS

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° SECTION I: INTRODUCTION

One of the major pieces of guidance discussed within the Water Quality Standards Handbook (November, 1983) is the "Water Body Survey and Assessment Guidance for Conducting Use Attainability Analyses" which discusses the framework for determining the attainable aquatic protection use. This guidance describes the framework and suggests parameters to be examined in order to determine:

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

(2) What are the potential uses that can be attained based on the physical, chemical and biological characteristics of the water body?; and,

(3) What are the causes of any impairment of the uses?

The purpose of the technical support manual is to highlight methods and approaches which can be used to address these questions as related to the aquatic life protection use. This document specifically addresses stream and river systems. EPA is presently developing guidance for estuarine and marine systems and plans to issue such guidance in 1984.

Several case studies were performed to test the applicability of the "Water Body Survey and Assessment" guidance. These case studies demonstrated that the guidance could successfully be applied to determine attainable uses. Several of the States involved in these studies suggested that it would be helpful if EPA could provide a more detailed and technical explanation of the procedures mentioned in the guidance. In response, EPA has prepared this technical support manual. The methods and procedures offered in this manual are optional and States may apply them selectively. States may also use their own techniques or methods for conducting use attainability analyses.

A State that intends to conduct a use attainability analysis is encouraged to consult with EPA before the analyses are initiated and frequently as they are carried out. EPA is striving to develop a partnership with the States to improve the scientific and technical bases of the water quality standards decision-making process. This consultation will allow for greater scientific discussion and better planning to ensure that the analyses are technically valid.

Consideration of the suitability of a water body for attaining a given 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 basis for evaluating whether the water body is suitable for a particular use. It is not envisioned that each water body would necessarily have a unique set of uses. Rather the characteristics necessary to support a use could be identified so that water bodies having those characteristics might be grouped together as likely to support particular uses.

Since the complexity of an aquatic ecosystem does not lend itself to simple evaluations, there is no single formula or model that will provide all the answers. Thus, the professional judgment of the evaluator is key to the interpretation of data which is gathered.

° SECTION II: PHYSICAL EVALUATIONS

OVERVIEW

The physical characteristics of a water body greatly influence its reaction to pollution and its natural purification processess. The physical characteristics also play a great role in the availability of suitable habitat for aquatic species. An understanding of the nature of these characteristics and influences is important to the intelligent planning and execution of a water body survey. Important physical factors include flow, temperature, substrate composition, suspended solids, depth, velocity and modifications made to the water body. Effects of some of these factors are so interrelated that it is difficult or even impossible to assign more or less importance to one or the other of them. For example, slope and roughness of channel influence both depth and velocity of flow, which together control turbulence. Turbulence, in turn, affects rates of mixing of wastes and tributary streams, reaeration, sedimentation or scour of solids, growths of attached biological forms and rates of purification (FWPCA, 1969). Thus evaluating the factors which constitute the physical environment cannot be done by just assessing one parameter but rather a broader assessment and view is needed.

The purpose of this section is to amplify the methods and types of assessments discussed in Chapter 3 of the Water Quality Standards Handbook for evaluating the physical characteristics of a water body. The analyses proposed in this section, as well as the other sections of this document, do not constitute required analyses nor are these all the analyses available or acceptable for conducting a use attainability analysis. States should design and choose assessment methodologies based on the site-specific considerations of the study area. The degree of complexity of the water body in question will usually dictate the amount of data and analysis needed. States should consult with EPA prior to conducting the survey to facilitate greater scientific discussion and better planning of the study.

CHAPTER II-1
FLOW ASSESSMENTS

The instream flow requirement for fish and wildlife is the flow regime necessary to maintain levels of fish, wildlife and other dependent organisms. Numerous methodological approaches for quantifying the instream flow requirements of fish, wildlife, recreation, and other instream uses exist. Each method has inherent limitations which must be examined to determine appropriate methods for recommending stream flow quantities on a site-specific basis. The following describes in detail several of the more commonly used and accepted methods.

TENNANT METHOD

One of the widely known examples of an instream flow method is the Tennant method (1976). Based on analyses conducted on 11 streams in Montana, Wyoming and Nebraska, Tennant determined the following:

(1) Changes in aquatic habitat are remarkably similar among streams having similar average flow regimes.
(2) An average stream depth of 0.3 meters and an average water velocity of 0.75 ft/sec were the critical minimum physical requirements for most aquatic organisms.

(3) Ten percent of the average annual flow would sustain short-term survival for most fish species.
(4) To sustain good survival habitat, thirty percent of the average annual flow was adequate since the depth and velocities generally would allow fish migration.
(5) Sixty percent of flow provides outstanding habitat.

Using the above information, Tennant proposed a range of percentages of the average annual flow regime needed to maintain desired flow conditions on a semi-annual basis. These ranges are summarized by the following:

Flow Description	Recommended flow regime October-March	April-September
Flushing	200% of the average annual flow	
Optimum range	60%-100% of the average annual flow	
Outstanding	40%	60%
Excellent	30%	50%
Good	20%	40%
Fair, Degrading	10%	30%
Poor, Minimum	10%	10%
Severe Degradation	<10%	<10%

The determination of average annual flow was conducted by Tennant by the summation of the average monthly flow for a ten year period. After average annual flows are determined, recommendations can be calculated by multiplying the average annual flow by the percentages in the above table.

INSTREAM FLOW INCREMENTAL METHODOLOGY (IFIM)

The IFIM is a computerized water management tool developed by the U.S. Fish and Wildlife Service for evaluating changes on aquatic life and recreational activities resulting from alterations in channel monphology, water quality and hydraulic components. Bovee (1982) outlined the underlying principles of IFIM as: (1) each species exhibits preferences within a range of habitat conditions that it can tolerate; (2) these ranges can be defined for each species; and (3) the area of stream providing these conditions can be quantified as a function of discharge and channel structure. IFIM is designed to simulate hydraulic conditions and habitat availability for a particular species and size class or usable waters for a particular recreational activity. The hydraulic and channel characteristics are simulated for IFIM by use of the Physical Habitat Simulation Model (PHABSIM).

PHABSIM is a series of computer programs which relate changes in flow and channel structure to changes in physical habitat availability. Hilgart (1982) summarized the PHABSIM model as comprised of two parts: (1) a hydraulic simulation program which will predict the values of hydraulic parameters for a range of flows from either a single measured flow (WSP) or two or more measured flows (IFG4), and (2) a habitat assessment program called HABTAT, which rates the predicted hydraulic conditions for their relative fisheries values. Rather than describing the stream reach as a series of depth, velocity and substrate contours, PHABSIM is used to describe the reach as a series of small cells (Figure II-1-1).

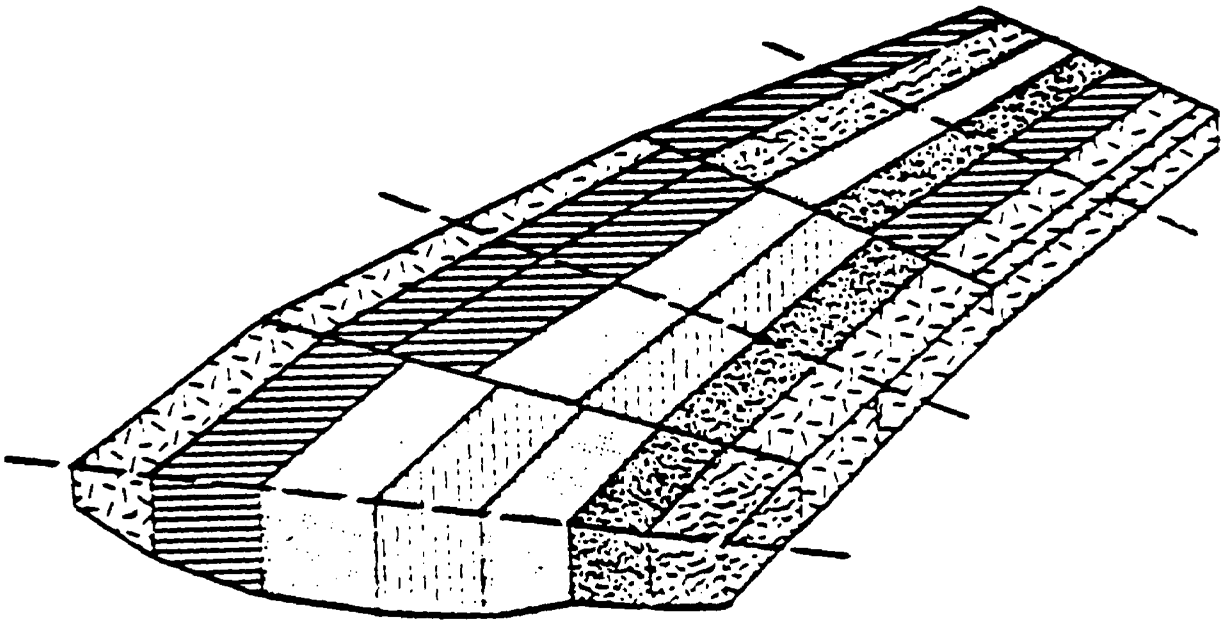


Figure II-1-1: Conceptualization of Simulated Stream Reach. Shaded Subsections Have Similar Depth and Velocity Ranges.

Instead of summarizing average depth and velocity for a cross section, PHABSIM is used to predict the average depth and velocity for each cell. Using curves showing the relative suitability of various stream attributes by species and life stage, a weighting factor for the depth, velocity, and substrate in each cell is determined. These weighting factors are multiplied together to estimate the composite suitability for that combination of variables, and this composite index is multiplied by the surface area of the cell. This process is repeated for each cell and the results are summed to calculate the total weighted usable area. Computer simulations are then produced for the distribution of microhabitat variables for existing and alternate flows, e.g., flows for a proposed and alternative actions which could affect flow regime.

The basic steps to IFIM can be summarized by the following:

STEP 1: Project Scoping - Scoping involves defining objectives for the delineation of study area boundaries, determining the stability of the microhabitat variables, selecting evaluation species, and defining their life history, food types, water quality tolerances and microhabitat.

STEP 2: Study Reach and Site Selection - Involves identifying and delineating critical reaches to be sampled, delineation of major changes and transition zones and the distribution of the evaluation species.

STEP 3: Data Collection - Transects are selected to adequately characterize the hydraulic and instream habitat conditions. Data gathering must be compatible to IFIH computer models.

STEP 4: Computer Simulation - Involves reducing field data and entering into programs described above.

STEP 5: Interpretation of Results - The output from the computer program is expressed as the Weighted Usable Area (WUA), a discrete value for each representative and critical study reach, for each life stage and species, and for each flow regime.

For further information on IFIM and PHABSIM the following publication should be consulted: "A Guidance to Stream Habitat Analysis Using the Instream Flow Incremental Methodology" U.S. FWS/OBS-82/26, June, 1982.

CHAPTER II-2
SUSPENDED SOLIDS AND SEDIMENTATION

The consideration of the potential effects of suspended solids and sedimentation on aquatic organisms may reveal important data and information pertinent to a use attainability analysis. Suspended solids generally may affect fish populations and fish in several major ways:

(1) "By acting directly on the fish swimming in water in which solids are suspended, and either killing them or reducing their growth rates, resistence to disease, etc.;

(2) By preventing the successful development of fish eggs and larvae;

(3) By modifying natural movements and migrations of fish; and

(4) By reducing the abundance of food available to the fish" (EIFAC, 1964).

(5) By hindering the foraging and mating abilities of visual feeders and those with visual mating displays.

The effects of sedimentation on aquatic organisms were summarized by Iwamoto et al. (1978). These effects include:

(1) clogging and abrasion of respiratory surfaces, especially gills;

(2) adhering to the chorion of eggs;

(3) providing conditions conducive to the entry and persistence of disease-related organisms;

(4) inducing behavioral modifications;

(5) entomb different life stages;

(6) altering water chemistry by the absorption and/or adsorption of chemicals;

(7) affecting utilizable habitat by the scouring and filling of pools and riffles and changing bedload composition;

(8) reducing photosynthetic growth and primary production, and;

(9) affecting intragravel permeability and dissolved oxygen levels.

This chapter of the manual will explore these effects in detail. An excellent review of the effects of suspended solids and sedimentation on warmwater fishes was conducted by EPA in 1979 entitled "Effects of Suspended Solids and Sediment on Reproduction and Early Life of Warmwater Fishes" (EPA-600-3-79-042) and should be consulted.

GENERAL ECOSYSTEM EFFECTS

Suspended solids and sedimentation may affect several trophic levels and components of the ecosystem. The interactions between components of the ecosystem are closely linked thus changes in one component can reverberate throughout the system. The following examines changes in each component resulting from suspended solids and sedimentation:

Influences on Primary Productivity

Increases in suspended solids can greatly alter primary productivity because of decreasing light penetration and subsequently decreasing photosynthetic activity. Cairns (1968) reviewed the literature on the effects on primary producers. The decrease in light penetration can affect the depth distribution of vascular aquatic plants and algae. Greatly reduced light penetration may shift algal composition from green to bluegreen since the latter are tolerant to higher levels of ultraviolet light. Butler (1964) observed an inverse relationship between turbidity and primary productivity; gross primary productivity in a clear pond was three-fold greater than an adjacent turbid pond (with Permian red clay). Benson and Cowell (1967) found that turbidity in Missouri River impoundments was the strongest limiting factor to plankton abundance and that plankton was of great importance to fish growth and survival.

Suspended solids can also alter the distribution of heat in a water body. Butler (1963) reported that colloidal clay in central Oklahoma was altering the heat distribution and consequently summer stratification was more pronounced in turbid situations. This stratification causes greater differences between the surface and bottom temperature in turbid water bodies.

To protect against the deleterious effects of suspended solids on aquatic life by decreasing photosynthetic activity, EPA (1976) developed the following criteria: "Settleable and suspended solids should not reduce the depth of the compensation point for photosynthetic activity by more than 10 percent from the seasonally established norm for aquatic life." The compensation point is the point at which incident light penetration is sufficient for plankton to photosynthetically produce enough oxygen to balance their respiration requirements. To determine this compensation point, a set of "light" bottle D.O. and "dark" bottle D.O. tests would be needed (see "Standard Methods", APHA, 1979 for details).

Effect on Zooplankton and Benthos

Benthic macroinvertebrates and zooplankton are major sources of food for fish which can be adversely affected by suspended matter and sediment. Depopulation and mortality of benthic organisms occurs with smothering or alteration of preferred habitats. Zooplankton populations may be reduced via decreasing primary productivity resulting from decreased light penetration. Ellis (1936) demonstrated that freshwater mussels were killed in silt deposits of 6.3 to 25.4 mm of primarily adobe clay. Major increases in stream suspended solids (25 ppm turbidity upstream vs. 390 ppm downstream) caused smothering of bottom invertebrates, reducing organism diversity to only 7.3 per square foot from 25.5 per square foot upstream (Tebo, 1955). Deposition of organic materials to bottom sediments can also cause imbalances in stream biota by increasing bottom animal density, principally oligochaete populations, and diversity is reduced as pollution sensitive forms disappear. Deposition of organic materials can also cause oxygen depletion and a change in the composition of bottom organisms. Increases in oligochaetes and midges may occur since certain species in these groups are tolerant of severe oxygen depletion.

Sensitivity of Fish Populations to Suspended Solids and Sediment

Field and laboratory studies have shown that fish species vary considerably in their population-level responses to suspended solids and sediment. Atchison and Menzel (1979) reviewed the population level effects on warmwater species and categorized species as either tolerant or intolerant based on their habitat preferences. This review also revealed species with a preference for turbid systems. Tables 1 and 2 have been adapted from this review and provide valuable information on population effects. As can be seen from these tables, the intolerant assemblage is composed of a large number of species with complex spawning behavior whereas the tolerant fishes include a larger percentage of simple spawners and forms with special early life adaptations for turbid waters.

Effects on Fish Reproduction

The impacts of suspended solids and sediments on fish reproduction vary with the phases of the reproductive cycle. The following describes several of the mechanisms of impairment:

(1) Diminished Light Penetration
Swingle (1956) provided data which shows that suspended materials might affect fish reproductive processes by reducing light penetration. He found that largemouth bass spawning was delayed by as much as 30 days in muddy ponds as compared to clear ponds.

(2) Visual Interference
Some species such as black bass and centrarchid sunfish have strong visual components in their reproductive behavior. For example, Trautman (1957) found that smallmouth bass populations in Lake Erie shunned potential spawning areas that were highly turbid. Chew (1969) observed that in turbid Lake Hollingsworth (Fla.) largemouth bass spawning was very limited and that most females failed to shed their eggs and gradually resorbed them.

(3) Loss of Spawning Habitat
Reproductive failure among many species is attributable to direct loss of spawning habitat through two pathways: (a) siltation of formerly clean bottom and (b) loss of vegetation due to the reduction of the photic zone by turbidity.

(4) Physiological Alterations
The major physiological alterations are:

(a) the failure of gonadal maturation at the appropriate time and (b) stress incurred by the organism thus creating increased susceptability to disease.

In general, laboratory bioassays indicate that larval stages of selected species are less tolerant of suspended solids than eggs or adults. Available evidence suggests that lethal levels for suspended solids are determined by interaction between biotic factors, including age-specific and species specific differences, and abiotic factors such as particle size, shape, concentration and amount of turbulence in the system.

TABLE 1: SELECTED MIDWESTERN WARMWATER FISHES WHICH ARE INTOLERANT OF SUSPENDED SOLIDS (TURBIDITY) AND SEDIMENT

Species	Effect		Impact through	
	Spawning	General	Suspended solids	Sediment
Ichthyomyzon castaneus - Chestnut lamprey	X			X
Acipenser fulvescens - Lake Sturgeon	X	X		X
Polyodon spathula - Paddlefish	X	X		X
Lepisosteus platostomus - Shortnose gar		X		X
Amia calva - Bowfin	X		X	
Hiodon tergisus - Mooneye		X	X	
Esox lucius - Northern pike	X		X	X
Esox masquinongy - Muskellunge		X	X	
Clinostomus elongatus - Redside dace		X		X
Dionda nubila - Minnow		X		X
Exoglossum laurae - Tonguetied minnow		X		X
Exoglossum maxillingua - Cutlips minnow		X		X
Hybopsis amblops - Bigeye chub		X	X	X
Hybopsis dissimilis - Streamline chub		X		X
Hybopsis x-punctata - Gravel chub		X		X
Nocomis biguttatus - Horneyhead chub	X			X
Nocomis micropogon - River chub		X	X	X
Notropis amnis - Pallid shiner		X	X	
Notropis boops - Bigeye shiner		X	X	X
Notropis cornutus - Common shiner		X		X
Notropis emiliae - Pugnose minnow		X	X	X
Notropis heterodon - Blacknose shiner		X	X	X
Notropis heterolepis - Blacknose shiner		X	X	
Notropis hudsonius - Spottail shiner		X	X	X
Notropis rubellus - Rosyface shiner		X	X	X
Notropis stramineus - Sand shiner		X	X	X
Notropis texanus - Weed shiner		X		X
Notropis topeka - Topeka shiner		X		X
Notropis volucellus - Mimic shiner		X	X	
Carpiodes velifer - Highfin carpsucker		X	X	
Cycleptus elongatus - Blue sucker		X		X
Erimyzon oblongus - Creek chubsucker		X	X	
Erimyzon sucetta - Lake chubsucker		X	X	X
Hypentelium nigricans - Northernhog sucker		X	X	X
Lagochila lacera - Harelip sucker		X	X	X
Minytrema melanops - Spotted sucker		X	X	
Moxoxtoma carinatum - River redhorse		X	X	X
Moxostoma duquesnei - Black redhorse		X	X	X
Moxostoma valenciennesi - Greater redhorse		X	X	X
Ictalurus furcatus - Blue Catfish		X	X	X

TABLE 1: SELECTED MIDWESTERN WARMWATER FISHES WHICH ARE INTOLERANT OF SUSPENDED SOLIDS (TURBIDITY) AND SEDIMENT (Cont'd)

Species	Effect: Spawning	Effect: General	Impact through: Suspended solids	Impact through: Sediment
Etheostoma blennioides - Greenside darter		X		X
Etheostoma exile - Iowa darter		X	X	
Etheostoma tippecanoe - Tippe canoe darter		X		X
Etheostoma zonale - Banded darter		X		X
Perca flavescens - Yellow perch	X	X	X	X
Percina caprodes - Log perch		X	X	X
Percina copelandi - Channel darter		X	X	X
Percina evides - Gilt darter		X	X	X
Percina maculata - Blackside darter		X	X	
Percina phoxocephala - Slenderhead darter		X	X	X
Noturus flavus - Stonecat		X		X
Noturus furiosus - Caroline madtom		X	X	
Noturus gyrinus - Tadpole madtom		X	X	X
Nocturus miurus - Brindled madtom		X	X	
Nocturus trautmani - Scioto madtom		X	X	X
Pylodictis olivaris - Flathead catfish		X	X	X
Percopsis omiscomaycus - Trout perch		X		X
Fundulus notatus - Blackstripe topminnow		X	X	
Labidesthes sicculus - Brook silverside		X	X	
Culaea inconstans - Brook stickleback		X	X	
Ambloplites rupestris - Rock bass		X	X	
Lepomis gibbosus - Pumpkin seed		X	X	X
Lepomis megalotus - Longear sunfish		X	X	
Micropterus dolomieui - Smallmouth bass	X	X	X	X
Micropterus salmoides - Largemouth bass		X	X	X
Ammocrypta asprella - Crystal darter		X		X
Ammocrypta clara - Western sand darter		X		X
Ammocrypta pellucida - Eastern sand darter		X		X

TABLE 2: WARMWATER FISHES WHICH ARE TOLERANT OF SUSPENDED SOLIDS AND SEDIMENT

Species	General tolerance	Preference for turbid systems
Scaphirhynchus albus - Pallid sturgeon	X	
Dorosoma cepedianum - Gizzard shad		X
Hiodon alosoides - Goldeye	X	
Carassius auratus - Goldfish	X	
Couesius plumbeus - Lake chub	X	
Cyprinus carpio - Common Carp		X
Ericymba buccata - Silverjaw minnow	X	X
Hybopsis gelida - Sturgeon chub	X	
Hybopsis gracilis - Flathead chub	X	
Notropis dorsalis - Bigmouth shiner		X
Notropis lutrensis - Red shiner		X
Orthodon microlepidotus - Sacramento blackfish	X	
Phenacobius mirabilis - Suckermouth minnow	X	
Phoxinus oreas - Mountain redbelly dace	X	
Pimephales promelas - Fathead minnow	X	X
Pimephales vigilax - Bullhead minnow	X	
Plagopterus argentissimus - Woundfin	X	
Semotilus atromaculatus - Creek chub	X	
Catostomus commersoni - White sucker	X	
Ictiobus cyprinellus - Bigmouth buffalo	X	
Moxostoma erythrurum - Golden redhorse	X	
Ictalurus catus - White catfish	X	
Ictalurus melas - Black bullhead	X	X
Aphredoderus sayanus - Pirate perch	X	
Lepomis cyanellus - Green sunfish	X	
Lepomis humilis - Orangespotted sunfish	X	
Lepomis microlophus - Redear sunfish	X	
Micropterus treculi - Guadalupe bass	X	
Pomoxis annularis - White crappie	X	
Pomoxis nigromaculatus - Black crappie	X	
Etheostoma gracile - Slough darter	X	
Etheostoma microperca - Least darter	X	
Etheostoma nigrum - Johnny darter	X	
Etheostoma spectabile - Orangethroat darter	X	
Stizostedion canadense - Sauger	X	
Aplodinotus grunniens - Freshwater drum	X	

CHAPTER II-3.
POOLS, RIFFLES AND SUBSTRATE COMPOSITION

AQUATIC INVERTEBRATES

Many factors regulate the occurrence and distribution of stream-dwelling invertebrates. The most important of these are current speed, shelter, temperature, the substratum (including vegetation), and dissolved substances. Other important factors are liability to drought and to floods, food and competition between species. Many of these factors are interrelated - current, for example, largely controls the type of substratum and consequently the amount and type of food available. Of these, current speed, the substratum, and the significance of riffle and pool areas will be discussed in greater detail in the following paragraphs.

Current Speed

Many invertebrates have an inherent need for current, either because they rely on it for feeding purposes or because their respiratory requirements demand it. However, persistently very rapid current may make life intolerable for almost all species. At the other extreme, stagnant or very slow areas in rivers which at time flow swiftly are often without much fauna. This is because silt collects during periods of low discharge, and the conditions become unsuitable for riverine animals. On the other hand, many common stream creatures (e.g. flatworms, annelids, crustaceans, and a great number of the insects) persist in running water simply because they avoid the current by living under stones or in the dead water behind obstructions. Still other animals which are poor swimmers and lack attachment mechanisms and therefore can only scuttle from one shelter to another select areas where the current is tolerable, and move further out or back into shelter as the flow varies. This applies to many genera of mayflies and to snails. Other animals actually burrow down into the substratum to avoid the current and require only to remain buried. Many animals, such as the annelids and some Diptera larvae, have this habit as a birthright; several other groups have acquired this habit, such as several genera and species of stoneflies and mayflies. Similarly, as the current changes from place to place in a stream at a given discharge so the fauna changes.

In conclusion, current speed is a factor of major importance in running water. It controls the occurrence and abundance of species and hence the whole structure of the animal community.

The Substratum and Its Effect On Aquatic Invertebrates

The substratum is the material (including vegetation) which makes up the streambed. It is true of many river systems that the further down a river the smaller the general size of the particles forming the bed. This is partly due to the fact that the shear stress on the bottom and hence the power to move (and break up) particles decreases with increasing discharge. In streams where current speeds do not normally exceed about 40 cm/sec a streambed is likely to be sand, or even silt at still lower maximum currents of about 20 cm/sec. However, large amounts of silt occur only in backwaters and shallows or as a temporary thin sheet over sand during periods of low flow; silt is certainly not a major component of the substratum in the main channels of the great majority of even base-level rivers. Where currents frequently exceed about 50 cm/sec on steep slopes the bed is likely to be stony and the animals which live there must be able to maintain their position.

The substratum is the major factor controlling the occurrence of animals and there is a fairly sharp distinction between the types of fauna found on hard and on soft streambeds. In general, clean and shifting sand is the poorest habitat with few specimens of few species. Bedrock, gravel and rubble on the one hand and clay and mud on the other, especially when mixed with sand, support increasing biomasses. The fauna of hard substrata has its own typical character, and it is here that most of the obviously specialized forms occur; that of the soft substrata is more generally shared with still water, and it shows much more geographical variety.

The fact that rubble supports more animals than does sand is almost certainly correlated with the amount of living space (shelter) and with the greater probability that organic matter will lodge among stones and provide food.

Another factor affecting the occurrence of fauna in the substratum is the temporary nature of some types of substratum themselves. For example, stony areas can be alternately covered with silt or sand and then cleared away by spring floods (spates). Streams that are more liable to spates or other similar phenomena (which greatly and rapidly alter the faunal density) have less abundant and less varied faunas than others. An interesting consequence of this is that small tributaries, being less exposed to the effects of storms covering limited areas, are richer than the larger streams into which they flow. Another consequence is that as development increases the intensity of runoff, the variety and abundance of stream fauna also decreases.

The presence of solid objects also affects the fauna, and the nature of the solid object affects the animals which colonize it. As shelter is more important, some animals prefer irregular stones as opposed to smooth ones. Still other animals occur only on wood.

Other factors which may account for differences of invertebrate biomass in streams or reaches of streams are the differences in plant detritus and in vegetation on the banks, which, of course, supplies food to the biota. Both the amounts and the nature of the deposits and the vegetation are important. In any case there are more animals in moss, rooted plants, and filamentous algae than there are on stones, and all plants are more heavily colonized than the nonvegetated areas of substratum.

Finally, the availability of food (whether it be organic detritus lodged amongst stones, vegetation, wood. . .) is an obvious factor controlling the abundance of species. Generally speaking species occur, or are common, only where their food is readily available, but it should not be forgotten that few running water invertebrates are very specialized in their diets.

It seems appropriate at this time to restate the three ecological principles of Theisemann (Hynes, 1970) which summarize the implications of the foregoing discussion. They are:

- The greater the diversity of the conditions in a locality the larger is the number of species which make up the biotic community.
- The more the conditions in a locality deviate from normal and hence from the normal optima of most species, the smaller is the number of individuals of each of the species which do occur.
- The longer a locality has been established in the same condition the richer is its biotic community and the more stable it is.

In conclusion, it can be stated that the fauna of clean, stable, diverse stony runs is richer than that of silty reaches and pools both in number of species and total biomass.

As previously discussed, certain species are confined to fairly well-defined types of substraum, and others are at least more abundant on one type than they are on others. The result of these preferences is that as the type of substratum varies from place to place so does the fauna. In general, the larger the stones, and hence the more complex the substratum, the more diverse is the invertebrate fauna.

The following groups of invertebrates almost invariably provide the major constituents of the fauna of stony streams:

- Parazoa
- Cnidaria
- Tricladida
- Oligochaeta
- Gastropoda
- Pelecypoda
- Peracarida
- Eurcarida
- Plecoptera
- Odonata
- Ephemeroptera
- Hemiptera
- Megaloptera
- Trichoptera
- Lepidoptera
- Coleoptera
- Diptera

The fauna of the softer substrata in rivers is much less evident than that of the hard substrata. However, there are still many genera of invertebrates such as Limnaea, Chironomus, Tubifex, and Limnodrilus which can be found in rivers in most continents, but the less-rigorous habitat of areas of slower current which allows less-specialized species to occur also permits the local character of the fauna to be dominant.

It is therefore difficult to generalize, but characteristic organisms of soft riverine substrata are: Tubificidae, Chironomidae, burrowing mayflies (Ephemeridae, Potomanthidae, Polymitarcidae), Prosobranchia, Unionidae, and Sphaeriidae, and when plants are present a great variety of organisms may be added.

Riffle/Pool Areas

Natural streams tend to have alternating deep and shallow areas - pools and riffles - especially where there are coarse constituents in the substratum. Riffles tend to be spaced at more or less regular distances of five to seven stream widths apart and to be most characteristic of gravel-bed streams. They do not naturally form in sandy streams, since their presence seems to be connected with some degree of heterogeneity of particle size. Riffles are formed when the larger particles (boulders, stone and gravel) congregate on bars.

The reasons for the regular spacing of riffles is unknown; however, it is known that riffles do not move, although the stones that compose them may migrate downstream, being replaced by others. Furthermore, it has been established that riffles are superficial features with the largest stones in the upper layer.

Pools tend to be wider and deeper than the average stream course. In contrast to the broken surface of riffles, the surface of a pool or backwater is smooth. In pools, the current is reduced, a little siltation may occur, and aquatic seed plants may form beds. The significance of riffle/pool areas to the production potential of aquatic invertebrates has been alluded to in the previous discussions of the current speed and the substratum. One result of the complex interaction of local factors on faunal density is that in streams with pool and riffle structure, the fauna is considerably denser on the latter. Similarly, aquatic invertebrates are most diverse in riffle areas with a rubble substrate. As a consequence the amount of drift produced by riffles is greater than that produced by pools.

FISHES

Like the invertebrates, there are many factors which regulate the occurrence and distribution of running water fishes. The most important of these are the substratum, food availability, cover, current speed, and the presence of a suitable spawning habitat. All of these are directly related to the distribution of pool/riffle areas in a stream, and for most fishes a 1:1 ratio of pool to riffle run areas is sufficient for successful propagation and maintenance. The significance of the substratum (type and amount), and the presence of both pools and riffle areas will be discussed in greater detail in the following paragraphs. Finally, the specific habitat requirements of several fish species (including black & white crappie, channel catfish, cutthroat trout, creek chub and bluegill) will be discussed in order to illustrate the importance of the substratum and the pool/riffle structure and to indicate the similarities and differences in requirements between species.

The Substratum and Its Effect on Fishes

A few fishes, particularly small benthic species, are more or less confined to rocky or stony substrata. These include all those with ventral suckers and friction plates (e.g. some species of darters). Many others are also fairly definitely associated with a specific type of substratum. For example, the gudgeon is associated with gravel, the sand darter with sand, and the mudfish with thick marginal vegetation.

For the great majority of fish species, however, the nature of the substratum is apparently of little consequence except at times of breeding. Nearly all species of fish have fairly well-defined breeding habits and requirements. The great majority of freshwater fishes spawn on a solid surface (such as a flat area under a large stone) in stoney or gravel substrata. Other species dig pits in gravel (e.g. the stoneroller) in which the eggs are laid. This requires that the gravel be a suitable size and be relatively free of silt and sand. Still other species make piles of pebbles (e.g. some chubs and minnows) through which water passes freely bringing oxygen to the buried eggs. Some species of trout and Atlantic salmon select places for spawning where there is a down-flow of water, say at the downstream end of pools, where the water flows into riffles. In summary, species which construct nests (see Table II-3-1) or redds are restricted not only in respect of the size of the material of the substratum, which they must be able to move, but by the need to be free of silt; and salmonids, and probably some other fishes, are also restricted to places where there is a natural intra-gravel flow of water.

On the other hand, there are a great many species (e.g. the whitefish, sterlet, grayling, etc.) which breed on gravel or stones but build no nests. In fact, this is probably the most common pattern of breeding among running-water species. Table II-3-2 is a partial list of fish species (which build no nests) along with their desired spawning habitat. The fishes which breed in this manner move onto the clean gravel in swifter and shallower water than is their normal adult habitat to spawn.

There are also those species which spawn on other substrata besides stones and gravel, including sand (e.g. the log-perch), mud (e.g. the Murray cod), and vegetation (e.g. some species of darters and most still-water species).

Finally, there are many riverine species (e.g. grass carp, some perch species) which lay buoyant or semi-buoyant eggs which float in the water and are carried downstream while they develop.

In conclusion, it can be seen from the previous discussion that breeding habitat requirements for fishes can be very restrictive, and consequently, the

TABLE II-3-1. EXAMPLES OF NEST-BUILDING FISH

Species	Type of Nest
Sticklebacks (Gasterosteidae) Largemouth Bass (Micropterus salmoides) Crappies (Pomoxis) Rock Basses (Ambloplites) Warmouth (Chaenobryttus) Bluegill (Lepomis macrochirus) Most Bullheads (Ictalurus)	Nest a circular depression in mud, silt, or sand and often in and among roots of aquatic flowering plants
Smallmouth Bass (Micropterus dolomieu) Trouts (Salmo) Stoneroller (Campostoma anomalum) Brook Trout (Salvelinus fontinalis)	Nest a circular depression in gravel
Creek Chubs (Semotilus) Bluntnose & Fathead Minnows (Pimephales)	Nest a pile of pebbles

TABLE II-3-1. EXAMPLES OF FISH THAT DO NOT BUILD NESTS

Species	Spawning Habitat
Northern Pike (Esox lucius) Carp (Cyprinus carpio) Goldfish (Carassium auratus) Golden Shiner (Notemigonus crysoleucas)	Scattering eggs over aquatic plants, or their roots or remains
Whitefishes (Coregonus) Ciscos (Leucicthys) Lake Trout (Salvelinus namaycush) Log Perch (Percina caprodes) Suckers (Catostomus) Walleyes (Stizosstedion)	Scattering eggs over shoals of sand, gravel, or boulders
Yellow Perch (Perca flavescens) White Perch (Morone americana) Grass Carp (Ctenopharyngodon idellus) Brook Silverside (Labidesthes sicculus) Alewife (Alosa pseudoharengus) Siamese Fighting Fish (Betta)	Semi-buoyant or buoyant eggs
Bitterling (Rhodeus)	Eggs deposited in the mantle cavity of a freshwater mussel
Lumpsucker (Careproctus)	Eggs deposited beneath the carapace of the Kamchatka crab

suitable breeding sites can be extremely limited. Furthermore, the requirements can be extremely varied among species. However, the general breeding habitat requirements fall into the following categories:

- o Build a nest and breed on stone or gravel substrata.
- o Breed on stone or gravel substrata without building a nest.
- o Breed on other substrata, including sand, mud, or vegetation.
- o Lay buoyant or semi-buoyant drifting eggs and larvae.

Pool Areas

Pool areas in a stream are essential for providing shelter for both resting and protection from predation. To a lesser extent pools are important as a spawning habitat and for food production (although food production is lower in pools than in riffles).

Even the streamlined species that are well adapted to fast-flowing water (e.g. salmon and trout) need time to rest or seek shelter to avoid predators. As a matter of fact all fishes spend most of their time resting in shelters in lower velocity pool areas. Still other species (e.g. channel catfish, particularly adults) reside primarily in pool areas and generally move only to riffle areas at night to feed.

Therefore, based on the foregoing discussion, one must conclude that the existence of pools is critical to the well-being of all fish species, since they provide resting cover and protection from predators.

Riffle/Run Areas

As discussed previously in the section on benthic invertebrates and again in the section on the substratum and its affect on fishes, it is apparent that riffle areas are most important due to their food producing capability (i.e. benthic invertebrates) and their suitability as a fish spawning habitat (i.e. it is in riffle areas where the silt-free stone or gravel exists and where oxygen to the eggs is constantly being renewed). Without an abundant food supply and the proper spawning habitat, propagation and maintenance of a fish species would be impossible.

Species Examples

Bluegill (Lepomis macrochirus)

The bluegill is native from the Lake Champlain and southern Ontario region through the Great Lakes to Minnesota, and south to northeastern Mexico, the Gulf States, and the Carolinas.

Bluegills are most abundant in large low velocity (<10 cm/sec preferably) streams. Abundance has been positively correlated to a high percentage (>60%) of pool area and negatively correlated to a high percentage of riffle/run areas.

Cover in the form of submerged vegetation, logs, brush and other debris is utilized by bluegills. Excessive vegetation can influence both feeding ability and abundance of food by inhibiting the utilization of prey by bluegills.

Bluegills are guarding, nest building lithophils. Nests are usually found in quiet shallow water over almost any substrate; however, fine gravel or sand is preferred.

In summary, riffles and substrate play a small role in the life cycle of the bluegill. In fact, excessive riffle/run areas have been negatively correlated with an abundance of bluegills. On the other hand, pools are significant as the typical bluegill habitat for resting, feeding, and spawning.

Creek Chub (Semotilus atromaculatus)

The Creek Chub is a widely distributed cyprinid ranging from the Rocky Mountains to the Atlantic Coast and from the Gulf of Mexico to southern Manitoba and Quebec. Within its range, it is one of the most characteristic and common fishes of small, clear streams.

The optimum habitat for creek chubs is small, clear, cool streams with moderate to high gradients, gravel substrate, well-defined riffles and pools with abundant food, and cover of cut-banks, roots, aquatic vegetation, brush, and large rocks. Creek chubs are found over all types of substrate with abundance correlated more with the amount of instream cover than with the substrate type. It is assumed that stream reaches with 40-60% pools are optimum for providing riffle areas for spawning habitat and pools for cover.

Rubble substrate in riffles, abundant aquatic vegetation, and abundant streambank vegetation are conditions associated with high production of food types consumed by creek chubs.

Spawning occurs in gravel nests constructed by the male in shallow areas just above and below riffles to insure a good water exchange rate through the creek chub redds. Reproductive success of creek chubs varies with the type of spawning substrate available. Production is highest in clean gravel substrate in riffle-run areas with velocities of 20-64 cm/sec. Production is negligible in sand or silt.

In summary, pools, riffles and substrate are important to the creek chub in the following manner.

1) Riffles - provide a suitable spawning habitat,
2) Substrate - a clean gravel substrate is required for spawning, and
3) Pools - provide resting cover and abundant food.

White Crappie (Pomoxis annularis)

The white crappie is native to freshwater lakes and streams from the southern Great Lakes, west to Nebraska, south to Texas and Alabama, east to North Carolina, then west of the Appalachian Mountains to New York. It has been widely introduced outside this range throughout North America.

White crappie are most numerous in base-level low gradient rivers preferring low velocity areas commonly found in pools, overflow areas, and backwaters of rivers. In these areas, cover is important for providing resting areas and protection from predation. Cover also provides habitat for insects and small forage fish, which are important food for the crappie. In addition, cover is important during reproduction as the male white crappie constructs and guards nests over a variety of substrates almost always near vegetation or around submerged objects.

In summary, riffles and substrate composition are for the most part insignificant to the white crappie. However, pools are important for resting, feeding, spawning and providing protection from predation.

Channel Catfish (Ictalurus punctatus)

The native range of channel catfish extends from the southern portions of the Canadian prairie provinces south to the Gulf States, west to the Rocky Mountains, and east to the Appalachian Mountains. They have been widely introduced outside this range and occur in essentially all of the Pacific and Atlantic drainages in the 48 contiguous states.

Optimum riverine habitat for the channel catfish is characterized by a diversity of velocities, depths and structural features that provide cover and food. Low velocity (<15 cm/sec) areas of deep pools and littoral areas and backwaters of rivers with greater than 40 percent suitable cover are desirable. Riffle and run areas with rubble substrate, pools, and areas with debris and aquatic vegetation are conditions associated with high production of aquatic insects consumed by channel catfish. A riverine habitat with 40-60% pools would be optimum for providing riffle habitat for food production and feeding and pool habitat for spawning and resting cover.

Adult channel catfish in rivers are found in large, deep pools with cover. They move to riffles and runs at night to feed. Catfish fry have strong shelter-seeking tendencies and cover availability is important in determining habitat suitability. However, dense aquatic vegetation generally does not provide optimum cover because predation on fry by centrarchids is high under these conditions.

Dark and secluded areas are required for nesting. Males build and guard nests in cavities, burrows, under rocks and in other protected sites.

In summary, the presence of riffles and pools are equally important to the successful propagation of channel catfish, with riffles providing a suitable habitat for food production and feeding and with pools providing a suitable habitat for spawning and resting. Additionally, channel catfish appear to be relatively insensitive to variations in the substrate type.

Cutthroat Trout (Salmo clarki)

Cutthroat trout are a polytypic species consisting of several geographically distinct forms with a broad distribution and a great amount of genetic diversity.

Optimal cutthroat trout riverine habitat is characterized by clear, cold water; a silt free rocky substrate in riffle-run areas; an approximately 1:1 pool/riffle ratio with areas of slow, deep water; well vegetated stream banks; abundant instream cover; and relatively stable water flow, temperature regimes and stream banks. A 1:1 ratio (40-60% pools) of pool to riffle area appears to provide an optimal mix of trout food producing and rearing areas.

Cover is recognized as one of the essential components of trout streams. Cover is provided by overhanging vegetation; submerged vegetation, undercut banks and instream objects. The main use of this cover is predator avoidance and resting.

Conditions for spawning require a gravel substrate with $\leq$ 5% fines. Greater than 30% fines will result in a low survival rate of embryos. Optimal substrate size averages 1.5 - 6.0 cm in diameter; however, gravel size as small as 0.3 cm in diameter is suitable for incubation.

Black Crappie (Pomoxis nigromaculatus)

The black crappie is native to freshwater lakes and streams from the Great Lakes south to the Gulf of Mexico and the southern Atlantic States, north to North Dakota and eastern Montana and east to the Appalachians.

Black crappie are common in base or low gradient streams of low velocities, preferring quiet, sluggish rivers with a high percentage of pools, backwaters, and cut-off areas. Black crappie prefer clear water and grow faster in areas of low turbidity.

Abundant cover, particularly in the form of aquatic vegetation, is necessary for growth and reproduction. Common daytime habitat is shallow water in dense vegetation and around submerged trees, brush or other objects.

Conclusions

In conclusion, a review of the substratum and its effects on benthic invertebrates and fishes reveals that the invertebrates are dependent on a suitable substrata for growth, successful reproduction, and maintenance, and the fishes are dependent on a suitable substrata primarily only during breeding. With the proper substrata, an adequate supply of benthic invertebrates is available as food for the fishes.

Similarly, it is the proper balance between pools and riffles (approximately 1:1 ratio) that will insure an abundant food supply for both invertebrates and fishes, the existence of the proper habitat for reproduction of both invertebrates and fishes, and adequate cover for resting and protection from predation.

CHAPTER II-4

CHANNEL CHARACTERISTICS AND EFFECTS OF CHANNELIZATION

INTRODUCTION

Channelization can be defined as modification of a stream system - including the stream channel, stream bank, and nearstream riparian areas - in order to increase the rate of drainage from the land and conveyance of water downstream. Simpson et al. (1982) listed the common methods of channelization as:

1. Clearing and Snagging. Removal of obstructions from the streambed and banks to increase the capacity of a system to convey water. Such operations include removal of bedload material, debris, pilings, head walls, or other manmade materials.

2. Rip-rapping. Placement of rock or other material in critical areas to minimize erosion.

3. Widening. Increase of channel width to improve the conveyance of water and increase the capacity of the system.

4. Deepening. Excavation of the channel bottom to a lower elevation so as to increase the capacity to convey water or to promote drainage or lowering of the water table, or to enhance navigation.

5. Realignment. Construction of a new channel or straightening of a channel to increase the capacity to convey water.

6. Lining. Placement of a nonvegetative lining on a portion of a channel to minimize erosion or increase the capacity of a stream to convey or conserve water.

Channelization projects are classified according to their magnitude as either short-reach or long-reach. Short-reach channelization is associated with road and bridge construction and may entail 0.5 km of stream length within the vicinity of the crossing. Although short-reach projects may adversely affect stream biota, they should not produce significant long-term impacts with proper mitigation (Bulkley et al. 1976). The comments in this chapter generally refer to the effects of long-reach channelization; those impacts are greater in duration, dimension, and severity. Simpson et al. (1982) listed the purposes of (long-reach) channelization as:

1. Local flood control to prevent damage to homes, industrial areas, and farms on the flood plain by increased stream conveyance of water past the protected areas;

2. Increase of arable land for agriculture by channel straightening, deepening, and widening to remove meanders, increase channel capacity, and lower the channel bed. Straightening reduces the stream area and length of bordering lands, increases land area at cutoffs, and increases flow velocity. Deepening and widening increases channel capacity and improves drainage from adjacent lands;

3. Increased navigability of waterborne commerce and recreational boating, usually performed in large streams; and

4. Restoration of hydraulic efficiency of streams following unusually severe storms.

In the interest of such goals, several thousand miles of streams in the United States have been altered over the past 150 years (Simpson et al. 1982). However, in achieving these goals, detrimental effects are often incurred on water quality and stream biota. This chapter addresses the effects of channelization on stream characteristics and the associated biological impacts.

CHARACTERISTICS OF THE STREAM SYSTEM

Stream Depth and Width

The depth and width of a stream are usually made uniform (generally by widening and deepening) by stream channelization in order to increase the hydraulic efficiency of the system. This practice results in a monotony of habitats throughout the modified reach. Gorman and Karr (1978) demonstrated the direct relationship that exists between habitat diversity (considering depth, substrate, and velocity) and fish species diversity. Alteration of stream depth involves the disturbance and removal of natural bottom materials. Increasing stream depth can lower the water table of the area. Probably the most significant impact of depth modification is the disruption of the run-riffle-pool sequence (See Chapter II-3: Pools, Riffles, and Substrate Composition). Widening a stream increases the surface area and often involves removal of streamside vegetation. These practices increase the amount of light received by the water column and can lead to changes in the productivity and trophic regime of the system. Increasing and regularizing stream width also may reduce the proportion of bank/water interface, which constitutes important wildlife habitat.

Stream Length

Stream channelization usually involves realignment of the stream channel in order to convey water more quickly out of the modified reach. By straightening a stream its overall length is decreased. Channelized streams have been shortened an average of 45 percent (ranging from 8 to 95 percent) in Iowa (Bulkley 1975) and approximately 31 percent in Southcentral Oklahoma (Barclay 1980). Shortening the linear distance between two points with a constant change in elevation increases the slope or gradient of the stream, causing a corresponding increase in current velocity. Reducing the time required for a given parcel of water to flow through a stream segment may lower the capacity of the stream to assimilate wastes and increase the organic loading on downstream reaches.

The obvious effect of reducing stream length is the loss of living space. Stream segments that are isolated by channelization eventually become eutrophic and fill with sediment (Winger et al. 1976), and their function is severely impaired. In these eutrophic habitats, normal stream benthos, especially mayflies, stoneflies, caddisflies, and hellgramites, are replaced by tolerant chironomids and oligochaetes (Hynes 1970).

In addition to the loss of total living space, the amount of valuable edge habitat is decreased by stream straightening. Fish are habitat specialists (Karr and Schlosser 1977) and are not found uniformly distributed throughout the water column. Most fish and macroinvertebrate species utilize cover in lotic systems, much of which is associated with the sloping stream bank.

Channel Configuration

A stream is straightened by cutting a linear channel that eliminates natural bends (meanders) from the main course of flow. Sinuosity is a measure of the degree of meandering by a stream and is measured as the ratio of channel length to linear length or down-valley distance (Leopold et al. 1964). Sinuosity index values may range from 1.0 for a straight conduit to as high as 3.5 for mature, winding rivers (Simpson et al. 1982). A high gradient mountain stream may have a sinuosity index of 1.1, while a value of 1.5 or greater justifies designation as a meandering stream (Leopold et al. 1964).

Channelization (straightening) decreases sinuosity. Reducing sinuosity decreases the total amount of habitat available to biota as well as the amount of effective and unique habitat. Zimmer and Bachman (1976, 1978) found that habitat diversity was directly related to the degree of meandering in natural and channelized streams in Iowa, and that as sinuosity increased the biomass and number of organisms in the macroinvertebrate drift increased. Drift of benthic invertebrates is a major food source of fish.

The S-shaped meanders commonly observed in streams serve as a natural system of dissipating the kinetic energy produced by water moving downstream (Leopold and Langbein 1966). When a stream is straightened the energy is expended more rapidly, resulting in increased scour during high-flow periods.

Bedform

Bedform, or vertical sinuosity, is a measure of riffle-pool periodicity and is expressed in terms of the average distance between pools measured in average stream widths for the section (Leopold et al. 1964). Leopold et al. (1964) reported that natural streams have a riffle-pool periodicity of five to seven stream widths. This is variable, however, and is dependent on gradient and geology (as is horizontal sinuosity). Channelization eliminates or reduces riffle-pool periodicity (Huggins and Moss 1975, Lund 1976, Winger et al. 1976, Bulkley et al. 1976, Griswold et al. 1978).

Disruption of the run-riffle-pool sequence has detrimental consequences on macroinvertebrate and fish populations. Creating a homogeneous bedform drastically reduces habitat diversity and leads to shifts in species composition. Griswold et al. (1978) concluded that riffle species (heptageniids, hydrosychid, elmids) in macroinvertebrate communities are replaced by slow water forms (chironomids and tubificids) after channelization of warmwater streams. Riffles are commonly considered to be the most productive areas in the stream in terms of macroinvertebrate density and diversity. Also, the benthic fauna adapted to riffles are highly desirable fish food species. Pools can support an abundant benthic fauna, but pool-adapted forms are not as heavily utilized by fish. Habitat diversity provided by the run-riffle-pool sequence also contributes greatly to species richness in the fish community.

Velocity and Discharge

Stream velocity is a function of stream gradient and channel roughness. Roughness is a measure of the irregularity in a drainage channel, which will reduce water velocity, and is affected by sinuosity, substrate size, instream vegetation, and other obstructions (Karr and Schlosser 1977).

Discharge or flow (Q) is the volume of water moving past a location per unit time, and is related to velocity as follows:

$$Q = VA$$

where Q = discharge (ft^3/s)
V = velocity (ft/s)
A = cross-sectional area (ft^2).

By increasing the slope and reducing roughness, channelization often increases water velocity (King and Carlander 1976, Simpson et al. 1982); however, if the cross-sectional area of the channel is sufficiently enlarged by widening and deepening, the average velocity may be unchanged or decrease (Bulkley et al. 1976, Griswold et al. 1978). In either case, the velocity is usually made uniform by channelization.

The concept of unit stream power has been developed to predict the rate of sediment transfer in streams. Unit stream power (USP) is defined as the rate of potential energy expenditure per unit weight of water in a channel (Karr and Schlosser 1977) and can be calculated by the following equation (Yang 1972):

$$USP = \frac{dY}{dt} = \frac{dX}{dt}\frac{dY}{dX} = VS$$

where t = time (s)
V = average stream velocity (ft/s)
S = slope or gradient of the channel (ft/100 ft)
Y = elevation above a given point and is equivalent to the potential energy per unit weight of water (i.e., foot-pounds of energy per pound of water)
X = longitudinal distance

USP = unit stream power, (foot-pounds of energy per pound of water per second)

The USP is a measure of the amount of energy available for sediment transport; however, a stream may carry less than the maximum load depending on the availability of sediment due to such factors as bank stability, substrate stability, vegetative cover, and surface erosion.

The effect of channelization on discharge is seasonally variable. During rainy periods a natural stream tends to overflow its banks, inundating adjacent low-lying areas. This flood water is temporarily stored and slowly percolates to the water table. Natural storage dampens runoff surges. Also, the roughness of natural streams slows conveyance, lengthening the time of energy dissipation. A variety of channelization practices designed to increase drainage and hydraulic efficiency (e.g., straightening, removal of channel obstructions, removal of instream and streamside vegetation, berming and leveeing) result in a sharper flow hydrograph and a shorter flow period following rainfall events (Huish and Pardue 1978). The hypothetical hydrographs shown in Figure II-4-1 illustrate the hydrologic/hydraulic effects of channelization. Channelization is designed to rapidly convey water off the land and downstream through the conduit. Properly-functioning channelized streams amplify the impact of high flows. Increased flow velocity, discharge, and unit stream power result in accentuated scour, erosion, bank cutting, sediment transport, and hydraulic loading (flooding); especially below channelized segments. Because of increased hydraulic efficiency, channelized streams return to base flow levels following rainfall more rapidly than natural streams (see Figure II-4-1), and can reduce water availability by lowering the water table. Griswold et al. (1978) concluded that in small, well-drained, agricultural watersheds channel alterations can lead to complete dewatering of long sections of the stream bed during drought conditions. Simpson et al. (1982) summarized the seasonal impacts of channelization as causing lower than natural base flows and higher than normal high flows.

Instream vegetation can be reduced, eliminated, or prevented from reestablishment by high stream velocity.

Current velocity has been cited as one of the most significant factors in determining the composition of stream benthic communities (Cummins 1975). Hynes (1970) suggested that many macroinvertebrates are associated with specific velocities because of their method of feeding and respiration. Macroinvertebrate

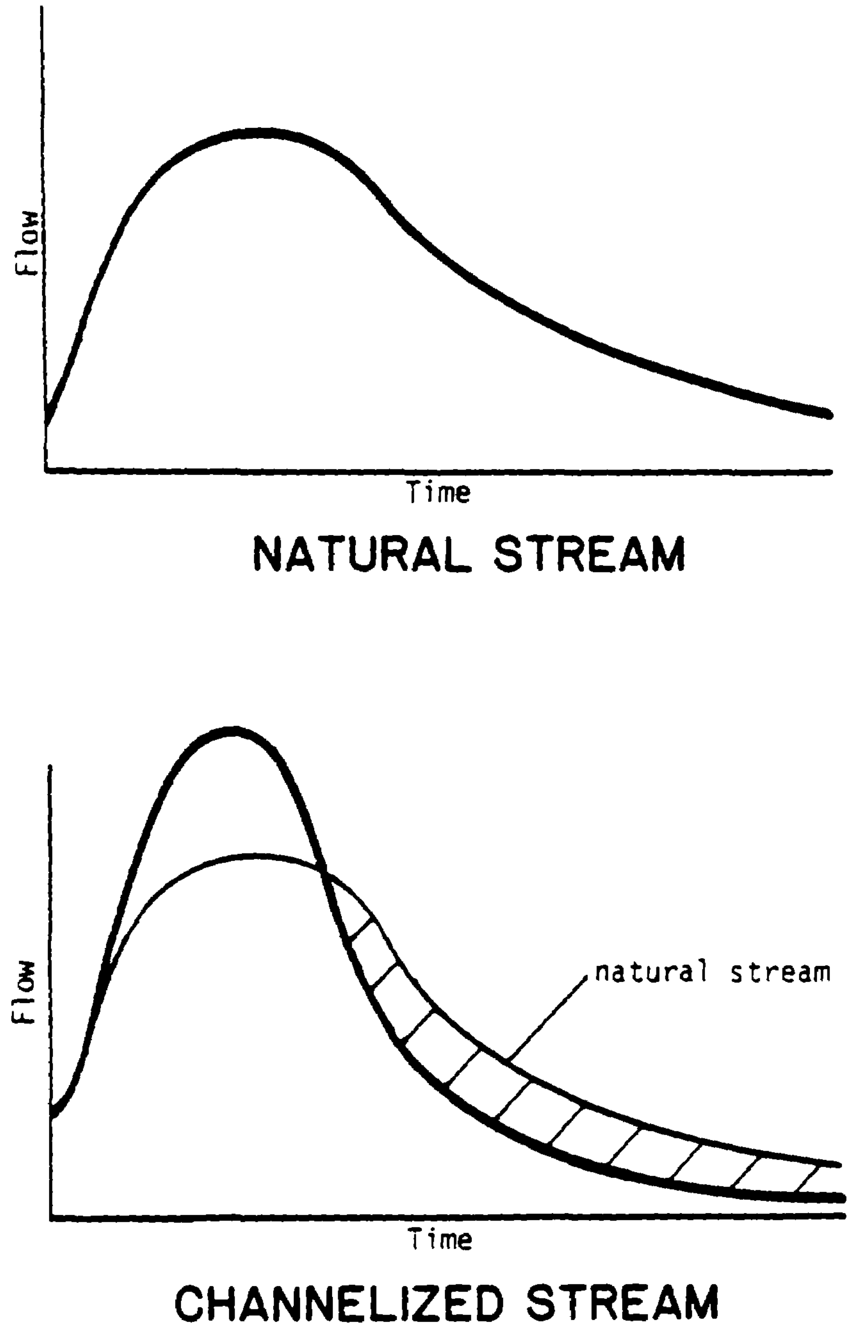


Figure II-4-1. Generalized hydrographs of natural and channelized streams following a rainfall event or season (modified from Simpson et al. 1982).

drift has been found to increase as discharge decreases (Minshall and Winger 1968) and as velocity increases (Walton 1977, Zimmer 1977).

By altering stream velocity, discharge, and unit stream power, channelization modifies the natural substrate. Disruption of the streambed may produce shifting substrates that are unstable habitats for macroinvertebrates. Scour and erosion due to high velocity increases stream turbidity and leads to siltation of downstream reaches. High turbidity can damage macroinvertebrate populations via abrasive action on fragile species (Hynes 1970) and clogging the gills of species without protective coverings (Cairns et al. 1971).

High turbidity and velocity in conjunction with a lack of cover is detrimental to fish. Usually, a very high concentration of sediment is required to directly kill adult fish by clogging the opercular cavity and gill filaments (Wallen 1951), but detrimental behavioral effects occur at much lower levels (Swenson et al. 1976). Turbid waters can also hinder the capture of prey by sight-feeders. An obvious impact of channelization is the loss of habitat due to reduced flow and dessication during drought conditions. Productive riffle areas can be exposed by low flows, thereby directly affecting the benthos and reducing the food supply of fish. Low dissolved oxygen levels during summer low flows can eliminate macroinvertebrates with high oxygen requirements (Hynes 1970), and can affect emergence (Nebeker 1971), drift (Lavandier and Caplancef 1975), and feeding and growth (Cummins 1974).The effects of reduced flow on fish include a degraded food source, and interference with spawning. Concentrating fish into a greatly reduced volume can lead to increased competition, predation, and disease.

Bulkley et al. (1976) found that gradient was a major factor affecting the distribution of fishes. Thus, modifications in gradient by channelization can drastically alter the species composition of a fish community.

Substrate

The stream substrate is ultimately a product of climatic conditions and the underlying geology of the watershed. It is specifically affected by factors such as gradient, weathering, erosion, sedimentation, biological activity, and land use. Channelization generally alters the substrate characteristics of a stream; more often than not, average substrate particle size is reduced (Etnier 1972, King 1973, Griswold et al. 1978).

The substrate of a stream is one of the most important factors controlling the distribution and abundance of aquatic macroinvertebrates (Cummins and Lauff 1969, Minshall and Minshall 1977, Williams and Mundie 1978), and therefore, the impact of channelization on benthic communities is directly related to the degree to which the substrate is affected. Siltation is especially detrimental to the benthos and can cause the following impacts:

1. Decreased habitat diversity due to filling of interstitial spaces (Simpson et al. 1982)

2. Decreased standing crop (Tebo 1955)

3. Decreased density (Gammon 1970)

4. Decreased number of taxa (Simpson et al. 1982)

5. Decreased reproductive success by affecting eggs (Chutter 1969)

6. Decreased productivity (King and Ball 1967)

7. Species shifts from valuable species to burrowing insects and oligochaetes (Morris et al. 1968)

Generally, the impact of channelization via substrate disruption is more significant in high gradient headwater streams (where coarse substrates are essential for protection from a strong current) than in low gradient warmwater streams. Little or no change in benthic communities has been observed in the latter stream type following channelization (Wolf et al. 1972, King and Carlander 1975, Possardt 1976), at least partially because the natural substrate of these ecosystems was not drastically altered by channelization. Shifting substrates are often a consequence of channelizing streams. The absence of a stable habitat leads to reductions in macroinvertebrate populations (Arner et al. 1976). In some streams where channelization has not permanently disturbed the substrate, rapid recoveries (within one year) in the benthic community have been observed (Meehan 1971, Possardt et al. 1976, King and Carlander 1976, Whitaker et al. 1979); however, recovery of macrobenthos can be very slow (Arner et al. 1976).

Changes in macroinvertebrate populations affect the fish community through the food chain. Substrate composition is also important to fish reproduction. For example: trout and salmon require a specific size of gravel in which to build redds and spawn; pikes broadcast eggs over aquatic vegetation which requires silt and mud to grow; sculpins require a slate-type substrate under which they deposit adhesive eggs; and catfish prefer natural cavities for reproduction (Pflieger 1975). Siltation can decrease reproductive success by smothering or suffocating eggs. Channelization can also affect fish adversely by reducing substrate heterogeneity, thereby decreasing habitat diversity.

Cover

Cover is anything that provides real or behavioral protection for an organism. It can allow escape from predators, alleviate the need to expend energy to maintain a position in the current, or provide a place to hide from potential prey or to just be out of sight. Cover includes rocks, logs, brush, instream and overhanging vegetation, snags, roots, undercut banks, crevices, interstices, riffles, backwaters, pools, and shadows. Channelization generally decreases the amount of cover in a stream. Practices such as modification of the streambed (usually into a uniform trapezoidal shape), snagging and clearing, and vegetation removal decrease the total amount and variety of cover, and reduce habitat diversity.

Cover such as logs, stumps, and snags provide valuable stable substrate for macroinvertebrates - especially in streams with a shifting substratum. Instream vegetation serves macroinvertebrates as a substrate for attachment, emergence, and egg deposition. Instream obstructions accumulate leaves, twigs, and other detritus. This coarse particulate organic matter (CPOM) is used as a food source by detritivorous invertebrates (shredders). Retention of CPOM reduces the organic loading on downstream reaches (Marzolf 1978).

Both fish and aquatic macroinvertebrates use cover for predator avoidance, resting, and concealment. Simpson et al. (1982) stated that cover can be regarded as a behavioral habitat requirement for many fish species, and that removal of cover adversely affects fish populations.

Inundation and Desiccation

The modified hydroperiod typical of channelized streams (illustrated in Figure II-4-1) often causes downstream reaches to flood more frequently and more intensely, altering floodplain soils and vegetation, and damaging land values and personal property.

By augmenting land drainage and hydraulic efficiency, channelization has also led to summer drying of streams and desiccation of adjacent and upstream land areas. Nearstream riparian areas provide a number of valuable functions which are often disrupted by channelization. Wetlands assimilate nutrients and trap sediment from runoff and stream overflow, thereby acting as natural purification systems (Karr and Schlosser 1977, Brown et al. 1979). Rapid conveyance and accumulation of nutrients has led to eutrophication problems downstream (Montalbano et al. 1979). Natural fertilization of the floodplain is prevented by restricting flow to the channel. In natural systems, detritus entering the stream from backwaters constitutes an important food source for benthic invertebrates (Wharton and Brinson 1977). Likewise, riparian areas are often rich sources of macroinvertebrates (Wharton and Brinson 1977) that can become available to stream fish during floods or serve as an epicenter for repopulating stream benthos. Some fish (e.g., Esocidae, the pike family) use swampy areas that are seasonally connected to a stream as spawning and nursery habitat. Loss of wetlands due to dewatering precludes these functions.

When wetland areas are drained they become available for other types of land use such as agriculture or development. Conversion of wetlands to pastures and cropland has frequently occurred following channelization. Relative to wetlands, agricultural land uses accentuate runoff, sedimentation, nutrient enrichment (from fertilizers and animal waste), and toxicant leaching (from pesticides).

The response of the benthic community to nutrient enrichment (i.e., from agricultural runoff) generally involves the demise of intolerant, "clean-water" taxa and an increase in numbers and biomass of forms that are tolerant of organic pollution and low dissolved oxygen; a decrease in species diversity often occurs as well.

Land use changes can increase the load of toxic chemicals reaching the stream. Agricultural and urban runoff contribute a variety of toxicants. Saltwater intrusion may become a problem following drainage of coastal wetlands. (Although sodium chloride is generally not considered a toxic chemical it can be lethal to freshwater organisms.) Potential impacts include lethal and chronic effects, biomagnification (via bioaccumulation and bioconcentration), and contamination of human food and recreational resources.

The impact of draining and dewatering riparian areas on terrestrial organisms is extensive. Vegetation (including bottomland hardwoods) tends to undergo a shift from water-tolerant to water-intolerant forms (i.e., hydric > mesic > xeric) (Fredrickson 1979, Maki et al. 1980, Barclay 1980). These vegetative changes along with land use changes and land drainage commonly cause the following impacts on terrestrial fauna:

loss of habitat
loss of cover
loss of food sources
species composition changes
reduced diversity, density, and productivity
increased susceptibility to predation
increased exposure to toxic chemicals.

Streamside Vegetation

Channelization may impact streamside vegetation indirectly through changes in drainage as described above or directly by the clearing of stream banks and the deposition of dredge spoils. Clearing, dredging, and spoil deposition typically result in reduced species diversity and vertical and horizontal structural diversity of streamside vegetation. Tree removal is performed in many channelization projects (Fredrickson 1979, Barclay 1980). Removal of woody species eliminates wildlife habitat, mast production, canopy cover, and shade. Other detrimental impacts of channelization on vegetation include dieback, sunscald, undercutting, and windthrow (Simpson et al. 1982). Spoils deposited on the streambank from channel cutting, dredging, and berming generally make infertile, sandy soils that are easily eroded. Subsequent channel maintenance procedures hinder ecological succession and delay recovery of the stream system.

Interception of rainfall by the vegetative canopy lessens the impact of raindrops on the soil, and bank stability is enhanced by the binding of soil by plant roots. Loss of these functions permits the rate of erosion and the stream sediment load to increase.

Removal of vegetation that shades the stream increases the intensity of sunlight reaching the water column. A resultant increase in the rate of photosynthesis causes changes in the natural pathways of energy flow and nutrient cycling (i.e., trophic structure). Increased primary production can lead to amplification of the diurnal variation in pH and dissolved oxygen concentration following channelization (O'Rear 1975, Huish and Pardue 1978, Parrish et al. 1978). Increasing the incident sunlight raises water temperature. Higher temperatures increase the rates of chemical reactions and biological processes, decrease oxygen solubility, and can exceed the physiological tolerance limits of some macroinvertebrates and fish - most notably trout (Schmal and Sanders 1978, Parrish et al. 1978).

In natural stream systems, allochthonous input of organic matter from streamside vegetation constitutes the major energy source in low-order streams (Cummins 1974). A functional group of benthic organisms called shredders uses allochthonously-derived detritus (CPOM) as a food source, and process it into fine particulate organic matter (FPOM) which is utilized by another functional group, the collectors. Removing streamside vegetation greatly reduces the input of allochthonous detritus and allows primary productivity to increase because of greater light availability. These factors bring about a decline in shredder populations and an increase in herbivorous grazers which take advantage of increasing algae abundance. In headwater areas, species diversity is likely to decrease due to the loss of detritivorous taxa, and macroinvertebrate density may decline because the swift current of those reaches is not conducive to planktonic and some periphytic algae forms. Loss of allochthonous material has less impact on intermediate-order streams because they are naturally autotrophic ($P/R>1$), except that channelization of upstream reaches reduces the amount of FPOM that is received via nutrient spiraling. The literature contains excellent discussions of energy and materials transport in streams (Cummins et al. 1973, Cummins 1974, Cummins 1975, Marzolf 1978, Vannote et al. 1980).

Reductions and changes in the macroinvertebrate community affect the food source of fishes. Changing availabilities of detritus and algae may skew the fish community with respect to trophic levels that utilize those energy sources. Clearing away nearstream vegetation also reduces the input of terrestrial insects that are eaten by fish.

In addition, streamside vegetation provides cover in the form of shadows, root masses, limbs, and trees which fall into the stream. Most game fish species prefer shaded habitats near the streambank.

SUMMARY

The benefits realized by channelizing a stream are often obtained at the expense of such impacts as:

1. Increased downstream flooding
2. Reduction of groundwater levels and stream dewatering
3. Increased bank erosion, turbidity, and sedimentation
4. Degradation of water quality
5. Promotion of wetland drainage and woodland destruction
6. Promotion of land development (agricultural, urban, residential, industrial)
7. Loss of habitat and reduced habitat diversity
8. Adverse effects on aquatic and terrestrial communities (productivity, diversity, species composition)
9. Lowered recreational values

The time required for a natural stream to return to a productive, visually-appealing body of water is highly variable. Natural recovery of some channelized streams requires better than 30 years. Restoration of the stream channel and biota can be accelerated by mitigation practices.

The potential negative impacts and time frame of recovery should weigh heavily in the evaluation of any newly-proposed channelization project.

CHAPTER II-5

TEMPERATURE

Temperature exerts an important influence on the chemical and biological processes in a water body. It determines the distribution of aquatic species; controls spawning and hatching; regulates activity; and stimulates or suppresses growth and development. The two most important causes of temperature change in a water body are process and cooling water discharges, and solar radiation. The consequences of temperature variation caused by thermal discharges (thermal pollution) continue to receive considerable attention. An excellent review on this subject may be found in the Thermal Effects section of the annual literature review issue of the Journal of the Water Pollution Control Federation. Discussion in this chapter is limited to the influence of seasonal temperature variation on a water body.

PHYSICAL EFFECTS

Annual climatological cycles and precipitation patterns are controlled by the annual cycle of solar radiation. Specific patterns of temperature and precipitation, which vary geographically, determine annual patterns of flow to lakes and streams. In general, winter precipitation in northern latitudes does not reach a body of water until the spring snow melt. For this reason, streamflow may be quite low in the winter but increase rapidly in the spring. Low flow typically occurs in the summer throughout North America.

Changes in season cause changes in water temperature in lakes and streams. The patterns of temperature change in lakes are well understood. Briefly, many lakes tend to stratify in the summer, with a warm upper layer (the epilimnion), a cold bottom layer (the hypolimnion) and a sharp temperature difference between the two, known as the thermocline. The depth of the thermocline is determined to large extent by the depth to which solar radiation penetrates the water body. The epilimnion tends to be well oxygenated, through both algal photosynthesis, and through oxygen transfer from the atmosphere. Surface wind shear forces help mix the epilimnion and keep it oxygenated. The thermocline presents a physical barrier, in a sense, to mixing between the epilimnion and the hypolimnion. If no photosynthesis takes place in the hypolimnion, due to diminished solar radiation, and if there is no exchange with the epilimnion, dissolved oxygen levels (DO) in the bottom layer may drop to critical levels, or below. Often water released through the bottom of a dam has no dissolved oxygen, and may severely jeopardize aquatic life downstream of the impoundment.

Typical summer and annual lake temperature profiles are presented in Figures II-5-1 and II-5-2, respectively. In the fall the thermocline disappears and the lake undergoes turnover and becomes well mixed. The temperature becomes fairly homogeneous in the winter (Figure II-5-2), there is another wind induced turnover in the spring and the cycle ends with the development of epilimnion, hypolimnion and thermocline in the summer.

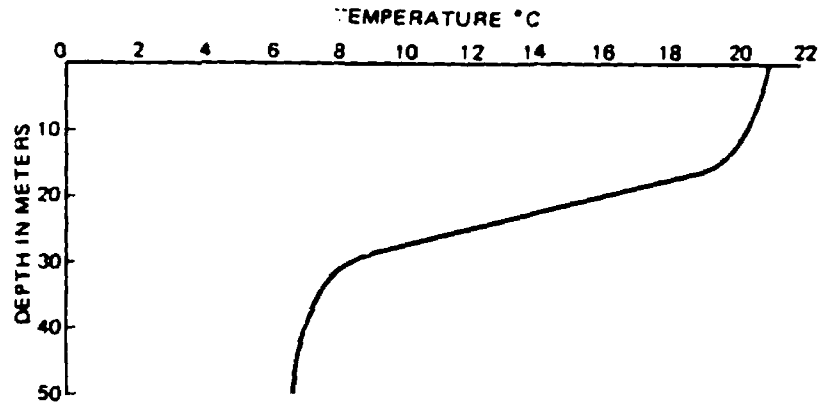


Figure II-5-1. Summer Temperature Conditions in a Typical (Hypothetical) Temperate-Region Lake.

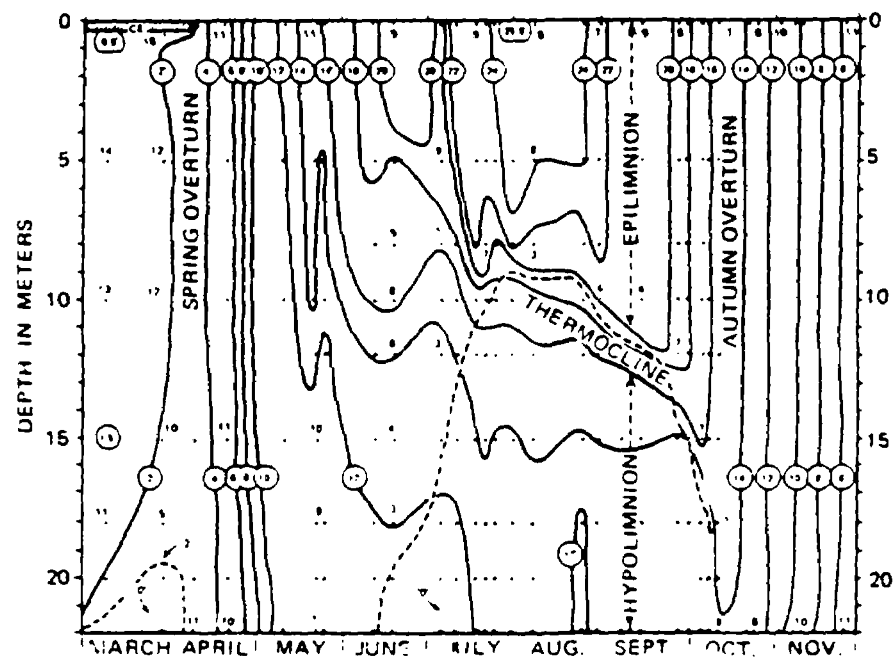


Figure II-5-2. The Seasonal Cycle of Temperature and Oxygen Conditions in Lake Mendota, Wisconsin, 1906, (Reid and Wood).

Rivers and streams generally show a much more homogeneous temperature profile, largely because turbulent stream flow assures good vertical mixing. Nevertheless, small streams may undergo temperature variation as flow passes through shaded or sunny areas, as it is augmented by cool groundwater or warm agricultural or other surface return flow, or as it becomes more turbid and captures solar radiation in the form of heat.

TEMPERATURE RELATED BIOLOGICAL EFFECTS

Warm blooded homeothermic animals, such as the mammals, have evolved a number of methods by which to control body temperature. Cold blooded poikilothermic animals, such as fish, have not evolved these mechanisms and are much more susceptible to variation in temperature than are warm blooded animals. Perhaps the most important adaptation of fish to temperature variation is seen in the timing of reproductive behavior.

Gradual seasonal changes in water temperature often trigger spawning, metamorphosis and migration. The eggs of some freshwater organisms must be chilled before they will hatch properly. The tolerable temperature range for fish is often more restrictive during the reproductive period than at other times during maturity. The temperature range tolerated by many species may be narrow during very early development but increases somewhat during maturity. Reproduction may be hindered significantly by increased temperature because this function takes place under restricted temperature ranges. Spawning may not occur at all when temperatures are too high. Thus, a fish population may exist in a heated area only because of continued immigration.

Because fish are cold-blooded, temperature is important in determining their standard metabolic rate. As temperature increases, all standard metabolic functions increase, including feeding rates. Water temperature need not reach lethal levels to eliminate a species. Temperatures that favor competitors, predators, parasites and disease can destroy a species at levels far below those that are lethal.

Since body temperature regulation is not possible in fish, any changes in ambient temperature are immediately communicated to blood circulating in the gills and thereby to the rest of the fish. The increase in temperature causes an increase in metabolic rates and the feeding activity of the fish must increase to satisfy the requirements of these elevated levels. Elevated biochemical rates facilitate the transport of toxic pollutants to the circulatory system via the gill structure, and hasten the effect these toxicants might exert on the fish. Increased temperature will also raise the rate at which detoxification takes place through metabolic assimilation, or excretion. Despite these mechanisms of detoxification, a rise in temperature increases the lethal effect of compounds toxic to fish. A literature review on this subject will also be found in the JWPCF annual literature review number.

The importance of temperature to fish may also be seen in Tables II-5-1 and II-5-2. The data in these tables were found in references by Carlander (1969, 1972) and Brungs and Jones (1977). Table II-5-1 shows the preferred temperature for a number of fish and Table II-5-2 shows the range of temperatures within which spawning may occur in several species of fish.

Preferred temperatures usually are determined through controlled laboratory experiments although some values published in the literature are based on field observations. Determination of final temperature preferenda of fish in the field is difficult because field environments cannot be controlled to match laboratory studies (Cherry and Cairns, 1982). Temperature preference studies are based on an acclimation temperature which is used as a reference point against which to examine the response of fish to different temperature levels. The acclimation temperature itself is critical for it affects the range of temperatures within which fish prefer to live. This may be seen in Figure II-5-3 which shows an increase in preferred temperature and in the upper threshold of avoidance with an increase in acclimation temperature. The range between the acclimation and the upper avoidance temperatures is species specific and is dependent on the acclimation temperature in which the fish were tested. A greater variability in fish avoidance response is observed in winter than in summer testing conditions (Cherry and Cairns, 1982).

Temperature preference/avoidance studies are important to an understanding of the effect of thermal pollution on the biota of a water body. The literature on temperature preference will be important to the water body survey in two ways: when the stream reach of interest is affected by thermal pollution or when ambient temperature patterns may be a contributing factor which determines the types of fish that might be expected to inhabit a water body under different management schemes identified during the assessment.

Temperature is also important because it strongly influences self-purification in streams. When a rise in temperature occurs in a stream polluted by organic matter, an increased rate of utilization of dissolved oxygen by biochemical processes is accompanied by a reduced availability of DO due to the reduced solubility of gases at higher temperatures. Because of this, many rivers which have adequate DO in the winter may be devoid of DO in the summer.

Bacteria and other microorganisms which mediate the breakdown of organic matter in streams are strongly influenced by temperature changes and are more active at higher than at lower temperatures. The rate of oxidation of organic matter is therefore much greater during the summer than during the winter. This means self purification will be more rapid, and the stream will recover from the effects of organic pollution in a shorter distance during the warmer months of the year than in the colder months of the year, provided there is an adequate supply of dissolved oxygen.

Temperature is an important regulator of natural conditions. It has a profound effect on habitat properties in lakes and streams; on the solubility of gases such as oxygen, upon which most aquatic life is dependent; on the toxicity of pollutants; on the rate and extent of chemical and biochemical reactions; and on the life cycle of poikilothermic aquatic life in general. Since in the context of the water body survey uses are framed in reference to the presence and

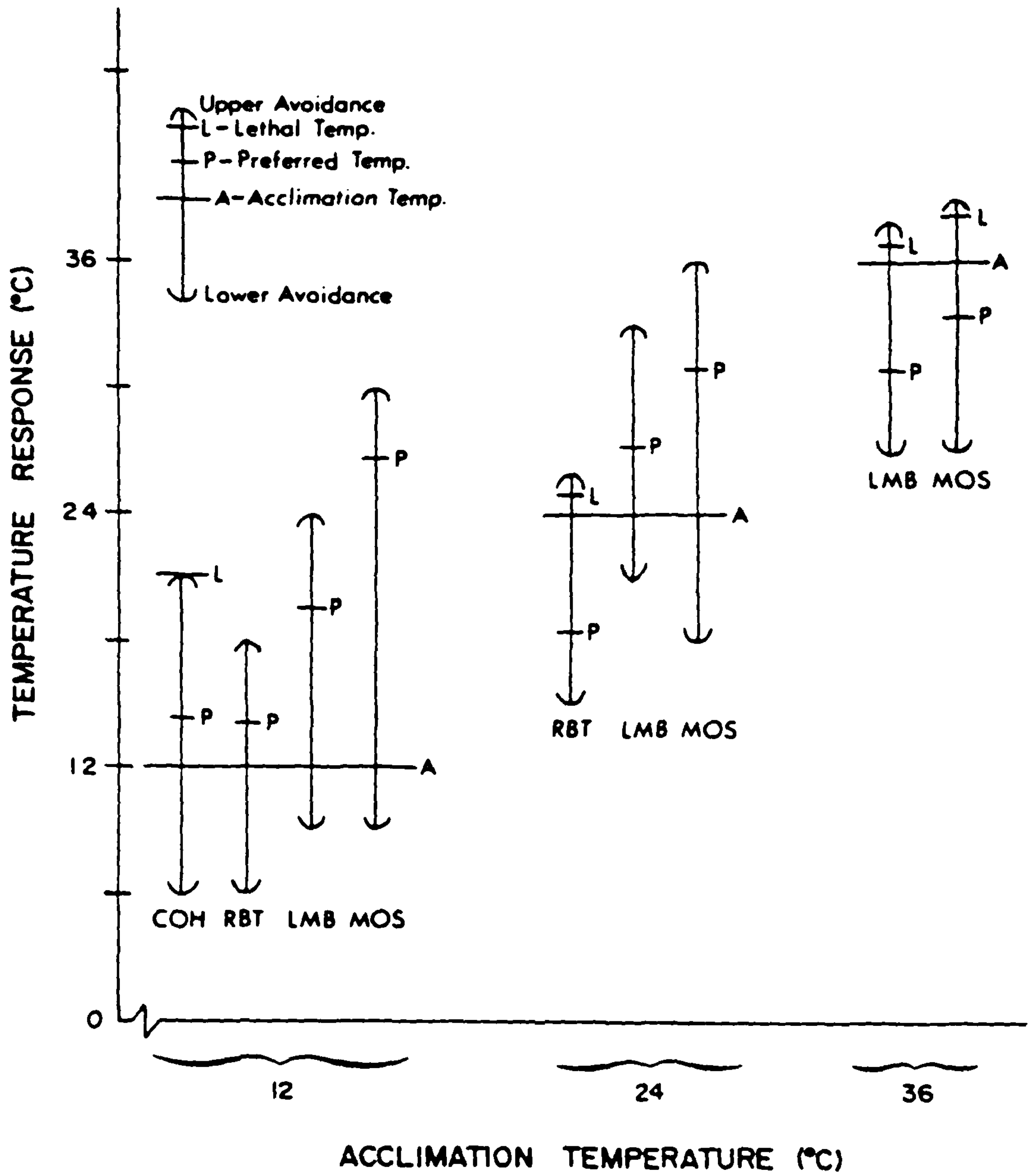


Figure II-5-3. Relationships of preffered (P), avoidance (⌶), and lethal temperatures to the acclimation (A) temperature for coho salmon (COH), rainbow trout (RBT), largemouth bass (LMB), and mosquitofish (MOS) from laboratory trials (from Cherry and Cairns, 1982).

the protection of aquatic life, those factors which support or jeopardize aquatic life must be considered.

Perhaps the most critical element in the aquatic environment is dissolved oxygen, whose solubility is a function of temperature. Oxygen is added to an aquatic system by photosynthesis and by transfer from the atmosphere. Unfortunately, the availability of dissolved oxygen is apt to be greatest when the requirement for DO is least, i.e., in the winter when metabolic activity has been substantially reduced. Conversely, the availability may be lowest when the demand is greatest.

Consideration of the relationship of temperature and availability of dissolved oxygen is important to the water body survey, and will require a close examination of natural seasonal variation in DO and its interaction with treatment process efficiency, with the oxygen demand of the CBOD and NBOD in wastewaters, and with the seasonal requirements of aquatic life.

TABLE II-5-1. PREFERRED TEMPERATURE OF SOME FISH SPECIES.

Species: Common name	Species: Latin name	Life Stage	Acclimation Temperature, °C	Preferred Temperature, °C
Alewife	Alosa pseudoharengus	J	18	20
		J	21	22
		A	24	23
		A	31	23
Threadfin shad	Dorosoma petenense	A		>19
Sockeye salmon	Oncorhynchus nerka	J		12-14
		A		10-15
Pink salmon	O. gorbuscha	J		12-14
Chum salmon	O. keta	J		12-14
Chinook salmon	O. tshawytscha	J		12-14
Coho salmon	O. Kisutch	J		12-14
		A		13
Cisco	Coregonus artedii	A		13
Lake whitefish	C. clupeaformis	A		13
Cutthroat trout	Salmo clarki	A		9-12
Rainbow trout	S. gairdneri	J	not given	14
		J	18	18
		J	24	22
		A		13
Atlantic salmon	S. salar	A		14-16
Brown trout	S. trutta	A		12-18
Brook trout	Salvelinus fontinalis	J	6	12
		J	24	19
		A		14-18
Lake trout	Salvelinus namaycush	J		8-15
Rainbow smelt	Osmerus mordax	A		6-14
Grass pickerel	Esox americanus vermiculatus	J,A		24-26

TABLE II-5-1. PREFERRED TEMPERATURE OF SOME FISH SPECIES. (Continued)

Species: Common name	Species: Latin name	Life Stage	Acclimation Temperature, °C	Preferred Temperature, °C
Muskellunge	Esox masquinongy	J		26
Common carp	Cyprinus carpio	J	10	17
		J	15	25
		J	20	27
		J	25	31
		J	35	32
		A	Summer	33-35
Emerald shiner	Notropis atherinoides	J	Summer	25
White sucker	Catostomus commersoni	A		19-21
Buffalo	Ictiobus sp.	A		31-34
Brown bullhead	Ictalurus nebulosus	J	18	21
		J	23	27
		J	26	31
		A		29-31
Channel catfish	Ictalurus punctatus	J	22-29	35
		A		30-32
White perch	Morone americana	J	6	10
		J	15	20
		J	20	25
		J	26-30	31-32
White bass	M. chrysops	A	Summer	28-30
Striped bass	M. saxatilis	J	5	12
		J	14	22
		J	21	26
		J	28	28
Rock bass	Ambloplites rupestris	A		26-30
Green sunfish	Lepomis cyanellus	J	6	16
		J	12	21
		J	18	25
		J	24	30
		J	30	31

TABLE II-5-1. PREFERRED TEMPERATURE OF SOME FISH SPECIES. (Continued)

Species: Common name	Species: Latin name	Life Stage	Acclimation Temperature, °C	Preferred Temperature, °C
Pumpkinseed	L. gibbosus	J	8	10
		J	19	21
		J	24	31
		J	26	33
		A		31-31
Bluegill	L. machrochirus	J	6	19
		J	12	24
		J	18	29
		J	24	31
		J	30	32
Smallmouth bass	Micropterus dolomieui	J	15	20
		J	18	23
		J	24	30
		J	30	31
Spotted bass	M. punctulatus	J	6	17
		J	12	20
		J	18	27
		J	24	30
		J	30	32
Largemouth bass	M. salmoides	J		26-32
White crappie	Pomoxis annularis	J	5	10
		J	24	26
		J	27	28
		A		28-29
Black crappie	P. nigromaculatus	J		27-29
		A		24-31
Yellow perch	Perca flavescens	J,A		19-24
Sauger	Stizostedion canadense	A		18-28
Walleye	S. vitreum	J,A		20-25
Freshwater drum	Aplodinotus grunniens	A		29-31

TABLE II-5-2. SPAWNING TEMPERATURE OF SOME FISH SPECIES.

Species: Common name	Species: Latin name	Spawning temperature, °C: approximate value or range	Spawning temperature, °C: optimum or peak	Spawning season month
Lamprey				
Northern brook	Ichthyomyzon fosser	13-77		May-Jun
Southern brook	Ichthyomyzon gagei	15		Mar-May
Allegheny brook	Ichthyomyzon greeleyi	19		May
Mountain brook	Ichthyomyzon hubbsi	10-12		Mar-Apr
Silver	Ichthyomyzon unicuspis	>10		Apr-Jun
Least brook	Ichthyomyzon aepyptera	10-16		Mar-May
Arctic	Lampetra japonica	12-15		May-Jul
American brook	Lampetra lamottei		17	Apr-Jun
Western brook	Lampetra richardsoni	8-20	9-11	Mar-Jun
Pacific	Lampetra tridentata	>8		Apr
Sea	Petromyzon marinus	11-24		Apr-Jul
Sturgeon				
Shortnose	Acipenser brevirostrum	8-12		Apr-Jun
Lake	Acipenser fulvenscens	12-19		Apr-Jun
Atlantic	Acipenser oxyrhynchus	13-18		Feb-Jul
White	Acipenser transmontanus	9-17		May-Jul
Paddlefish	Polydon spathula	16		May-Jun
Gar				
Longnose	Lepisosteus osseus	>11		Mar-Aug
Shortnose	Lepisosteus platostomus	19-24		May-Jul
Bowfin	Amia calva	16-19		Apr-Jul
Blueback herring	Alosa aestivalis	14-27		Apr-Jul
Shad				
Alabama	Alosa alabamae	19-22		Jan-Jul
Hickory	Alosa mediocris	18-21		May-Jun
Alewife	Alosa pseudoharengus	13-28		Apr-Aug
American	Alosa sapidissima	11-19		Jan-Jul
Gizzard	Dorosoma cepedium	17-29		Mar-Aug
Threadfin	Dorosoma petenense	14-23	21	Apr-Aug

TABLE II-5-2. SPAWNING TEMPERATURE OF SOME FISH SPECIES. (Continued)

Species: Common name	Species: Latin name	Spawning temperature, °C: approximate value or range	Spawning temperature, °C: optimum or peak	Spawning season month
Salmon				
Pink	Oncorhynchus gorbuscha		10	Jul-Oct
Sockeye	Oncorhynchus nerka (anadromous)	3-7		Jul-Dec
(Kokanee)	Oncorhynchus nerka (landlocked)	5-10		Aug-Feb
Coho	Oncorhynchus kisutch	7-13		Oct-Jan
Whitefish				
Cisco	Coregonus artedii	1-5	3	Nov-Dec
Lake	Coregonus clupeaformis	1-10		Sep-Dec
Bloater	Coregonus hoyi	5		Nov-Mar
Alaska	Coregonus nelsoni	0-3		Sep-Oct
Least cisco	Coregonus sardinella	0-3		Sep-Oct
Kiyi	Coregonus kiyi	2-5		Oct-Jan
Shortnose cisco	Coregonus reighardi	3-5		Apr-Jun
Pygmy	Prospium coulteri	0-4		Oct-Jan
Round	Prospium cylindraceum	0-4		Oct-Dec
Mountain	Prospium spilonotus	5-12		Sep-Dec
Trout				
Golden	Salmo aguabonita	7-10		Jun-Jul
Arizona	Salmo apache	8		May
Cutthroat	Salmo clarki	10		Jan-May
Rainbow	Salmo gairdneri	5-17	9-13	Apr-Jul/Nov-Feb
Gila	Salmo gilae	8		Apr-May
Atlantic salmon	Salmo salar	2-10	4-6	Oct-Dec
Brown	Salmo trutta	1-13	7-9	Oct-Feb
Arctic char	Salvelinus alpinus	1-13	3-4	Sep-Dec
Brook trout	Salvelinus fontinalis	3-12	9	Aug-Dec

TABLE II-5-2. SPAWNING TEMPERATURE OF SOME FISH SPECIES. (Continued)

Species: Common name	Species: Latin name	Spawning temperature, °C: approximate value or range	Spawning temperature, °C: optimum or peak	Spawning season month
Dolly Varden	Salvelinus malma	5-8		Sep-Nov
Lake	Salvelinus namaycush	3-14		Aug-Dec
Inconnu	Stenodus leucichthys	1-5		Sep-Oct
Arctic grayling	Thymallus arcticus	4-11		Mar-Jun
Rainbow smelt	Osmerus mordax	1-15		Feb-May
Eulachon	Thaleichthys pacificus	4-8		Mar-May
Goldeye	Hiodon alosoides	10-13		May-Jul
Alaska blackfish	Dallia pectoralis	10-15		May-Aug
Central mudminnow	Umbra limi	13		Apr
Pickerel				
Redfin	Esox americanus americanus	10		Feb-Apr
Grass	Esox americanus vermiculatus	7-12	10	Mar-May/Aug-Oct
Chain	Esox niger	6-16	8	Mar-May
Northern pike	Esox lucius	3-19		Feb-Jul
Muskellunge	Esox masquinongy	9-15	13	Apr-May
Chiselmouth	Acrocheilus alutaceus	17		Jun-Jul
Central stoneroller	Campostoma anomalum	13-27		Apr-Jun
Goldfish	Carassius auratus	16-30		Feb-Nov
Redside dace	Clinostomus elongatus	>18		May
Lake chub	Couesius plumbeus	14-19		May-Jun
Common carp	Cyprinus carpio	14-26	19-23	Mar-Aug

TABLE II-5-2. SPAWNING TEMPERATURE OF SOME FISH SPECIES. (Continued)

Species: *Common name*	Species: *Latin name*	Spawning temperature, °C: approximate value or *range*	Spawning temperature, °C: optimum *or peak*	Spawning season *month*
Utah chub	Gila atraria	12-16		Apr-Aug
Tui chub	Gila bicolor	16		Apr-Jun
Brassy minnow	Hybognathus hankinsoni	10-13		May-Jun
Silvery minnow	Hybognathus nuchalis	13-21		Apr-May
Chub				
River	Hybobsis micropogon	19-28		May-Aug
Silver	Hybobsis storeriana	18-21		May-Jun
Clear	Hybobsis winchelli	10-17		Feb-Mar
Rosyface	Hybobsis rubriformes	19-23		Apr-Jun
Peamouth	Mylocheilus caurinus	11-22		May-Jun
Hornyhead chub	Nocomis biguttatus	24		Spring
Shiner				
Golden	Notemigonus crysoleucas	16-21		May-Aug
Satinfin	Notropis analostanus	18-27		May-Aug
Emerald	Notropis atherinoides	20-28	24	May-Aug
Bridle	Notropis bifrenatus	14-27		May-Jul
Warpaint	Notropis coccogenis	20-24		Jun-Jul
Common	Notropis cornutus	15-28	19-21	Apr-Jul
Fluvial	Notropis edwardraneyi	28		Jun
Whitetail	Notropis galacturus	24-28		May-Jun
Spottail	Notropis hudsonius	20		May-Jul
Rosyface	Notropis rubellus	20-29		May-Jul
Saffron	Notropis rubricroceus	19-30		May-Jul
Sacremento blackfish	Orthodon microlepidotus	15		Apr-Jun
Bluntnose minnow	Pimephales notatus	21-26		Apr-Sep
Fathead minnow	Pimephales promelas	14-30	23-24	May-Aug
Sacremento squawfish	Ptychocheilus grandis	4		Apr-Jun
Northern squawfish	Ptychocheilus oregonensis	12-22	18	May-Jun

TABLE II-5-2. SPAWNING TEMPERATURE OF SOME FISH SPECIES. (Continued)

Species: Common name	Species: Latin name	Spawning temperature, °C: approximate value or range	Spawning temperature, °C: optimum or peak	Spawning season month
Blacknose dace	Rhinichthys atratulus	16-22	21	May-Jun
Longnose dace	Rhinichthys cataractae	12-16		May-Aug
Redside shiner	Richardsonius balteatus	10-18		Apr-Jul
Creek chub	Semotilus atromaculatus	>12		Apr-Jul
Fallfish	Semotilus corporalis	>16		May-Jun
Pearl dace	Semotilus margarita	17-18		May-Jun
Sucker				
Longnose	Catostomus catostomus	>5		May-Jun
White	Catostomus commersoni	8-21		Mar-Jun
Flannelmouth	Catostomus latipinnis	13		Apr-Jun
Largescale	Catostomus macrocheilus	>7		Apr-Jun
Mountain	Catostomus platyrhynchus	10-19		Jun-Jul
Tahoe	Catostomus tahoensis	11-14		Apr-Jun
Blue	Catostomus elongatus	10-15		Apr-Jun
Northern hog	Hypentelium nigricans	>15		May
Smallmouth buffalo	Ictiobus bubalus	14-28	17-24	Mar-Sep
Bigmouth buffalo	Ictiobus cyprinellus	14-27	16-18	Apr-Jun
Spotted sucker	Minytrema melanops	13-18		Apr-May
Blackfin sucker	Moxostoma atripinne	12-18		Apr
Redhorse				
Silver redhorse	Moxostoma anisurum	>13		Apr-May
River	Moxostoma breviceps	22-25		Apr
Black	Moxostoma duquesnei	13-23		Apr-May
Golden	Moxostoma erythrurum	15-22		Apr-May
Shorthead	Moxostoma macrolepidotum	11-22		Apr-May
Greater	Moxostoma valenciennesi	16-19		May-Jul
Humpback sucker	Xyrauchen texanus	12-22		Mar-Apr

TABLE II-5-2. SPAWNING TEMPERATURE OF SOME FISH SPECIES. (Continued)

Species: Common name	Species: Latin name	Spawning temperature, °C: approximate value or range	Spawning temperature, °C: optimum or peak	Spawning season month
Catfish				
White	Ictalurus catus	20-29		Jun-Jul
Blue	Ictalurus furcatus	>22		Apr-Jun
Black bullhead	Ictalurus melas	>21		May-Jul
Brown bullhead	Ictalurus nebulosus	>21		Mar-Sep
Channel	Ictalurus punctatus	21-29	27	Mar-Jul
Flathead	Pylodictis olivaris	22-28		May-Jul
Stonecat	Noturus flavus	27		Jun-Aug
Bridled madtom	Noturus miurus	25-26		Jul-Aug
White River springfish	Crenichthys baileyi	32		
Desert pupfish	Cyprinodon macularius	>20	28-32	Apr-Oct
Banded kilifish	Fundulus diaphanus	>21	23	Apr-Sep
Plains kilifish	Fundulus kansae	28		Jun-Aug
Mosquitofish	Gambusia affinis	23		Mar-Oct
Burbot	Lota lota	0-2		Jan-Feb
Brook stickleback	Eucalia inconstans	4-21		Apr-Jul
Threespine stickleback	Gasterosteus aculeatus	5-20		Apr-Sep
Trout-perch	Percopsis omiscomaycus	6-21		May-Aug
White perch	Morone americana	11-20		May-Jul
White bass	Morone chrysops	12-21		Apr-Jun
Striped bass	Morone saxatilis	12-22	16-19	Apr-Jun
Rock bass	Ambloplites rupestris	16-26		Apr-Jun
Sacremento perch	Archoplites interruptus	22-28		May-Aug
Flier	Centrarchus macropterus	17		Mar-May

TABLE II-5-2. SPAWNING TEMPERATURE OF SOME FISH SPECIES. (Continued)

Species		Spawning temperature, °C		Spawning
Common name	Latin name	approximate value or range	optimum or peak	season month
Banded pygmy sunfish	Elassoma zonatum	14-23		Mar-May
Sunfish				
Redbreast	Lepomis auritus	17-29		Apr-Aug
Green	Lepomis cyanellus	20-28		May-Aug
Pumpkinseed	Lepomis gibbosus	19-29		May-Aug
Warmouth	Lepomis gulosus	21-26		May-Aug
Orangespotted	Lepomis humilis	>18		May-Aug
Bluegill	Lepomis machrochirus	19-32	25	Feb-Aug
Longear	Lepomis megalotis	22-30		May-Aug
Redear	Lepomis microlophus	20-32		Mar-Sep
Spotted	Lepomis punctatus	18-33		Mar-Nov
Bass				
Redeye	Micropterus coosae	17-23		Apr-Jul
Smallmouth	Micropterus dolomieui	13-23	17-18	Apr-Jul
Suwannee	Micropterus notius	>19		Feb-Jun
Spotted	Micropterus punctulatus	15-21		May-Jun
Largemouth	Micropterus salmoides	12-27	21	Apr-Jun/Nov-May
White crappie	Pomoxis annularis	14-23	16-20	Mar-Jul
Black crappie	Pomoxis nigromaculatus	14-20		Mar-Jul
Yellow perch	Perca flavescens	4-15	12	Mar-Jul
Sauger	Stizostedion canadense	4-15	9-15	Mar-Jul
Walleye	Stizostedion vitreum	4-17	6-9	Mar-Jun
Greenside darter	Etheostoma blennioides	>10		Apr-Jun
Johnny darter	Etheostoma nigrum	>18		
Channel darter	Percina copelandi	20-21		Jul
Blackside darter	Percina maculata	16-17		May-Jun
Mottled sculpin	Cottus bairdi	10		Apr-May
Freshwater drum	Aplodinotus grunniens	18-24	23	May-Aug

CHAPTER II-6
RIPARIAN EVALUATIONS

Riparian ecosystems can be variously identified but their common element is that they are adjacent to aquatic systems. Brinson et al., (1981) defines them as "riverine floodplain and streambank ecosystems. Cowardin et al., (1979) in their"Classification of Wetlands Habitats of the U.S.", do not clearly delineate riparian and wetland zones. For this chapter emphasis will be given to floodplain, riverine and lacustrine riparian habitats and no distinction has been made between riparian and wetland land environments.

The primary legislative justification for riparian protection is the Clean Water Act, specifically that section dealing with water quality. Many factors enter into the relationship between riparian ecosystems and water quality; a simple correlation between any single measure of riparian habitat and water quality does not exist. A well developed riparian zone is frequently the juncture between terrestrial and aqautic environments and its characteristics are governed to some extent by both. The riparian zone is usually related to the adjacent terrestrial environment with respect to climatic conditions, soil types, land topography etc. The aquatic system is an integration of upstream drainage (Lotspeich 1980) and has the riparian zone as an important component. The aquatic effects to the riparian ecosystem will vary with factors such as stream size, climatic vegetation and soil type. Although no ideal riparian habitat water quality scenario is possible, general relationships can be derived.

A critical relationship exists between stream size and the extent of riparian habitat. Small streams canopied by riparian vegetation will be more influenced than large streams where riparian canopy represents only a small fraction of the immediate channel. The small riparian zone in relation to stream size of many large streams has frequently been cited in order to diminish the importance of this habitat. The presumption is made that riparian importance is minimal because the riparian/river size ratio is small. It is also argued that alteration of smaller streams is insignificant with respect to the total drainage basin and that such activities have minimal implications for larger streams. An obvious impact of large stream riparian modification is shore line destruction and subsequent loss of near shore stream habitat. Although modification of a single small tributary may have a minimal effect on the larger water body, major drainage basin alterations could seriously damage water resources, the larger stream being a product of its tributaries.

Riparian system have unique ecosystem qualities which should be considered in addition to their water qualiy values. Riparian zones are cited as classical ecotones which will usually support greater species and numerical diversity than adjacent aquatic or terrestrial environments. Large numbers of rare and endangered animal and plant species reside here. It is often critical habitat for an entire life span or it may be used in a transitory manner for reproduction, migration or as hunting territory for raptors and carnivorous mammals. Even though organisms may not use the riparian zone as their primary living habitat, its loss may seriously disrupt foodchain mechanisms and life history processes. Significant changes in species numbers, diversity and types may occur in both the terrestrial and aquatic environments following riparian destruction. It is estimated that less than two percent of the land area in the U.S. is riparian habitat (Brinson et al., 1981). Large portions have been converted to agricultural use, e.g. the Mississippi bottomland hardwoods, and stream channelization has destroyed adjacent riparian ecosystems. Timber removal has greatly reduced riparian habitat in forested regions. Livestock grazing has had extremely detrimental riparian effects on semi-arid rangelands. Land values have favored agricultural and urban development immediately adjacent to the aquatic environment with the exclusion of most natural vegetation.

PHYSICAL RELATIONSHIPS

Key physical stream characteristics are affected by the riparian ecosystem. Water temperature responds to almost any riparian alteration in smaller streams. Several studies (Karr and Schlosser 1978, Moring 1975, Campbell 1970) have demonstrated that shade afforded by adjacent vegetation significantly moderates water temperature, reducing summer highs and decreasing winter lows. This can have significant effects on many chemical and biological processes. Chemical reaction rates are temperature dependent and increased temperature generally increases reaction rates. Adsorption, absorption, precipitation reactions, decomposition rates, and nutrient recycling dynamics could all be altered. Many aquatic organisms have relatively specific temperature requirements. Elevated temperatures increase poikilotherm metabolic rates causing excessively low production during food deprivation and the increased temperature may disrupt critical life stages such as reproduction. Temperatures exceeding or substantially below optimal requirements, even for relatively brief periods, can completely alter the biota. Larger streams may not be physically affected as readily as the smaller tributaries but large scale tributary modifications could have dramatic downstream consequences.

Another direct physical consequence is alteration in the quality and quantity of incident solar radiation. Optimal photosynthetic wave lengths, especially for diatoms, may be altered by the canopy, but as will be elaborated later, this may not have serious consequences to a diversified biota. Turbidity will be reduced by riparian vegetation. This too will be discussed in greater detail. A further loss with reduction in riparian habitat is the fine particulate matter, especially the nutrient rich organic material. This may be transferred to the adjacent terrestrial environment during floods or carried directly to the large streams with such a reduced residence time in the smaller stream that they become nutrient limited.

FLUVIAL RELATIONSHIPS

Fluvial characteristics are governed by such processes as stream bank stability, flow rates, rainfall seasonality and water volumes. Stream bank stability is important in maintaining stream integrity. This stability is a function of the local geology and riparian vegetation.

Streams are not static but new channel formation rates are slowed with increased bank stability. During high water, bank erosion is minimized and excess flow energy dissipated over floodplains with minimal environmental damage. Without riparian vegetation, flooding is more erosive and extensive. Energies are not dissipated readily but remain excessive for the duration of the high water. The geomorphological consequences can be considerable; extreme erosion, formation of additional channels, upland sediment deposition etc. The biological impact can be devastating, with the aquatic habitat physically destroyed or silted to the extent it is no longer a biologically viable unit. Under extreme conditions, silt levels may be sufficient to cause embryo death and physiological damage to gill breathing organisms. This scenario is best illustrated using the example of stream channelization. High energy water movement leads to rapid land drainage but also to extremely damaging floods when stream banks overflow. Biological communities may become species depauperate, biomass greatly reduced and those populations remaining may be undesirable compared to previous inhabitants.

Riparian zone groundwater levels are controlled by adjacent surface water levels. The vegetated riparian system retains more water and releases it at slower rates than non-vegetated shore zones. This has important implications for stream water quality. Flood surge may be diminished downstream of precipitation events by water movement into non-saturated riparian soils. This would reduce sediment transport capacity, flooding and channel erosion. Water movement into the terrestrial water table is especially important to stream stability in arid regions where rainfall may occur rarely but may lead to devastating floods. Stream-side vegetation moderates the potential impact of local rainfall events by retaining surface runoff. Groundwater can moderate stream temperatures where significant flow is derived from underground sources.

BIOLOGICAL RELATIONSHIPS

Primary production is controlled by the quality and quantity of incident solar radiation, nutrients and plant community structure. In smaller streams with extensive canopies the radiation quantity may be significantly reduced and the wavelength distribution altered. This may reduce production in that section but may at the same time make nutrients more available to downstream organisms. Water temperature will also be affected, and photosynthesis may be reduced by cooler water but also temporarily extended by a reduction in seasonal temperature extremes. Many stream primary producers, especially diatoms and mosses, have adapted to reduced light intensity, and relatively high photosynthetic rates are maintained under low light conditions.

Stream flow characteristics are also affected by debris. Flow rates are moderated by the pool-riffle morphology common to streams with well developed riparian systems. It has been demonstrated that the rate of water movement can be significantly different for a given elevation loss between well developed pool/riffle complexes and streams which allow free water flow. The streams with the most complex morphology retain the water for the greatest period. This has important secondary implications for groundwater, hydrologic regime, water temperature and biota.

Perhaps the most severe effect on water quality following riparian destruction is increased channel sedimentation. Agricultural and forestry practices frequently remove vegetation to the immediate streambank thus allowing unhindered surface water movement directly into the stream. Riparian vegetation will retard surface sheet flow, substantially reducing stream sediment loads. Stream sedimentation results in extreme habitat diversity loss, and the bottom morphology becomes a monotony of fine grained sediments. The immediate biotic symptom may be acute suffocation of the invertebrate fauna with the possibility of chronic physiological stress. The long term effects are extensive. Table II-6-1 prepared by Karr and Schlosser (1978) illustrates the relationships between land use practices and stream sediment loads.

Table II-6-1: POTENTIAL EFFECTS OF VARYING MANAGEMENT PRACTICES ON EQUILIBRIUMS OF EQUIVALENT WATERSHEDS. THESE ARE BEST ESTIMATES OF RELATIVE EFFECTS FOR A VARIETY OF WATERSHED CONDITIONS, INCLUDING SOURCES AND AMOUNTS OF SEDIMENTS.

	Relative Amount of Sediment From			
Management Practice	Land Surface	Stream Channel	Suspended Solids Load in Stream	Source of Sediment
Natural watershed	Very low	Very low	Very low	
Clear land for rowcrop agriculture; maintain natural stream channel	High	Low	Medium	Land surface
Channelize stream in forested watershed	Very low	High	High	Channel banks
Clear land and channelize stream	High	High	Very high	Land surface and channel banks
Best land surface management with channelization	Low	High	Medium to high	Channel banks
Best land surface and natural channel	Low	Low	Low to medium	Equilibrium between land and channel

TABLE II-6-2: COMPARISON OF THE EFFECT OF WELL DEVELOPED AND REDUCED RIPARIAN ZONES ON WATER QUALITY OF SMALL STREAMS

	Flow	Temperature	Sedimentation	Primary Production	Nutrient Load
Riparian system well developed.	1. Extremes moderated 2. Little reaction to local events	1. High and low extremes moderated 2. Reduced daily fluctuations	Moderated by vegetation	Reduced speciation related to organisms able to photosynthesis with reduced light intensity	1. Moderated by riparian uptake 2. Regulated release through highly organic soils. 3. Available supplies because of riparian primary production
Reduced riparian system	1. Erratic flow 2. Reacts to local rain events	1. Extreme seasonal variation 2. Extreme daily fluctuation	Usually higher loads, partic-ularly following watershed disruption	1. Increased production but often of undesirable species. 2. High nutrient loading and temperatures favor undesirable speciation (filamentous blue-green algae or macrophytes)	1. Large seasonal fluctuations 2. Availability to stream biota related to wash out rate, flooding may remove nutrients before they are utilized by aquatic biota

TABLE II-6-2 (Cont'd)

Diversity	No. Individuals	Biomass	Groundwater	Riparian Vegetation	Surface Water
1. Diverse speciation with diverse habitat selection 2. May have large speciation in fish and invertebrates or as common to western streams, large invertebrate population diversity with little fish diversity	May have large number of species with few organisms for each taxa	Diversity of organism types and able to sustain large biomass	Slow change in elevation gaged to changes in stream level	Self sustaining with respect to water, nutrients, habitat etc.	1. Little flooding water generally retained in channel 2. If flood occur, energy dissipated by vegetation
Low species numbers	Large number of organisms for a few taxa	Large biomass with little diversity	1. Rapid change following changes in stream flow 2. Rapid soil drying	Once system degrades may no longer be possible to sustain riparian habitat without extensive reworking of the stream bed and adjacent upland	1. Large scale flooding may occur 2. High energy water flow causing large erosional losses

Several studies have investigated the use of riparian wetlands for waste water treatment. Generally, significant phosphorus and nitrogen reductions occur following varying wetland exposure. EPA Regions IV and V have prepared documentation for generic EIS statements which address the wetland alternative to secondary and tertiary waste treatment technology. Riparian vegetation has also been used to treat urban runoff where it has been found to significantly reduce treatment costs and sediment loads, and to improve water quality and greatly moderate flows.

Recent research has indicated that humic acids released from some riparian ecosystems, particularly wetlands, can significantly affect water quality. Humates are generally large organic molecules which may sequester substances making them biologically unavailable or may, conversely, act as chelating agents making them more available. These phenomena can also occur with toxic materials. Humates may cause considerable oxygen demand and significantly affect such chemical properties as COD. These substances remain largely unclassified and their exact effects unknown.

RIPARIAN CASE HISTORY STUDIES

A long standing controversy has developed in western States where cattle are permitted to graze adjacent to or in both permanent and intermittent streams beds (Behnke 1979). The unprotected riparian vegetation is altered in virtually all respects; species change, biomass is reduced, herbs and shrubs become almost non-existent. A critical question is how this affects water quality and ultimately the fishery. Platts (1982), following an extensive literature review, concluded that studies conducted by fisheries personnel generally found significant biomass and speciation changes following "heavy grazing". Similar studies by range personnel frequently repudiated these results but Platts suggests many were improperly designed or alternative data interpretations are possible. Platts' overall conclusion is "Regardless of the biases in the studies, when the findings of all studies are considered together there is evidence indicating that past livestock grazing has degraded riparian- stream habitats and in turn decreased fish populations".

Studies are underway in the western U.S. testing stream exclosures as means to improve riparian and stream habitat. These are usually qualitative efforts and frequently do not emphasize water quality or stream biota surveys. Hughes (personal communication) observed distinct physical and biological differences between grazed and upgrazed small streams in a study of a Montana watershed. Crouse and Kindschy (1982) have observed consideration variation in riparian vegetation recovery following both long and short term cattle exclosure.

Studies conducted in the Kissimmee-Okeechobee basin, Florida (Council of Environmental Quality 1978), indicate distinct physical and biological differences that follow everglade stream channelization. Nutrients once removed by riparian vegetation make their way to lakes and aid in accelerating eutrophication. The Corps of Engineers (Council of Environmental Quality 1978) is using the Charles River watershed in Massachusetts to control downstream flooding. This project has preserved large riparian watershed tracts to serve as "sponges" to control abnormally high runoff. The preservation of southwestern playas and their vegetation has assumed added importance following realization of their function in groundwater recharge and wildfowl preservation (Bolen 1982). Prarier potholes have long been recognized as critical bird and mammal habitat and recent studies have demonstrated that they too act as nutrient sinks, groundwater recharge areas and as important mechanisms to retain excessive precipitation and surface runoff (van de Valk et al., 1980). Southern bottomland hardwood forests are essential for both indigenous fauna and migratory birds but also are critical water management areas to retain excessive runoff to prevent flooding.

The value of the freshwater tidal riparian zone to aquatic fauna is considerable. Many commercially important anandromous fish require nearly pristine environmental conditions to breed. Perhaps the best documented example is the Pacific Coast Salmonid fishery which is extremely sensitive to physical and chemical alterations. Increased sedimentation and temperatures associated with riparian vegetation removal can destroy a historical fishery. Large number of commercial and non-commercial (sniffen, personal communication) east coast fish depend on extensive freshwater floodplains during their life cycle. South eastern U.S. salt marshes, perhaps an extended riparian definition, are critical for numerous commercially important organisms. The panaeid shrimp totally depend on this environment during the early stages of their life cycle (Vetter, personal communication). It has been hypothesized that these marshes are critical to many near shore organisms through organic carbon export (Odum 1973). Several midwestern fish species also are dependent on riparian habitat, the muskelunge requiring it for completion of their life cycle.

Table II-6-2 is an abbreviated summary of differences between small stream with well developed riparian zones and streams with a reduced riparian zone.

ASSESSMENT OF RELATIONSHIPS BETWEEN RIPARIAN AND AQUATIC SYSTEMS

A variety of methods exist to measure water quality in physical, chemical and biological terms. These are treated in CHapter III-2 and will not be discussed here. Riparian environmental measures are similar to those used in terrestrial ecology (Mueller-Dumbois and Ellenberg 1974).

Ties between the aquatic and riparian or the aquatic, riparian and upland environments can only be estimated. There is a paucity of such information because of the extremely high research costs and the inability to devise procedures to test experimental hypotheses.

The results are that most such evaluations are qualitative. Their quality is based on the integrity and knowledge of the person making the evaluation. The remainder of this section lists physical, chemical and biological factors which might be considered when evaluating the riparian aquatic interaction. It is not meant to be exhaustive but only an example of factors affecting the interactions.

I. Riparian Measures and Their Effect on Water Quality
   A. Geomorphology (erosion, runoff rate, sediment loads)
      1. Slope
      2. Topography
      3. Parent material

B. Soils (sediment loads, nutrient inputs, runoff rates)
   1. Particle size distribution
   2. Porosity
   3. Field saturation
   4. Organic component
   5. Profile (presence or absence of mottling)
   6. Cation exchange capacity
   7. Redox (Eh)
   8. pH

C. Hydrology (water budget, flooding potential, nutrient loads)
   1. Groundwater
      a. Elevation
      b. Chemical quality
      c. Rate of movement
   2. Climatic factors
      a. Total annual rainfall and temporal distribution
         1) Chemical quality
      b. Temperature
      c. Humidity
      d. Light

II. Vegetative and Faunal Characteristics
   A. Floristics ("community health", disturbance levels)
      1. Presence/absence
      2. Nativity
   B. Vegetation (nutrient loads, "community health", disturbance levels)
      1. Production
      2. Biomass
      3. Decomposition
      4. Litter dynamics
         a. Detritus
            1) Size
            2) Transportability
            3) Quantity
      5. Plant size classes
         a. Grasses, herbs (forbs), shrubs, trees
      6. Canopy density and cover
         a. Light intensity
      7. Cover values

   C. Fauna (community disturbance, community health)
      1. Production
      2. Biomass
      3. Mortality

   D. Community structure
      1. Diversity
      2. Evenness

III. Physiological Processes
   A. Transpirational water loss (community health)
   B. Photosynthetic rates (community health)

IV. Streambank characteristics
   A. Stream sinvosity
   B. Stream bank stability (sediment loads, habitat availability)

°SECTION III : CHEMICAL EVALUATIONS

CHAPTER III-1
WATER QUALITY INDICES

One of the most effective ways of communicating information on environmental trends to policy makers and the general public is by use of indices. Many water quality indices have been developed which seek to summarize a number of water quality parameters into a single numerical index. As with all indices the various components need to be evaluated in addition to the single number. U.S. EPA (1978) published an excellent review of water quality indices entitled "Water Quality Indices: A Survey of Indices Used in the U.S." which provides the reader with the types of indices used by various water pollution control agencies. The purpose of this chapter is to identify and explain the various indices that would be applicable to a use attainability analysis. The choice of indices is at the discretion of the States and will primarily be dictated by the water quality parameters traditionally analyzed by the State.

NATIONAL SANITATION FOUNDATION INDEX (NSFI)/WATER QUALITY INDEX (WQI)

Brown et al (1970) presented a water quality index based upon a national survey of water quality experts. In this survey respondents were asked (1) which variables should be included in a water quality index, (2) the importance (weighting) of each variable and (3) the rating scales (sub-index relationships) to be used for each variable. Based on this survey, nine variables were identified: dissolved oxygen pH, nitrates, phosphates, temperature, turbidity, total solids, fecal coliform, and 5-day biochemical oxygen demand. Appropriate weights were assigned to each parameter. The index is arithmetic and is based on the equation:

$WQIA = \sum w_i q_i$

where: $WQIA$ = the water quality index, a number between 0 and 100.

q_i = a quality rating using the rating transformation curve.

w_i = relative weight of the th parameter such that =1.

Figures A-1-9 show the rating curves and relative weights for each of the parameters. To determine the water quality index follow these steps:

(1) determine the measured values for each parameter
(2) determine q for an individual parameter by finding the appropriate value from curves (Figures A 1-9)
(3) multiply by the weight (w) listed on each figure
(4) add the wq for all parameters to determine the water quality index (a number from 0-100)

The water quality index can then be compared to a "worst" or "best" case stream. Examples of a best and worst quality stream cases follow:

Best Quality Stream

	Measured values	Individual quality rating (q_i)	Weights (w_i)	Overall quality rating ($q_i \times w_i$)
DO, percent sat.	100	98	0.17	16.7
Fecal coliform density, # /100 ml	0	100	0.15	15.0
pH	7.0	92	0.11	10.1
BOD mg/l	0.0	100	0.11	11.0
Nitrate, mg/l	0.0	98	0.10	9.8
Phosphate, mg/l	0.0	98	0.10	9.8
Temperature °C departure from equil	0.0	94	0.10	9.4
Turbidity, units	0	98	0.08	7.8
Total solids, mg/l	25	84	0.08	6.7

$$WQI = \sum w_i q_i = 96.3$$

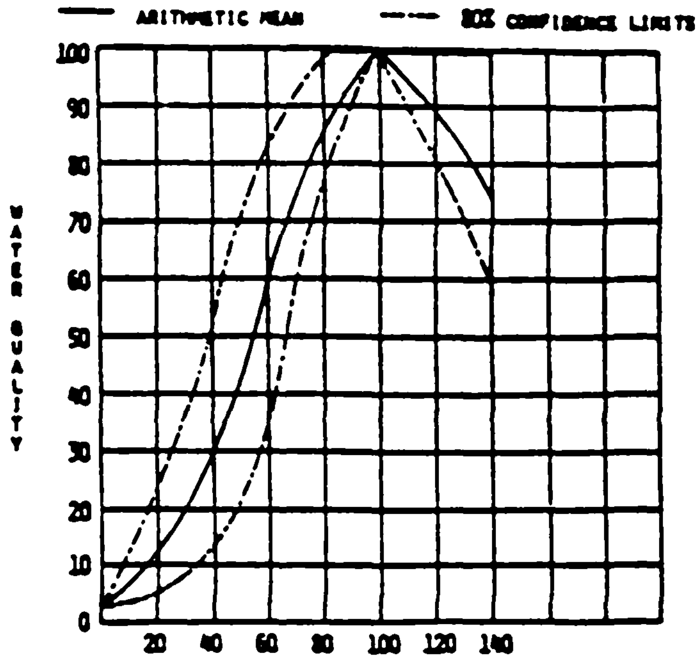
Worst Quality Stream

Parameters	Measured values	Individual quality rating (q_i)	Weights (w_i)	Overall quality rating ($q_i \times w_i$)
DO. percent sat.	0	0	0.17	0
Fecal coliform density, # /100 ml	5	4	0.15	0.6
pH	2	4	0.11	0.4
BOD , mg/l	30	8	0.11	0.9
Nitrate, mg/l	100	2	0.10	0.2
Phosphate, mg/l	10	6	0.10	0.6
Temperature °C departure from equil	+15	10	0.10	1.0
Turbidity, units	100	18	0.08	1.4
Total solids, mg/l	500	20	0.08	2.4

$$WQI = \sum w_i q_i = 7.5$$

WATER QUALITY INDEX

DISSOLVED OXYGEN



DISSOLVED OXYGEN, % SATURATION

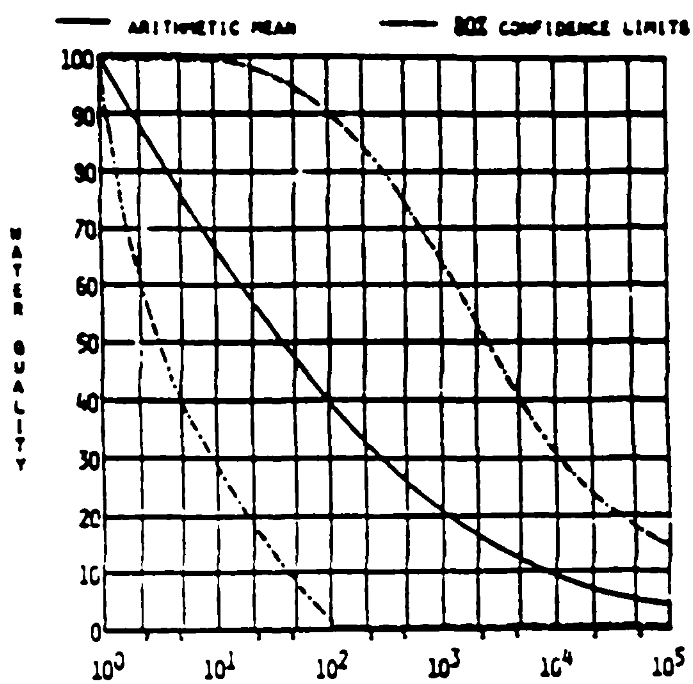
Figure A-1

w = 0.17

NOTE:
FOR D.O. >140% SAT,
q_i = 50.

WATER QUALITY INDEX

FECAL COLIFORMS



FECAL COLIFORMS, AVERAGE NUMBER OF ORGANISMS PER 100 ML

Figure A-2

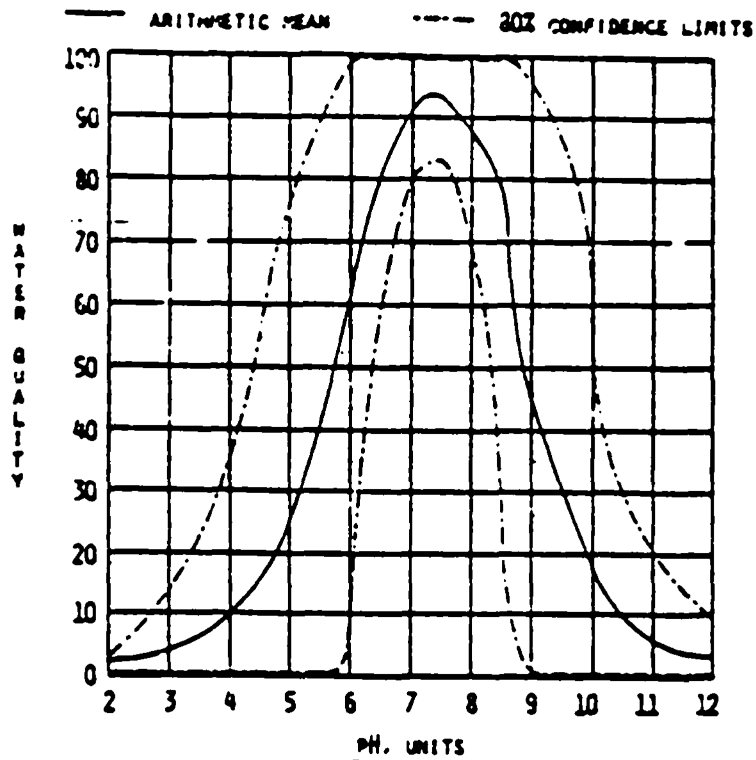
w = 0.15

KEY FOR LOG SCALE INTERPOLATION

NOTE:
FOR F.C. >10(5)/100 ml
q_i = 2.

WATER QUALITY INDEX

pH

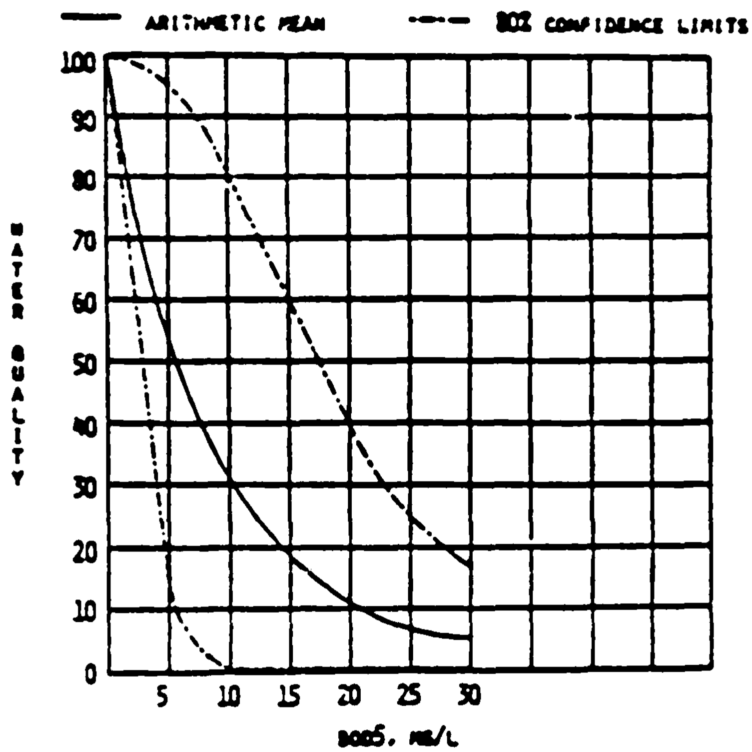


$w = 0.12$

Figure A-3

WATER QUALITY INDEX

BOD5



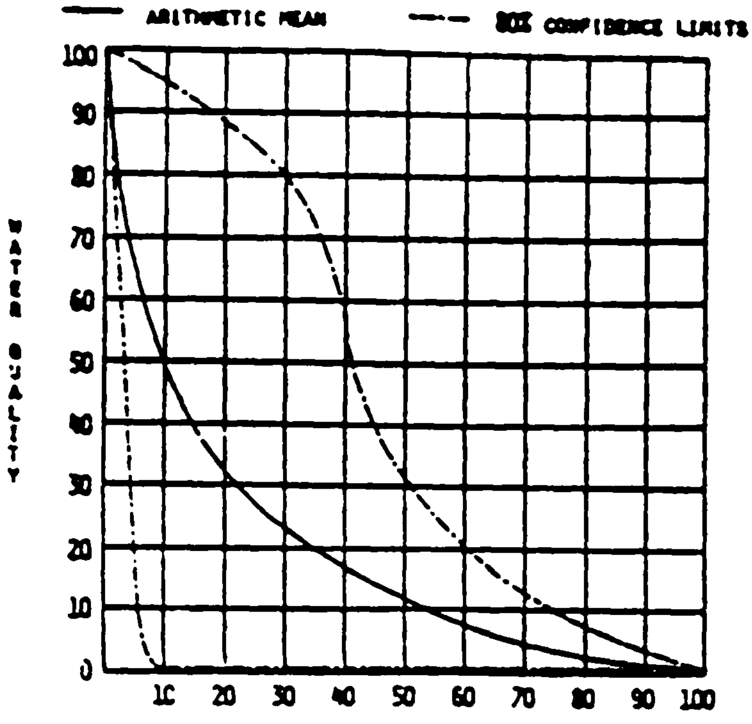
$w = 0.10$

NOTE:
FOR $BOD_5 > 30$,
$q_1 = 2$.

Figure A-4

WATER QUALITY INDEX

NITRATES



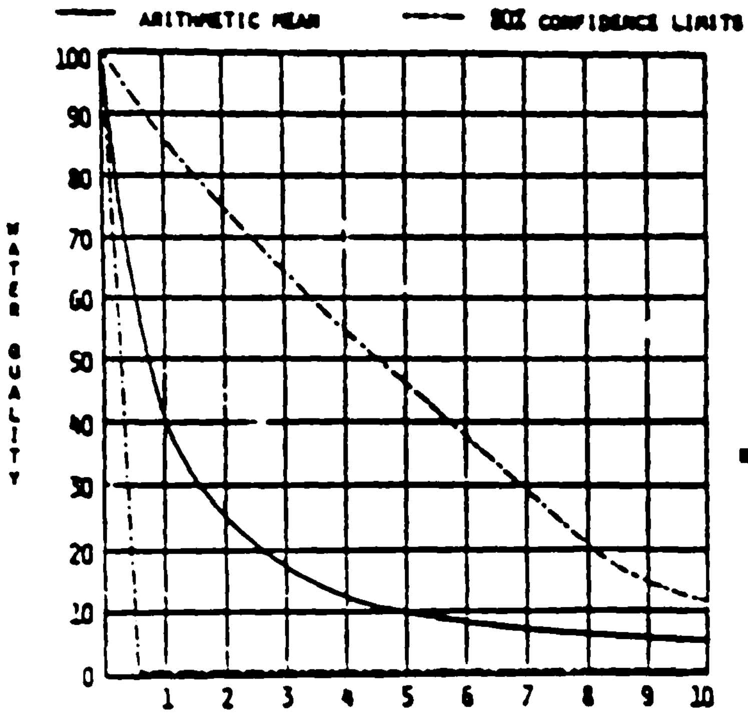
w = 0.10

NOTE:
FOR NITRATES >100 MG/L
q_i = 1.

Figure A-5

WATER QUALITY INDEX

TOTAL PHOSPHATES



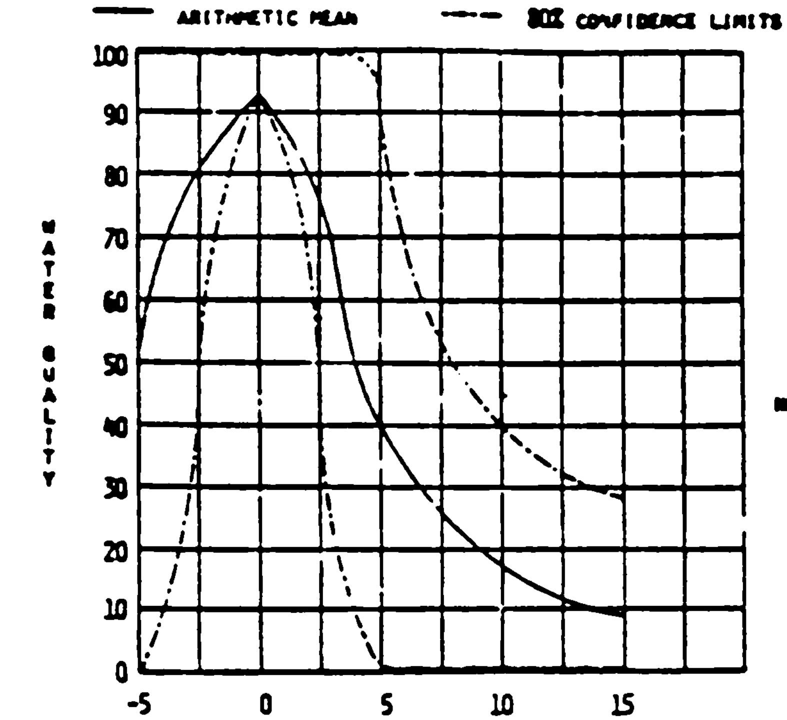
w = 0.10

NOTE:
FOR TOTAL PHOSPHATES
>10 MG/L, q_i = 2.

Figure A-6

WATER QUALITY INDEX

TEMPERATURE



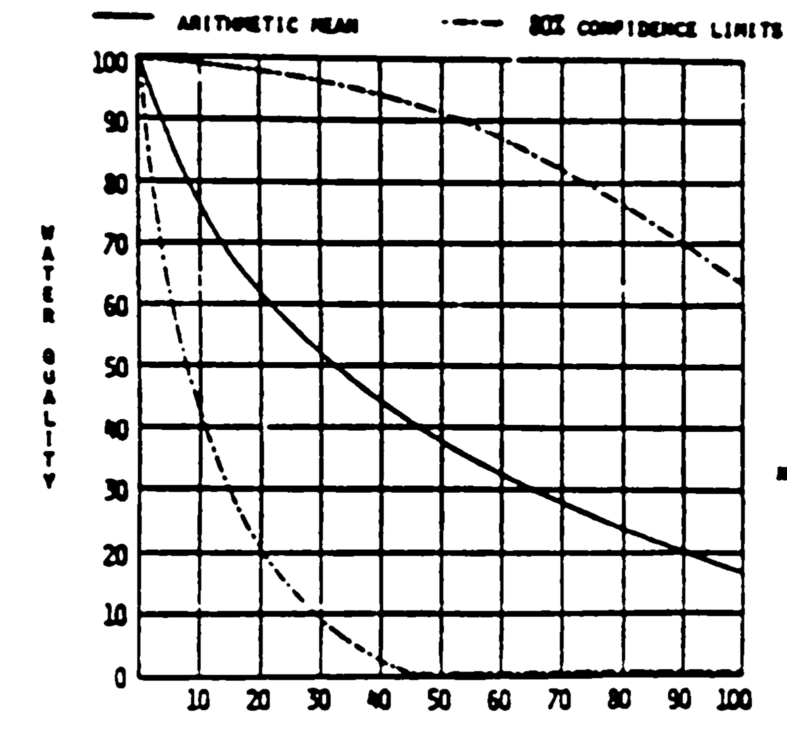
w = 0.10

NOTE:
FOR TEMPERATURE DEVIATION >15°C, q_i = 5.

Figure A-7

WATER QUALITY INDEX

TURBIDITY



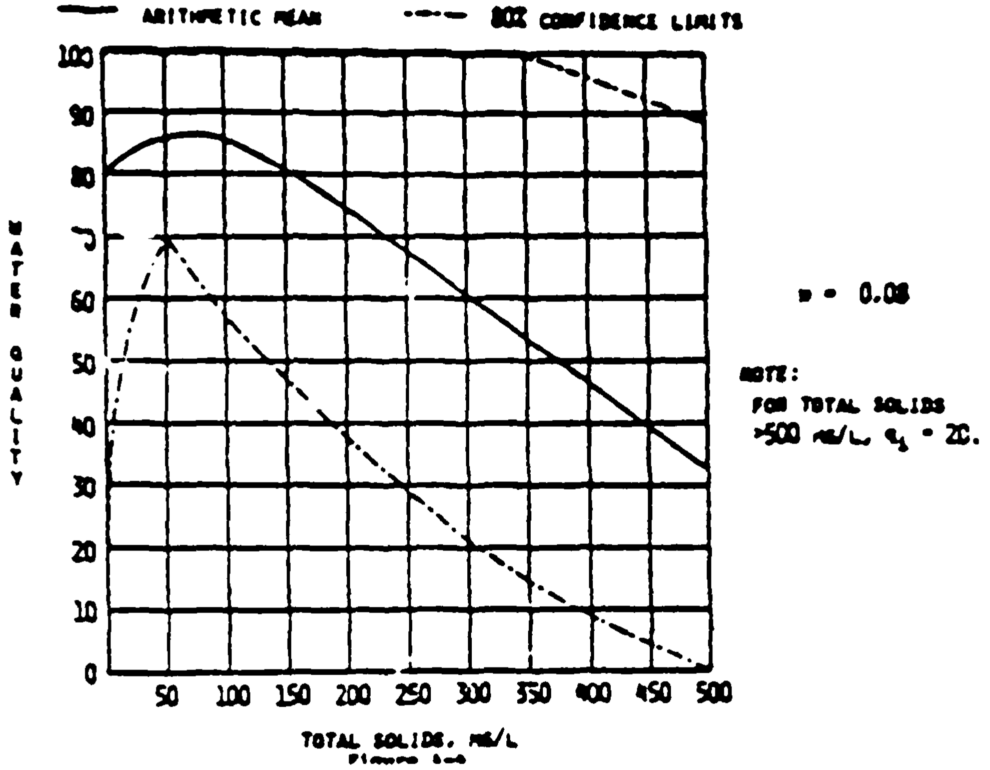
w = 0.08

NOTE:
FOR TURBIDITY >100 JTU
q_i = 5.

Figure A-8

WATER QUALITY INDEX

TOTAL SOLIDS



DINIUS WATER QUALITY INDEX

In 1972, Dinius proposed a water quality index as part of a larger social accounting system designed to evaluate water pollution control expenditures. This index includes 11 variables and like the NSFI, it has a scale which decreases with increased pollution, ranging from 0 to 100. The index is computed as the weighted sum of its sub indices. The 11 variables included in the index are: dissolved oxygen, biological oxygen demand, Eschericia coli, alkalinity, hardness, specific conductivity, chlorides, pH, temperature, coliform, and color. This index is unique in that the calculated water quality index could be matched to specific water uses. Dinius proposed different descriptor language for different index ranges depending on the specific water use under consideration as illustrated in Figure A-100. The index values can be derived from the following formula:

$$Q = \frac{5(DO) + 214(BOD)^{-0.642} + 400(5E.Coli)^{-0.30} + 300(Coli)^{-0.30} + 535\,(SC)^{-0.3565} + 62.9(Cl)^{-0.207} + 10^{1.974-0.00132(HA)} + 54(ALK)^{-0.178} + 10^{0.255\,pH + 0.440} + 8(Ta-Ts) + 224 + 128(C)^{-0.288}}{5 + 2 + 4 + 3 + 1 + .5 + 1 + .5 + 1 + 2 + 1}$$

Note: If the pH is between 6.7 and 7.3, 100 should be substituted for for the pH expression. If pH is greater than 7.3, the pH expression should be 10

DO = dissolved oxygen in percent saturation
BOD = biological oxygen demand in mg/l
E.coli = Eschericia coli as E.coli per ml
Coli = coliform per ml
SC = specific conductivity expressed in microhms per cm at 25°C
Cl = chlorides in mg/l
HA = hardness as ppm CaCO
ALK = alkalinity as ppm CaCO
pH = pH units
Ta = actual temperature
Ts = standard temperature (average monthly temperature)
C = Color units

Once the quality unit is determined based on the above calculation, a comparison to Figure A-10 should reveal the quality of the water for a specific use.

HARKINS/KENDALL WATER QUALITY INDEX

A statistical index was developed by Harkins (1974) using a nonparametric classification procedure developed by Kendall (1963). The procedure was summarized by Harkins by the following four steps:

(1) For each water quality parameter used, choose a minimum or maximum value as a starting point. This sector of values is the control observation from which standardized distances will be computed.

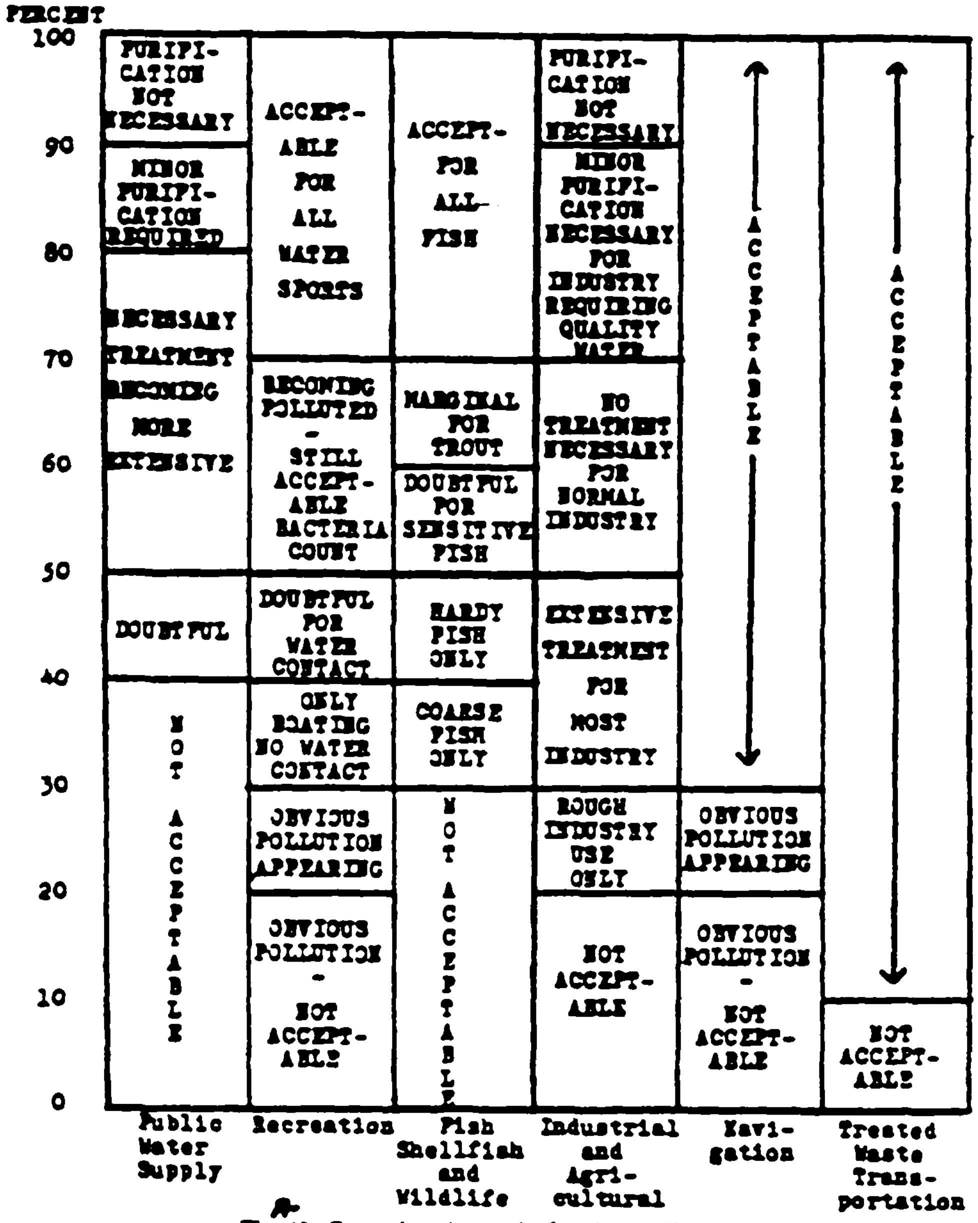


Fig. 18 General rating scale for the quality unit.

(2) Rank each column of water quality parameters including the control value. Tied ranks are split in the usual manner.

(3) Compute the rank variance for each parameter using the equation:

$$\text{Variance } (R_i) = \frac{1}{12n} \times \left[(n^3 - n) - \sum_{i=1} (t_k^3 - t_k)\right]$$

where: $i = 1,2...p,$
p = the number of parameter being used
n = the number of observations plus the number of control points, and
k = the number of ties encountered.

These variances are used to standardize the indices computed.

(4) For each member of observation vector, compute the standardized distances:

$$S_n = \sum_{i=1}^{p} (R_i - R_i)^2 \; / \; (R_i),$$

where R is the rank of the control value.

This index is meant as a method for summarizing a large amount of data to present a concise picture of overall trends. This method provides a simple, expedient method whereby one station can be compared with another or previous time periods from a particular station may be compared with another time period at the same station. A detailed example of this index may be found in Harkins (1974).

OTHER INDICES

Many other water quality indices have been developed; some being variations of the indices described previously. Several States (Georgia, Oregon, Nevada, Illinois) have developed their own systems based on the characteristics of the water bodies of the State. McDuffie and Haney (1973) proposed an eight-variable water quality index which was applied to streams in New York State.

CHAPTER III-2

pH, HARDNESS, ALKALINITY AND SALINITY

INTRODUCTION

The chemical composition and the chemical interactions of the aquatic environment exert an important influence on the aquatic life of a water body. Many chemical constituents in a body of water have the ability to alter the toxicity of specific pollutants, or to protect organisms from toxic materials by removing them or by blocking their action. The importance to aquatic life of four water quality parameters - pH, alkalinity, hardness and salinity - is discussed in this section.

pH

The pH of water is a measure of its acid or alkaline nature. Specifically, it is an expression of the hydrogen ion activity of the solution. Hydrogen ion activity is mathematically related to the hydrogen ion concentration $[H^+]$, and for most natural waters these may be considered equivalent. pH is expressed as the negative logarithm of the hydrogen ion concentration:

$$pH = -\log [H^+]$$

The water molecule, H_2O, ionizes to yield one hydrogen and one hydroxyl ion:

$$H_2O \rightleftharpoons H^+ + OH^-$$

The equilibrium expression for this reaction is:

$$K = \frac{[H^+][OH^-]}{[H_2O]}$$

The concentration of water, $[H_2O]$, is considered to be a constant, and the equation simplifies to:

$$K_w = [H^+][OH^-] = 10^{-14}$$

Because the product of the concentration of both ions is always 10^{-14}, when they are equal to each other,

$$[H^+] = [OH^-] = 10^{-7}, \text{ and}$$

$$pH = -\log (10^{-7}) = 7.$$

At pH 7 the solution is neutral. When there are more hydrogen ions than hydroxyl ions, the pH is less than 7 and the solution is acidic. When there are more hydroxyl ions, the pH is greater than 7 and the solution is alkaline.

The pH of most natural freshwaters in the U.S. is between 6 and 9. It is interesting to note that the pH of most ocean waters falls in a much narrower range, 8.1 to 8.3 (Warren 1971). This is due to the presence of several buffering systems in salt water which control pH changes. In freshwater, pH is regulated primarily by the carbonate buffer system. Biological activities such as photosynthesis or respiration can cause significant diel variations in pH. Extreme pH values or variations in pH can be caused by pollution such as acid mine drainage.

Importance to Aquatic Life

The importance of pH to aquatic organisms resides primarily in its effect on other environmental factors. In general, the change in pH itself is not directly harmful. Rather, the impact on aquatic life accompanies a change in an associated variable such as the solubility or toxicity of a toxic pollutant. The pH range 6.5-9.0 is considered to be generally protective for fish and the range 5.0-9.0 is not considered directly lethal (EIFAC 1965).

Aquatic organisms have protective membranes and internal regulatory systems which afford a degree of protection from the direct effects of hydrogen and hydroxyl ions. The indirect effects of pH seem to intensify as the pH deviates from the optimum (EIFAC 1969).

The degree of dissocation of weak acids is pH-dependent and thus the toxicity of several common pollutants is affected. Ammonia (NH_3), hydrogen sulfide (H_2S), and hydrocyanic acid (HCN) are examples. Under low pH conditions the NH_3 molecule ionizes and becomes the NH_4^+ ion (Thurston, et al. 1974). The toxicity of ammonia is attributed to the un-ionized form (NH_3), so that increased pH conditions result in increased levels of the toxic un-ionized fraction.

The lower the pH, the smaller the degree of dissociation of hydrocyanic acid to hydrogen and cyanide ions. The molecular form (HCN) is the toxic form, and so the toxicity of cyanide is favored by low pH. The undissociated form of hydrogen sulfide (H_2S) is the primary source of sulfide toxicity. Therefore, under low pH conditions, very little H_2S is dissociated, and toxicity is increased.

The solubility of toxic metals is a function of pH. Metals in water tend to form complexes with such anions as sulfate, carbonate or hydroxide. The solubility of these complexes increases with decreasing pH, as illustrated for hydroxides in Figure III-2-1, so that low pH conditions may cause the release of metals from sediment deposits into the water column. Metal toxicity is believed to be related to the total metal concentration (i.e., free ions plus complexed ions) in solution (Calavari et al. 1980). Table III-2-1 illustrates the effect of pH on metal concentrations in natural waters.

Due to the complexity of its interactions with elements of the environment, there may be several mechanisms by which pH affects toxicity. The exact mecha-

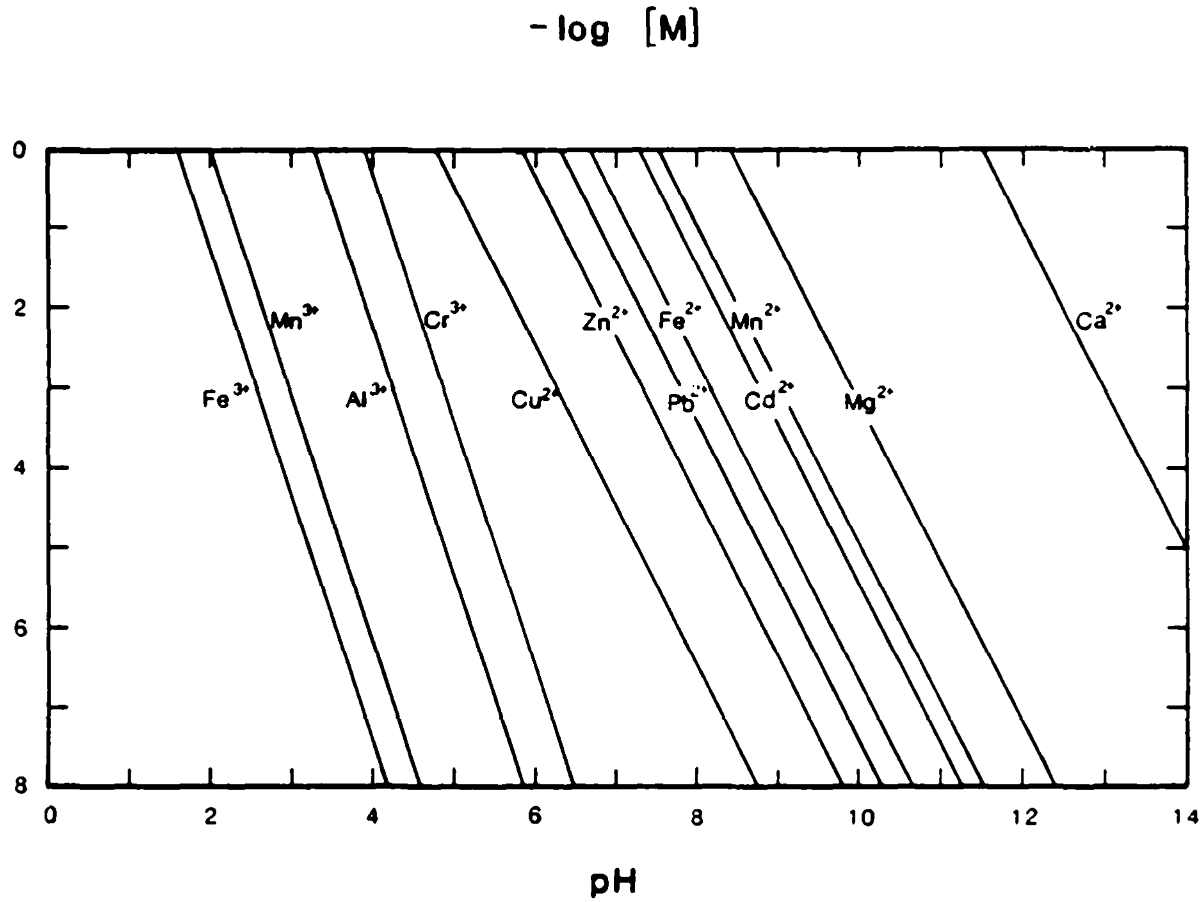


Figure III-2-1. Relationship Between pH and Solubility of Metallic Hydroxides

TABLE III-2-1. CONCENTRATION (ug/l) OF METALS IN LAKE WATERS OF VARIOUS ACIDITIES (From Haines, 1981).

Locality	Metal						
	Al	Cu	Cd	Mn	Ni	Pb	Zn
				Nonacidified (pH 6.0–7.8)			
102 lakes, Ontario (average)		2	<0.1	3	<3	<1	<1
Blue Chalk Lake, Ontario	13	8		40	3		9
Lake Panache, Sudbury, Ontario		6			28		6
North Sweden (range)	<50		0.05–0.23	<100			10–30
Central Norway (range)		1–10	0–0.5			0–5	1–17
North Norway (range)	<20–65						
				Intermediate (pH 5.5–6.0)			
South-central Ontario, 14 lakes (average)		5.7		49	3.6		12.6
Nelson Lake, Ontario	13	13		18	10		16
				Acidified (pH 4.1–5.3)			
Four lakes, Ontario (average)		3	0.4	239	10	2	30
Clearwater Lake, Sudbury, Ontario	453	97		300	213		46
Four lakes, Sudbury, Ontario (average)		450		338	820		83
West coast Sweden (range)	200–600		0.08–0.63	300–400		1–5	30–122
Southeast Norway (range)		1–10	0–0.6			1–10	3–35
Lake Langtjern, Norway (average)	218	6	0.21			2	15
South Norway (range)	50–600						
Adirondack lakes, New York (average)	286			45			23
South Norway (range)	40–600						
Laxforsen, west Sweden	288	1	0.2	190		3	28

nism of direct toxicity of pH in water is not certain. It has been suggested that at very low pH values, oxygen uptake may be affected and this may be the toxic event. Acid-base regulation and ionoregulation appear to be affected at higher, but still acidic, pH values (Graham and Wood 1981). There is evidence that the chronic effects of pH on fish include effects on reproduction, such as reduced egg production and hatchability (Peterson, et al. 1980), and on behavior (Mount 1973). Some mobile organisms may have the ability to avoid low pH conditions if the detrimental conditions are localized. Evidence suggests (U.S. EPA 1960, p. 180) that outside a range of 6.5 to 9.0, fish suffer adverse physiological effects which increase in severity as the degree of deviation increases. Tables III-2-2 and III-2-3 present pH values that have been found to cause adverse effects on a number of fish species in the field and in laboratory investigations, respectively. These values represent only the low end of the tolerated range of pH. (The lower limit is most often exceeded due to anthropogenic causes such as acid rainfall, acid mine drainage and industrial discharges.)

Marine organisms, as a group, tend to be much less tolerant of extreme pH conditions. As mentioned previously, the marine environment is buffered more effectively than freshwater. As a result, these organisms have not evolved an ability to cope with pH variations outside their narrow optimum range.

ALKALINITY

Alkalinity is the property of water which resists or buffers against changes in pH upon addition of acid or base. The primary buffer in freshwater is the carbonate-bicarbonate system. Phosphates, borates, and organic acids also impart buffer capacity to water. These additional buffer systems are more significant in saltwater than in freshwater.

Bicarbonate (HCO_3^-) is the major form of alkalinity. Carbon dioxide (CO_2) dissolved in water is carbonic acid (H_2CO_3). Carbonic acid dissociates in two steps to form bicarbonate and carbonate ($CO_3^=$) ions as follows:

$$CO_2 + H_2O \rightleftharpoons H_2CO_3 \rightleftharpoons H^+ + HCO_3^-$$

$$HCO_3^- \rightleftharpoons H^+ + CO_3^=$$

The ability of these chemical reactions to shift back and forth with changes in hydrogen ion concentration (pH) to "absorb" these changes is what imparts buffer capacity. This system tends to control pH best in the neutral range.

The form of alkalinity in solution is governed by pH. Figure III-2-2 illustrates this effect. Biological activities such as photosynthesis and respiration cause shifts in pH and in the relative concentrations of the forms of alkalinity, without significant effect on the total alkalinity. The production of CO_2 during respiration shifts the equilibrium to the right, toward carbonate formation. The removal of CO_2 from solution during algal photosynthesis shifts the alkalinity equilibrium to the left, toward the bicarbonate form.

TABLE III-2-2. SPECIES OF FISH THAT CEASED REPRODUCING, DECLINED, OR DISAPPEARED FROM NATURAL POPULATIONS AS A RESULT OF ACIDIFICATION FROM ACID PRECIPITATION, AND THE APPARENT pH AT WHICH THIS OCCURRED (From Haines, 1981).

Family and species	Apparent pH at which population ceased reproduction, declined, or disappeared
Salmonidae	
Lake trout *Salvelinus namaycush*	5.2–5.5 ; 5.2–5.8 ; 4.4–6.8
Brook trout *Salvelinus fontinalis*	4.5–4.8 ; ~5
Aurora trout *Salvelinus fontinalis timagamiensis*	5.0–5.5
Arctic char *Salvelinus alpinus*	~5
Rainbow trout *Salmo gairdneri*	5.5–6.0
Brown trout *Salmo trutta*	5.0 ; 5.0–5.5 ; 4.5–5.5
Atlantic salmon *Salmo salar*	5.0–5.5
Lake herring *Coregonus artedii*	4.5–4.7 ; <4.7 ; 4.4
Lake whitefish *Coregonus clupeaformis*	<4.4
Esocidae	
Northern pike *Esox lucius*	4.7–5.2 ; 4.2–5.0
Cyprinidae	
Golden shiner *Notemigonus crysoleucas*	4.8–5.2
Common shiner *Notropis cornutus*	<5.7
Lake chub *Couesius plumbeus*	4.5–4.7
Bluntnose minnow *Pimephales notatus*	5.7–6.0
Roach *Rutilus rutilus*	5.5–5.7
Catostomidae	
White sucker *Catostomus commersoni*	4.7–5.2 ; 4.2–5.0
Ictaluridae	
Brown bullhead *Ictalurus nebulosus*	4.5–5.2 ; 4.6–5.0
Percopsidae	
Trout-perch *Percopsis omiscomaycus*	5.2–5.5
Gadidae	
Burbot *Lota lota*	5.5–6.0 ; 5.2–5.8
Centrarchidae	
Smallmouth bass *Micropterus dolomieui*	5.5–6.0 ; >5.5 ; ~5.8 ; 4.4–5.0
Largemouth bass *Micropterus salmoides*	4.4–5.2
Rock bass *Ambloplites rupestris*	4.7–5.2 ; 4.2–5.0
Pumpkinseed *Lepomis gibbosus*	4.7–5.2 ; <4.2
Bluegill *Lepomis macrochirus*	<4.2
Percidae	
Johnny darter *Etheostoma nigrum*	5.0–5.9
Iowa darter *Etheostoma exile*	4.8–5.9
Walleye *Stizostedion v. vitreum*	5.5–6.0 ; 5.2–5.8
Yellow perch *Perca flavescens*	4.5–4.8 ; <4.7 ; 4.2–4.4
European perch *Perca fluviatilis*	5.0–5.5

TABLE III-2-3. VALUES OF pH FOUND IN LABORATORY EXPERIMENTS TO CAUSE VARIOUS ADVERSE EFFECTS ON FISH SPECIES (From Haines, 1981).

Family and species	Increased mortality: Embryo	Increased mortality: Fry	Increased mortality: Juveniles or adults	Reduced growth	Other effects
Salmonidae					
Brook trout	6.5	4.4	4.5	6.5	Reduced egg viability: 5.0
	5.6	4.5	4.1	4.6	Tissue damage: 5.2
	4.5	6.1	3.5		
Arctic char				4.8	
Rainbow trout	5.5	4.3	3.6–4.1	4.8	
Brown trout	4.0	5.0			
	4.1				
Atlantic salmon	3.4–4.4	4.0			Tissue damage: 5.0
	3.6	4.3			
	3.9	4.3			
	4.0	5.0			
	4.0–5.5				
	4.1				
Esocidae					
Northern pike	5.0				
Cyprinidae					
Roach	5.6				
Fathead minnow	5.9	5.9	2.1	4.5	Reduced egg viability: 6.6
Catostomidae					
White sucker	4.5	5.3		4.5	Ceased feeding: 4.5
		4.0			Bone deformity: 4.2 : 5.0
Percidae					
European perch	5.6				
	5.5				

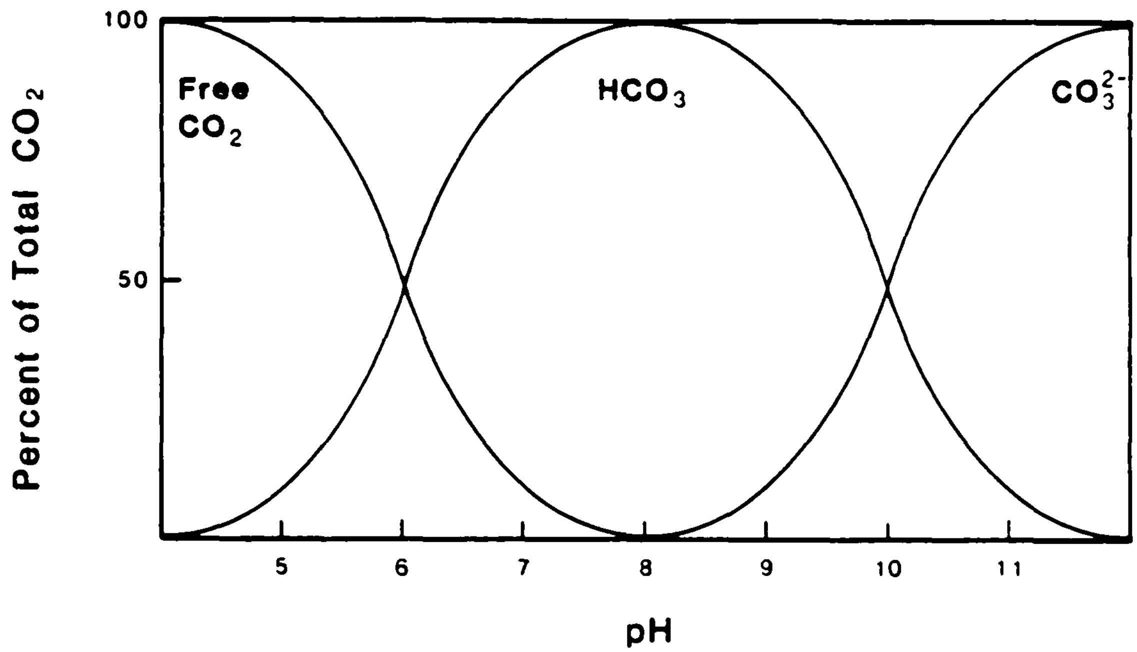


FIGURE III-2-2. The relationship between pH and the forms of CO_2 in water.

Importance to Aquatic Life

The forms of alkalinity are biologically significant because they serve as a source of the essential elements carbon, oxygen, and hydrogen. When free CO_2 is not available, algae are capable of using bicarbonate as their carbon source. Free CO_2 in solution regulates a variety of biological processes such as seed germination, plant growth (photosynthesis), respiration, and oxygen transport in the blood.

Alkalinity is critical to the maintenance of healthy conditions in aquatic systems, particularly where they are stressed by pollution. Alkalinity helps to maintain pH in the optimum range for biological activities. The impact of acidic wastes such as coal ash or basic wastes such as metal plating discharges can be moderated to a degree by the natural buffering capacity of the receiving water. The indirect effects of alkalinity on toxicity are also important. In particular, alkalinity reacts with the toxic soluble metal fraction in water to form insoluble carbonate and hydroxide precipitates. Figure III-2-3 illustrates that the concentration of heavy metals drops rapidly as the concentration of carbonate increases. Metals which are precipitated from the water column are effectively removed from the aquatic environment and no longer represent an immediate source of toxicity to aquatic life.

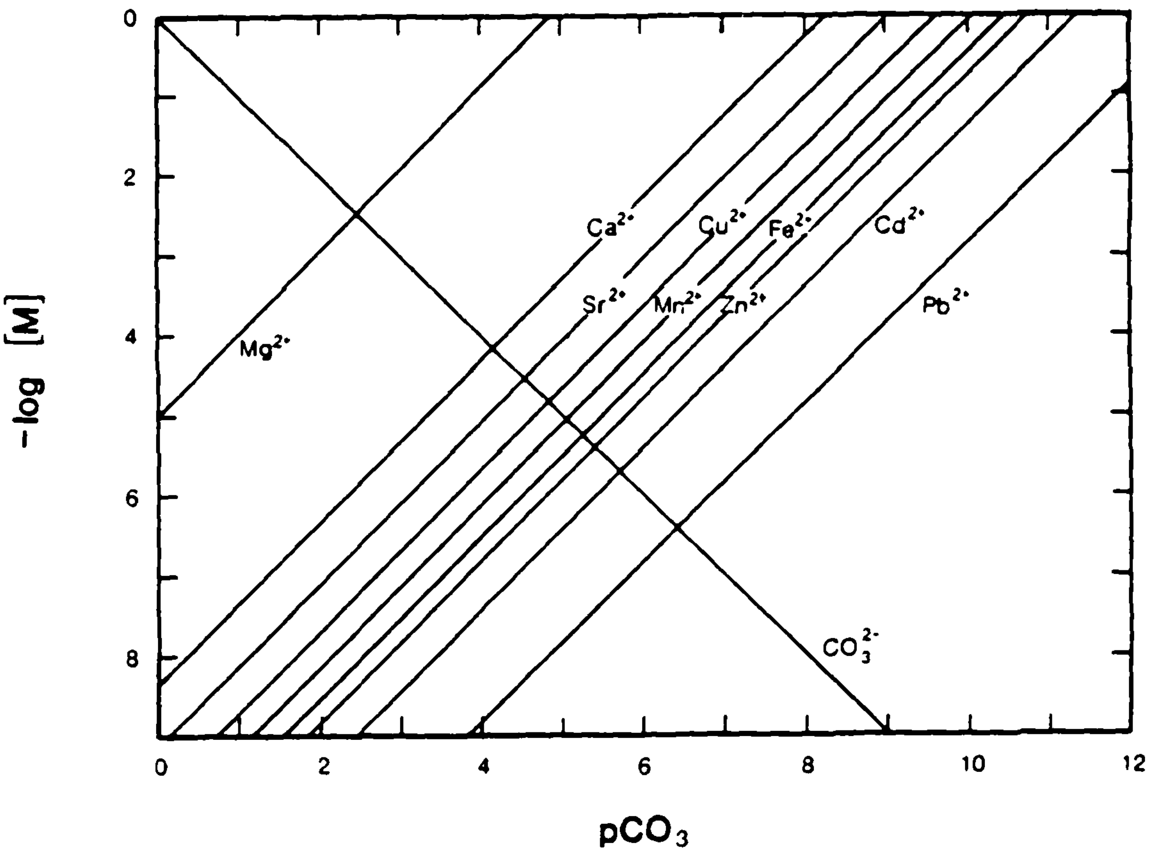


Figure III-2-3. Relationships of metallic carbonate solubility and carbonate concentrations

HARDNESS

Water hardness generally refers to the capacity of the water to precipitate soap from solution. The constituents which impart hardness to water are polyvalent cations, chiefly calcium (Ca) and magnesium (Mg). These form insoluble complexes with a variety of anions, notably the salts of organic acids (soaps). By convention, hardness is reported on the basis of equivalence as mg/l calcium carbonate ($CaCO_3$).

Hardness cations are primarily associated with carbonate or sulfate anions. Calcium and magnesium carbonate are referred to as carbonate hardness. When the anion is other than carbonate, such as sulfate or nitrate, this is referred to as noncarbonate hardness. Because alkalinity and hardness are both expressed as mg/l $CaCO_3$, it can be concluded that carbonate alkalinity will be responsible for forming carbonate hardness and that hardness in excess of the alkalinity is noncarbonate.

Importance to Aquatic Life

Hardness, the capacity of water to precipitate soap, is an aesthetic consideration important to potable water supply. The importance of hardness to aquatic life is related to the ions which impart hardness to water. There is some evidence to suggest that hard water environments are more favorable for aquatic life because they support more diverse and abundant biological communities (Reid 1961).

There is a large body of evidence that hardness mediates the toxicity of heavy metals to aquatic organisms. Mathematical correlations between the toxicity of several heavy metals (Cr^{+3}, Pb, Ag, Ni, Zn, Cd, and Cu) have been developed. Table III-2-4 presents the equations (taken from the Water Quality Criteria Documents) which enable the calculation of allowable metal concentrations as a function of hardness. Although increased hardness can be correlated directly with decreased toxicity, the mechanism of this effect is not certain. Two different mechanisms have been proposed, one chemical and one biological. Calamari, et al. (1980) have reviewed the literature concerning these mechanisms, and discussed both with regard to their own experimental data.

Hardness may operate through two chemical mechanisms to reduce heavy metal toxicity. Complexation of the toxic metal with carbonate might be the mechanism if the free metal ion is the toxic species. Data may be found in the literature to support (Stiff 1971, Pagenkopf et al. 1974, Calamari and Marchetti 1975, Andrew et al. 1977), or contradict (Shaw and Brown 1974, Calamari et al. 1980) this suggestion. It is also possible that it is the calcium or magnesium ion alone, rather than the associated carbonate, that is protective. Carroll et al. (1979) present data which show that the calcium ion, much more than magnesium, seems to reduce cadmium toxicity to brook trout.

Further, the question remains whether the hardness ions are antagonistic to the action of the toxic metals and they may function biologically through competitive inhibition of metal uptake or binding of sites of action. Kinkade and Erdman (1975) published data to support the uptake inhibition mechanism. Lloyd (1965) suggests that calcium has a protective effect on fish gill tissue, an organ which is significantly involved in heavy metal uptake. Calcium has been shown to decrease gill permeability to water, which would influence metal uptake (Maetz and Bornancin 1975).

TABLE III-2-4. DEPENDENCE OF HEAVY METAL TOXICITY ON WATER HARDNESS*

Metal	Calculation of Maximum Allowable Concentration
Cadmium (Cd)	$e^{(1.05[\ln (\text{hardness})]-3.73)}$
Chromium (Cr^{+3})	$e^{(1.08[\ln (\text{hardness})]+3.48)}$
Copper (Cu)	$e^{(0.94[\ln (\text{hardness})]-1.23)}$
Lead (Pb)	$e^{(1.22[\ln (\text{hardness})]-0.47)}$
Nickel (Ni)	$e^{(0.76[\ln (\text{hardness})]+4.02)}$
Silver (Ag)	$e^{(1.72[\ln (\text{hardness})]-6.52)}$
Zinc (Zn)	$e^{(0.83[\ln (\text{hardness})]+1.95)}$

* EPA Ambient Water Quality Criteria Documents (1980).

There is evidence that calcium may be protective against the toxic action of pollutants other than metals. Hillaby and Randal (1979) found that increased calcium concentration decreased the acute toxicity of ammonia to rainbow trout. Calcium concentration has also been associated with increased survival of fish in acidic conditions (Haranath et al. 1978).

SALINITY

Salinity is a measure of the weight of dissolved salts per unit volume of water. The chloride content of water, the chlorinity, is strongly correlated with salinity. In freshwater, the total concentration of ionic components constitutes salinity. The major anions are commonly carbonate, chloride, sulfate, and nitrate. The predominant associated cations are sodium, calcium, potassium, and magnesium.

The source of these materials is the substrate upon which the water lies and the earth through and over which water flows. The salinity of a given body of water is a function of the quantity and quality of inflow, rainfall, and evaporation.

Importance to Aquatic Life

Salinity has an impact on a variety of parameters related to biological functions. It controls the ability of organisms to live in or pass through various waters. It also has an effect on the presence of various food or habitat-forming plants.

Salinity is important not only in an absolute sense, but the degree of variation in the salinity of a given water is biologically important. The invasion of species to or from fresh or saltwater depends on their ability to tolerate changes in salinity. Rapid changes in salinity cause disruption of osmoregulation in aquatic organisms and can cause plasmolysis in plants. Organisms that can tolerate a range of salinity can frequently use salinity gradients to evade less tolerant predators.

Salinity is important to the heat capacity of aquatic systems. As salinity increases, the specific heat of water decreases. This means that there is less heat required to warm the water. Temperature is a significant factor in biological activity and governs many physical processes in water as well.

Salinity also governs the dissolved oxygen concentration in water. For a given temperature, the solubility of oxygen decreases with increasing salinity. Table III-2-6 illustrates this effect. The dissolved oxygen concentration is among the most critical of all water quality parameters to aquatic life.

The ions which make up the total salinity of water have individual effects as well. The effects of calcium, magnesium, and carbonate have been discussed previously with respect to their effect on the toxicity of pollutants. Several of the ions (e.g., nitrate, and potassium) are plant nutrients.

Aquatic organisms have evolved a variety of physiological adaptations to the salinity of their environments. These adaptations are largely related to their osmoregulatory systems whose primary function is to solve the problem of the difference between the salt concentration of the internal fluids of the organism and the salt concentration of the surrounding water. Freshwater organisms must maintain an internal salt concentration against the tendency to gain water from and lose salts to the environment. Osmoregulation in freshwater fish results in the production of high volumes of liquid waste with a low salt concentration. In contrast, marine organisms must maintain an internal salt concentration that is lower than that of the environment, against a tendency to lose water and gain salts. Osmoregulation in salt water fish results in the production of small volumes of liquid waste carrying a relatively high salt concentration.

The gills and kidneys of both types of fish are specially developed to accomplish these actions against the natural environmental gradient. Therefore, the nature of these systems governs the ability of organisms to survive in regions of varying salinity or to successfully migrate through them.

TABLE III-2-5. SOLUBILITY OF DISSOLVED OXYGEN IN WATER IN EQUILIBRIUM WITH DRY AIR AT 760 mm Hg AND CONTAINING 20.9 PERCENT OXYGEN.

Tempera-ture. °C	Chloride concentration. mg/l				
	0	5000	10.000	15.000	20.000
0	14.6	13.8	13.0	12.1	11.3
1	14.2	13.4	12.6	11.8	11.0
2	13.8	13.1	12.3	11.5	10.8
3	13.5	12.7	12.0	11.2	10.5
4	13.1	12.4	11.7	11.0	10.3
5	12.8	12.1	11.4	10.7	10.0
6	12.5	11.8	11.1	10.5	9.8
7	12.2	11.5	10.9	10.2	9.6
8	11.9	11.2	10.6	10.0	9.4
9	11.6	11.0	10.4	9.8	9.2
10	11.3	10.7	10.1	9.6	9.0
11	11.1	10.5	9.9	9.4	8.8
12	10.8	10.3	9.7	9.2	8.6
13	10.6	10.1	9.5	9.0	8.5
14	10.4	9.9	9.3	8.8	8.3
15	10.2	9.7	9.1	8.6	8.1
16	10.0	9.5	9.0	8.5	8.0
17	9.7	9.3	8.8	8.3	7.8
18	9.5	9.1	8.6	8.2	7.7
19	9.4	8.9	8.5	8.0	7.6
20	9.2	8.7	8.3	7.9	7.4
21	9.0	8.6	8.1	7.7	7.3
22	8.8	8.4	8.0	7.6	7.1
23	8.7	8.3	7.9	7.4	7.0
24	8.5	8.1	7.7	7.3	6.9
25	8.4	8.0	7.6	7.2	6.7
26	8.2	7.8	7.4	7.0	6.6
27	8.1	7.7	7.3	6.9	6.5
28	7.9	7.5	7.1	6.8	6.4
29	7.8	7.4	7.0	6.6	6.3
30	7.6	7.3	6.9	6.5	6.1

° SECTION IV: BIOLOGICAL EVALUATIONS

CHAPTER IV-1
HABITAT SUITABILITY INDICES

Habitat Suitability Index (HSI) models developed by the U.S. Fish and Wildlife Service are used to evaluate habitat quality for a fish species. HSI models can be used independently or in conjunction with the Habitat Evaluation Procedures (HEP) applications described in Chapter II-1.

The HSI models provide a basic understanding of species habitat requirements, and have utility and applicability to use attainability analyses. There are several types of HSI models including pattern recognition, word models, statistical, linear regression, and mechanistic forms in the FWS model publication series. Use of models is predicated on two assumptions: (1) an HSI value has a positive relationship to potential animal numbers: and (2) there is a positive relationship between habitat quality and some measure of carrying capacity. The mechanistic model (Figure 1) sometimes referred to as a structural model is one type that would be useful for use attainability assessments. Information from literature reviews, expert opinion, and study results is integrated in these models to define relationships between variables and habitat suitability. Suitability Index (SI) graphs are developed for each model variable (Figure 2). The variables included in a model represent key habitat features known to affect the growth, survival, abundance, standing crop, and distribution for specific species. The model provides a verbal or mathematical comparison of the habitat being evaluated to the optimum habitat for a particular evaluation species. For some mechanistic models (Figure 3) a mathematical aggregation procedure is used to integrate relationships of model components. In others (Figure 4) an HSI value is defined as the lowest SI value for any variable in the model. Nonmechanistic models (e.g., statistical models for standing crop and harvest) do not require use of SI graphs. Output from an HSI model, regardless of the type, is used to determine the quantity of habitat for a specific species at a site, and an HSI value ranges from 0 to 1, with 1 representing optimum conditions. The relationship:

Habitat area x Habitat quality (HSI) = Habitat Units (HU's)

provides the basis for obtaining habitat data to compare before and after conditions for a site if pollution problems or other environmental problems are solved.

As with all models, some potential sources of subjectivity exist in HSI models. Potential subjectivity in mechanistic models may occur when: (1) determining which variables should be included in the model; (2) developing suitability index graphs from contradictory or incomplete data; (3) incorporating information for similar species of different life stages in the suitability index graphs; (4) determining whether or not highly correlated variable really affect habitat suitability independently and which variables, if any, should be eliminated from the model; (5) determining when, where and how model variables should be measured; and (6) converting assumed relationships between variables into mathematical equations that aggregate suitability indices for individual variables into a species HSI (Terrell et al., 1982). All models developed and published by the U.S. Fish and Wildlife Service are subjected to reviews by species experts to eliminate as much subjectivity as possible.

Appendix A-1 of this manual is a reprint of the HSI developed for the channel catfish. Readers are encouraged to read the appendix to gain greater understanding of features of the model. HSI models for 19 aquatic and estuarine fish species were published in FY 82, and an additional 20 are under development and planned for publication in FY 83. Models have been published for striped bass, channel catfish, creek chub, cutthroat trout, black crappie, white crappie, blue gill, slough darter, common carp, smallmouth buffalo, black bullhead, green sunfish, largemouth bass, northern pike, juvenile spot, juvenile Atlantic croaker, gulf menhaden, brook trout, and the southern kingfish. Models for coastal species were developed at the National Coastal Ecosystems Team (NCET) and those for inland species were developed at the Western Energy and Land Use Team (WELUT).

For more information concerning models for inland species, contact: Team Leader, Western Energy and Land Use Team, 2627 Redwing Road, Fort Collins, Colorado 80526 (FTS 323-5100, or comm. 303-226-9100). Individuals interested in models for coastal species should contact Team Leader, National Coastal Ecosystems Team, 1010 Gause Boulevard, Slidell, Louisiana 70458 (FTS 685-6511, or comm. 504-255-6511).

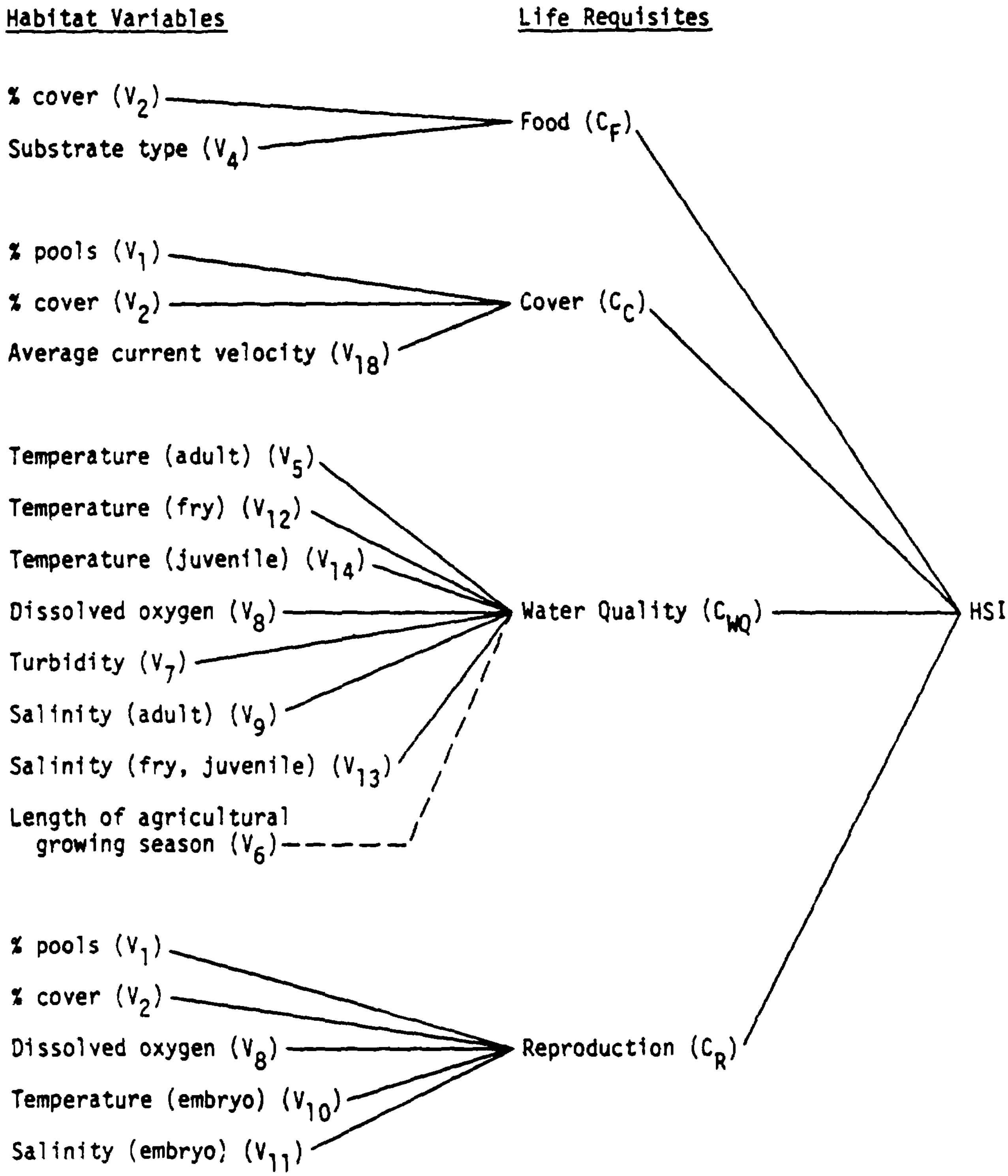
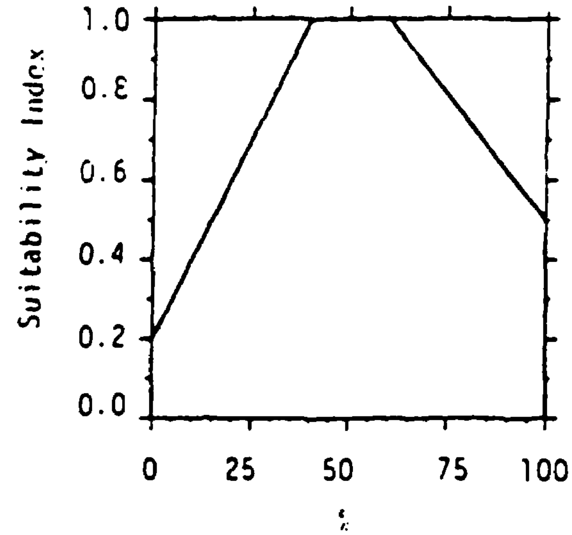


Figure 1. Tree diagram illustrating the relationship of habitat variables and life requisites in the riverine model for the channel catfish HSI model. The dashed line for the length of agricultural growing season (V_6) is for optional use in the model (McMahon and Terrell 1982).

Suitability Graph

Variable

(V_1) Percent pools during average summer flow.



(V_2) Percent cover (logs, boulders, cavities, brush, debris, or standing timber) during summer within pools, backwater areas, and littoral areas.

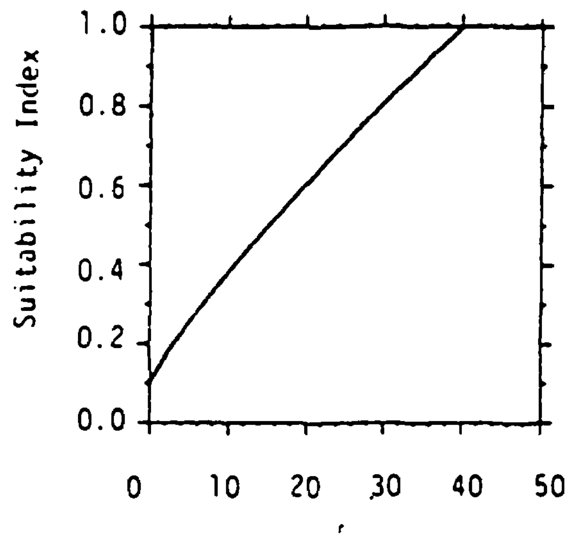


Figure 2. Suitability Index graphs for variables V_1 and V_2 in the channel catfish riverine model. A SI value can range from 0 to 1 with 1 representing an optimum condition (McMahon and Terrell 1982).

Food (C_F).

$$C_F = \frac{V_2 + V_4}{2}$$

Cover (C_C).

$$C_C = (V_1 \times V_2 \times V_{18})^{1/3}$$

Water Quality (C_{WQ}).

$$C_{WQ} = \frac{\frac{2(V_5 + V_{12} + V_{14})}{3} + V_7 \times 2(V_8) + V_9 + V_{13}}{7}$$

If V_5, V_{12}, V_{14}, V_8, V_9, or V_{13} is ≤ 0.4, then C_{WQ} equals the lowest of the following: V_5, V_{12}, V_{14}, V_8, V_9, V_{13}, or the above equation.

Note: If temperature data are unavailable, $2(V_6)$ (length of agricultural growing season) may be substituted for the term

$\frac{2(V_5 + V_{12} + V_{14})}{3}$ in the above equation

Reproduction (C_R).

$$C_R = (V_1 \times V_2^2 \times V_8^2 \times V_{10}^2 \times V_{11})^{1/8}$$

If V_8, V_{10}, or V_{11} is ≤ 0.4, then C_R equals the lowest of the following: V_6, V_{10}, V_{11}, or the above equation.

HSI determination.

$$HSI = (C_F \times C_C \times C_{WQ}^2 \times C_R^2)^{1/6}, \text{ or}$$

If C_{WQ} or C_R is ≤ 0.4, then the HSI equals the lowest of the following: C_{WQ}, C_R, or the above equation.

Figure 3. Formulas for the channel catfish riverine HSI model (McMahon and Terrell 1982).

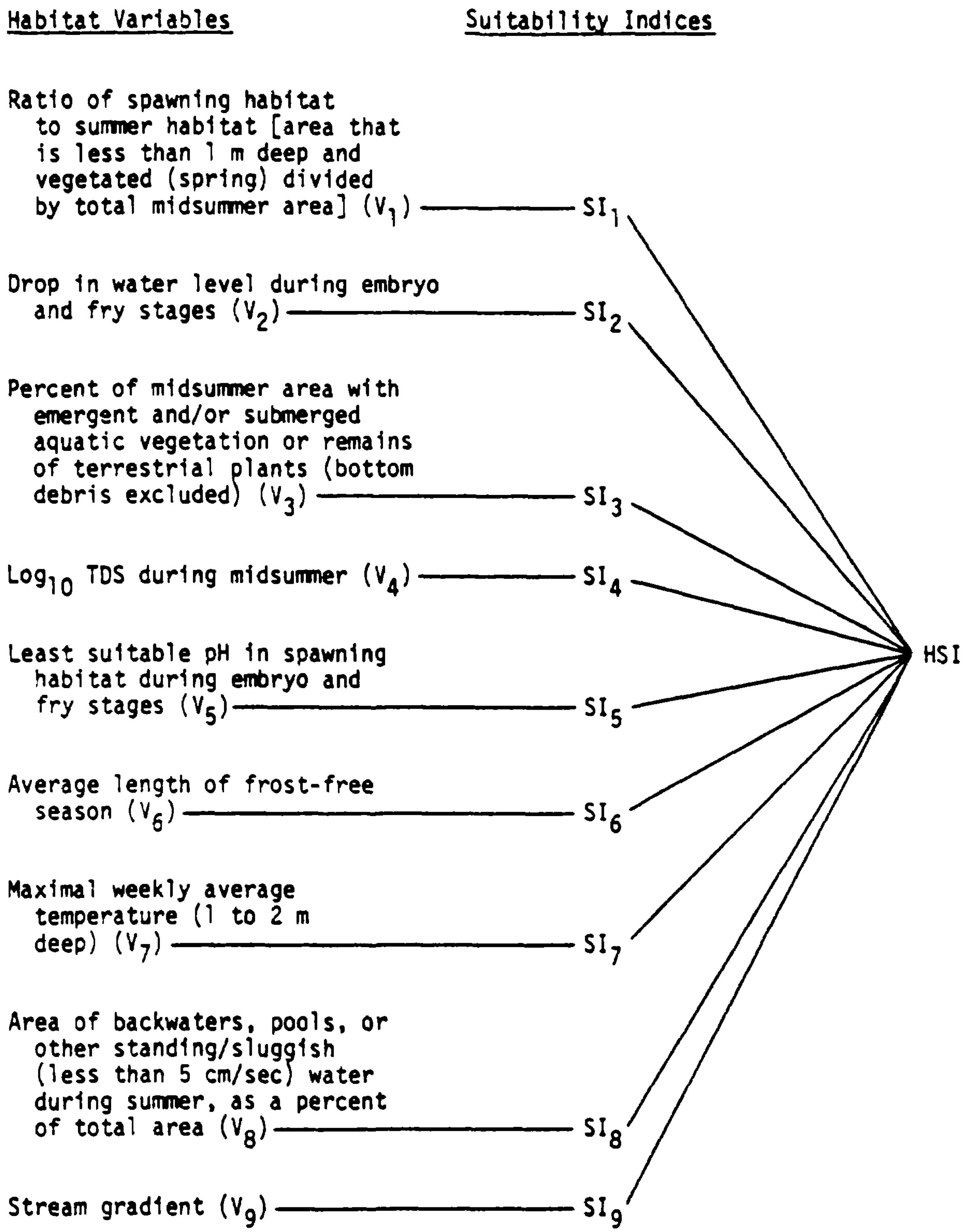


Figure 4. A tree diagram for the northern pike riverine HSI model. Note that habitat variables are not aggregated for separate life requisite components (Inskip 1982).

CHAPTER IV-2
DIVERSITY INDICES AND MEASURES OF COMMUNITY STRUCTURE

Diversity is an attribute of biological community structure. The concepts of richness and composition are commonly associated with diversity. Species richness is simply the number of species, while composition refers to the relative distribution of individuals among the species, or evenness. Odum (1959) defined diversity indices as mathematical expressions which describe the ratio between species and individuals in a biotic community. A major advantage of diversity indices is that they permit the summarization of large amounts of data about the numbers and kinds of organisms into a single numerical description of community structure which is comprehensible and useful to people not immediately familiar with the specific biota. Some diversity indices are expressions of the number of taxa, usually species, in the community. Whittaker (1964) referred to these formulas as indices of "species diversity", i.e. the more species - the greater the diversity. "Dominance diversity indices" (Whittaker, 1964) incorporate the concepts of both richness and evenness; thus, diversity increases as the number of species increases or as the individuals become more evenly distributed between the species.

The response of bottom fauna to four types of pollution is represented in Figure IV-2-1 (Keup 1966). Figure IV-2-1A shows that organic pollutants generally decrease the number of species present while increasing the numbers of surviving taxa, whereas toxic pollutants tend to reduce both numbers and kinds of organisms (Figure IV-2-1B). In general, the effect of all types of pollutant stress on community structure is the loss of diversity. The value of diversity in natural communities lies in the fact that the presence of many species insures the likelihood of "redundancy of function" (Cairns et al. 1973). As explained by Cairns and Dickson (1971), in a highly diverse community, the constantly changing environment will probably affect only a small portion of the complex bottom fauna community at any time. Because there are many different kinds of organisms present, the role of those eliminated as a result of natural environmental change will be filled by other organisms. Thus the food cycle and the system as a whole remain stable. On the other hand, natural environmental variation might eliminate a significant portion of a community that has been simplified by pollutant stress. With no organism available to fill the vacated niche, the functional capacity of the unstable community may be jeopardized. Generally, maintenance of diversity is important because it enhances the stability of a system.

Diversity indices are commonly computed as one tool among many in the analysis of aquatic (as well as terrestrial) communities. Some prevalent reasons for measuring community diversity are listed below (these purposes are by no means independent of each other):

- ° To investigate community structure or functions
- ° To establish its relationship to other community properties such as productivity and stability
- ° To establish its relationship to environmental conditions

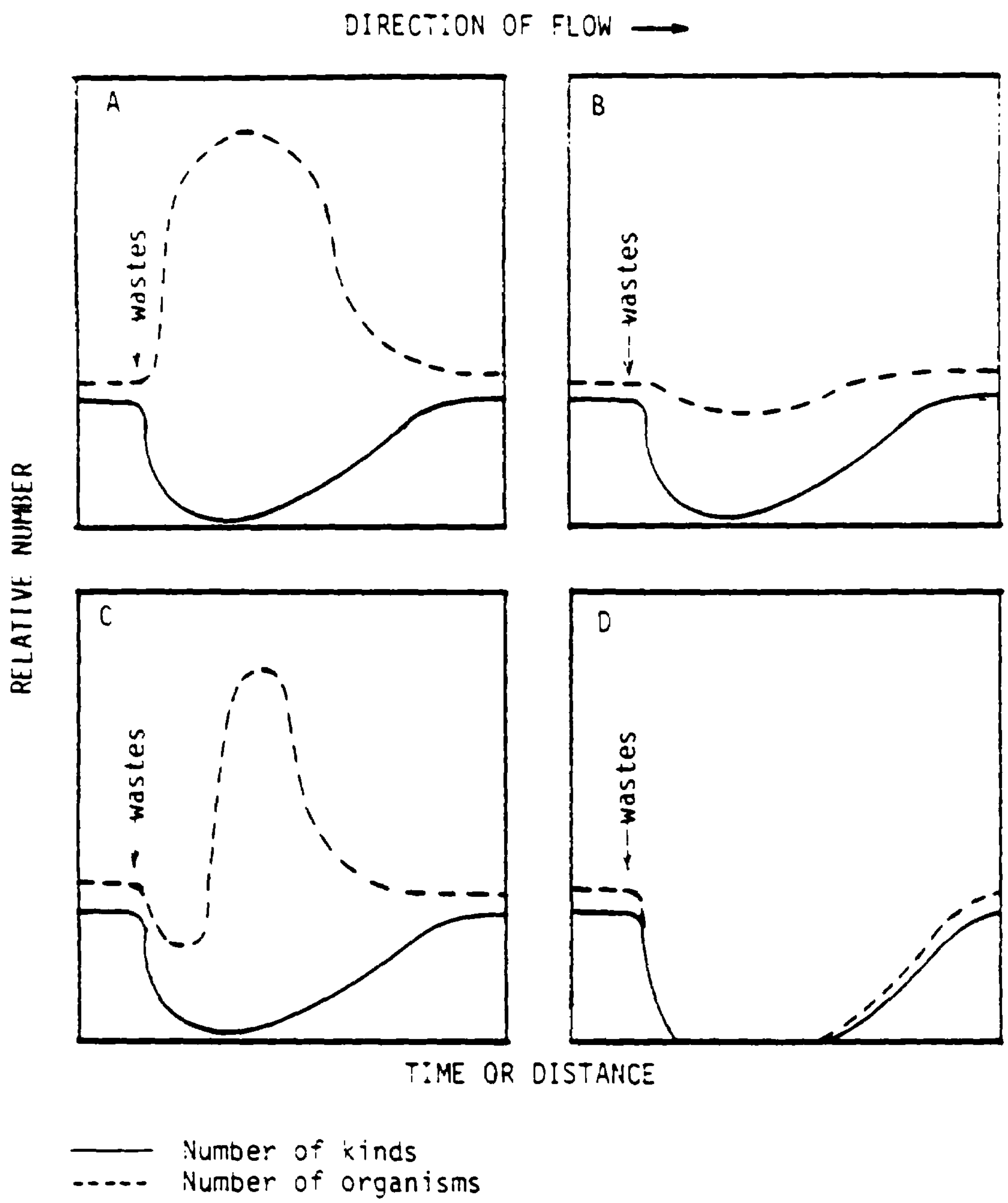


Figure IV-2-1. Response of bottom fauna to pollution: A=organic wastes; B=toxic wastes; C=organic wastes showing temporary toxicity; D=organic wastes mixed with toxic chemicals (from Keup,1966).

- ° To compare communities
- ° To evaluate the biotic health of the community
- ° To assess the effects of pollutant discharges
- ° To monitor water quality by biological rather than physicochemical means

In analyses of freshwater aquatic communities, diversity studies generally involve benthic macroinvertebrates or fish. Several advantages and disadvantages have been given for the study of these groups (Cairns and Dickson 1971, Karr 1981), and are listed in Table IV-2-1. These two groups are generally considered to be the most suitable organisms for evaluation of community integrity. Whereas it might be desirable to investigate the diversity of both fish and macroinvertebrates, the two groups generally are not used in combination to calculate a single diversity index because of differences in sampling selectivity and error.

DIVERSITY INDICES

Many indices of diversity have been developed. Some indices selected from the literature are presented in Table IV-2-2, and the more common ones are discussed below.

Species Diversity Indices

Of the expressions described as species diversity indices (equations 1 through 4 in Table IV-2-2, plus others), the Margalef formula is probably the most popular. Once the sampling and identification is completed, it is an easy matter to calculate the diversity index using the Margalef formula by substituting the number of species(s) and the total number of individuals (n) into the equation below.

$$d = \frac{s-1}{\ln n}$$

The use of this formula, and others of the type, has some important limitations. First, it is not independent of sample size. Menhinick (1964) found that for sample sizes from 64 to 300 individuals the Margalef diversity index varied from 3.05 to 14.74, respectively. In that study, four species diversity indices were evaluated for variation with sample size and all were found unsatisfactory except for the equation referred to as the Menhinick formula in Table IV-2-2. The second limitation of species diversity indices is that, by definition, they do not consider the relative abundance among species, and, therefore, rare species exert a high contribution to the index value. To illustrate this limitation, Wilhm (1972) calculated diversity by the Margalef and Menhinick formulas for three hypothetical communities each containing five species and 100 individuals (see Table IV-2-3). Communities A, B, and C exhibit a wide range of relative distribution of individuals between the five species. Intuitively, community A is more diverse than community C, but the two species diversity indices fail to express any difference.

TABLE IV-2-1. ADVANTAGES AND DISADVANTAGES OF USING MACROINVERTEBRATES AND FISH IN EVALUATION OF THE BIOTIC INTEGRITY OF FRESHWATER AQUATIC COMMUNITIES (CAIRNS AND DICKSON, 1971; KARR, 1981)

MACROINVERTEBRATES

Advantages	Disadvantages
° Fish that are highly valued by humans are dependent on bottom fauna as a food source. ° Many species are extremely sensitive to pollution and respond quickly to it. ° Bottom fauna usually have a complex life cycle of a year or more, and if at any time during their life cycle environmental conditions are outside their tolerance limits, they die. ° Many have an attached or sessile mode of life and are not subject to rapid migrations, therefore they serve as natural monitors of water quality.	° They require specialized taxonomic expertise for identification, which is also time-consuming. ° Background life-history information is lacking for many species and groups. ° Results are difficult to translate into values meaningful to the general public.

FISH

Advantages	Disadvantages
° Life history information is extensive for most species. ° Fish communities generally include a range of species that represent a variety of trophic levels (omnivores, herbivores, insectivores, planktivores, piscivores) and utilize foods of both aquatic and terrestial origin. Their position at the top of the aquatic food web also helps provide an integrated view of the watershed environment. ° Fish are relatively easy to identify. Most samples can be sorted and identified in the field, and then released. ° The general public can relate to statements about conditions of the fish community. ° Both acute toxicity (missing taxa) and stress effects (depressed growth and reproductive success) can be evaluated. Careful examination of recruitment and growth dynamics among years can help pinpoint periods of unusual stress.	° Sampling fish communities is selective in nature. ° Fish are highly mobile. This can cause sampling difficulties and also creates situations of preference and avoidance. Fish also undergo movements on diel and seasonal time scales. ° There is a high requirement for manpower and equipment for field sampling.

TABLE IV-2-2. SUMMARY OF DIVERSITY INDICES

Descriptive Name	Formula	Reference
1. Simplest possible ratio of species per individual	$d = \frac{s}{n}$	Wilhm, 1967
2. Gleason	$d = \frac{s}{\log n}$	Menhinick, 1964, Gleason, 1922
3. Margalef	$d = \frac{s-1}{\ln n}$	Margalef, 1951 1956
4. Menhinick	$d = \frac{s}{(n)^{1/2}}$	Menhinick, 1964
5. McIntosh	$\bar{d} = \frac{n - (\sum n_i^2)^{1/2}}{n - (n)^{1/2}}$	McIntosh, 1967
6. Simpson	$\bar{d} = \frac{\sum n_i (n_i - 1)}{n (n-1)}$	Simpson, 1949
7. Brillouin	$H = (\frac{1}{n}) (\log n! - \sum_{i=1}^{s} \log n_i!)$	Brillouin, 1960
8. Shannon-Wiener	$H = -\sum (p_i \log_2 p_i)$	Shannon and Weaver, 1963; Wiener, 1948
Approximate form of the Shannon Index	$H' = \bar{d} = -\sum (\frac{n_i}{n}) \log_2 (\frac{n_i}{n})$	
Shannon Index using biomass (weight) units	$\bar{d} = -\sum (\frac{w_i}{w}) \log_2 (\frac{w_i}{w})$	Wilhm, 1968

TABLE IV-2-2. (Cont'd)

9.	Hierarchical Diversity Index (HDI)	$HDI = H'(F)+H'_F+H'_{GF}(S)$	Pielou, 1969, 1975
10.	Hierarchical Trophic-Based D.I. (HTDI)	$HTDI = H'(T_1)+H'_{T1}(T_2)+H'_{T1,T2}(T_3)$	Osborne et al., 1980
11.	Redundancy (r)	$r = \frac{\bar{d}_{max} - \bar{d}}{\bar{d}_{max} - \bar{d}_{min}}$	Patten, 1962; Wilhm, 1967
12.	Equitability (e)	$e = \frac{s'}{s}$	Lloyd and Ghelardi, 1964
13.	Evenness (J,J', v)	$J = \frac{H}{H_{max}}$	Pielou 1969, 1975; Hurlbert, 1971
		$J' = \frac{\bar{d}}{\bar{d}_{max}} = \frac{\bar{d}}{\log s}$	
		$v = \frac{\bar{d} - \bar{d}_{min}}{\bar{d}_{max} - \bar{d}_{min}}$	
14.	Number of moves (NM)	$NM = \frac{n\ (s + 1)}{2} - \Sigma R_i n_i$	Fager, 1972
15.	Sequential Comparison Index	$DI_1 = \frac{\text{number of runs}}{\text{number of species}}$ $DI_T = (\overline{DI}_1)(\text{number of taxa})$	Cairns et al. 1968; Cairns & Dickson, 1971; Buikema et al. 1980

TABLE IV-2-2. (Cont'd)

KEY

$H = d \equiv H' = \bar{d}$ = diversity index.

n = total number of individuals.

n_i = number of individuals in species i.

s = total number of species.

p_i = probability of selecting an element of state i $\equiv \frac{n_i}{n}$.

R_i = rank of species i.

s' = the species required to produce the calculated d.i. value if the individuals were distributed among the species according to MacArthur's (1957, 1960) "broken-stick" model.

TABLE IV-2-3. DIVERSITY OF THREE HYPOTHETICAL COMMUNITIES EVALUATED BY THE MARGALEF, MENHINICK, AND SHANNON-WIENER INDICES

Community	n_1	n_2	n_3	n_4	n_5	n	s	$\frac{s-1}{\ln n}$	$\frac{s}{n^{1/2}}$	$\bar{d}$
A	20	20	20	20	20	100	5	0.87	0.50	2.32
B	40	30	15	10	5	100	5	0.87	0.50	1.67
C	1	1	1	1	96	100	5	0.87	0.50	0.12

Another shortcoming of species per individual formulas is that they are not dimensionless, thus substitution of alternate variables for numbers - such as biomass or energy flow - would produce values dependent on the arbitrary choice of units.

The major advantage of using species diversity indices is the simplicity of calculation; however, certain conditions for their proper use must be considered. Since these formulas are dependent on sample size (except possibly, the Menhinick equation), for intercommunity comparison the sample sizes should be as nearly identical as possible. It must be kept in mind that these expressions represent only the number of species and not any expression of relative abundance. Finally, for use of variables other than numbers, the units must be specified and kept consistent.

Dominance Diversity Indices

The most prominent dominance diversity index (equations 5 through 8 in Table IV-2-2, plus others) is the Shannon-Wiener formula. This index is used extensively in research projects, as is the Simpson equation. The Shannon-Weiner diversity index evolved from information theory to the functional equation shown below:

$$\bar{d} = -\sum \left(\frac{n_i}{n}\right)\log_2\left(\frac{n_i}{n}\right)$$

in which the ratio of the number of individuals collected of species i to the total *number of individuals in the sample* (n_i/n) estimates the total population value (N_i/N), which is an approximation of the probability of collecting an individual of species i (p_i). It should be noted that the units of d using $\log_2$ is the binary unit, or bit. Natural logarithms or $\log_{10}$ are sometimes substituted into the equation for convenience, in which case different index values would be obtained, with the units of nats or decits, respectively. The Shannon-Wiener diversity index is calculated using base 10 logarithms, for two simple, hypothetical samples in Example IV-2-1 (see statistical analysis section). A formula for conversion between differently-based logarithms is given below:

$$\log_2 Y = 1.443 \ln Y = 3.323 \log_{10} Y$$

The logarithm base and units should always be given when reporting data.

The dominance and species diversity indices discussed can be used to measure the diversity of virtually any biological community (including macroinvertebrates and fish), and their application is limited only by sampling effectiveness. Wilhm and Dorris (1968) evaluated species diversity of benthic macroinvertebrates using the Shannon-Wiener formula and obtained values less than 1.0 in areas of heavy pollution, values from 1.0 to 3.0 in areas of moderate pollution, and values exceeding 3.0 in clean water areas (values given are in decits).

Disadvantages of using the Shannon index (or others of the type) include the considerable time, expense, and expertise involved in sampling, sorting, and identification of samples. Calculation of the index value can be mathematically tedious if done manually, but is greatly simplified if a computer is available. Computer programs for computing d and r are provided in the literature (Wilhm, 1970; Cairns and Dickson, 1971).

The Shannon-Wiener formula has a number of features which enhance its usefulness. This index of diversity is much more independent of sample size than the species diversity indices (Wilhm 1972). Since it incorporates the concept of dominance diversity, the relative importance of each species collected is expressed and the contribution of rare species to diversity is low. This is illustrated by the d values calculated using the Shannon equation for the three communities in Table IV-2-3. Also, the Shannon formula is dimensionless, facilitating the measurement of biomass diversity. Odum (1959) recognized that the structure of the biomass pyramid held more ecological (trophic) significance than the numbers pyramid because it takes many small individuals to equal the mass of one large individual. The Shannon-Wiener equation can easily be modified to accomodate any units of weight as shown below:

$$\bar{d} = -\sum \left(\frac{w_i}{w}\right)\log_2\left(\frac{w_i}{w}\right)$$

Wilhm (1968) pointed out that use of this diversity index with units of energy flow might be even more valuable to the study of community structure and function.

Hierarchical Diversity

Diversity indices, such as the Shannon-Wiener index, can be partitioned to reflect the contribution made by different taxonomic and trophic levels. Pielou (1975) suggested that a community showing more diversity at higher taxonomic levels (e.g. genus and family) should be considered to be more diverse than a community with the same number of species but congeneric or cofamilial. Osborne et al (1980)questioned the ecological significance of Pielou's suggestion, but investigated the use of the hierarchical diversity index (HDI) shown below:

$$HDI = H'(F) + H'_F(G) + H'_{FG}(S)$$

in which $H'(F)$ is the familial component of the total diversity, $H'_F(G)$ is the generic component of the total diversity, and $H'_{FG}(S)$ is the specific component of the total diversity. The equation used by Kaesler et al. (1978) illustrates the calculation of the hierarchical components. They used

$$H = \alpha H_O + \beta \sum_{i=1}^{o} \frac{N_i}{N} H_{F,i} + \gamma \sum_{i=1}^{o} \sum_{j=1}^{fi} \frac{N_{ij}}{N} H_{G,ij} + \delta \sum_{i=1}^{o} \sum_{j=1}^{fi} \sum_{k=1}^{gij} \frac{N_{ijk}}{N} H_{s,ijk}$$

where α, β, γ, and δ are weighting coefficients; subscripts O, F, G, and S represent order, family, genus, and species, respectively; o, f, and g represent number of orders, families within orders, and genera within families, respectively; N represents the number of individuals; and N_i represents the number of individuals in the ith group. Osborne et al. (1980) concluded that identification to the family level was sufficient to detect intersite differences in that study, while the order level (Hughes, 1978) and generic level (Kaesler et al., 1978) were sufficient in other studies. Determination that identification to species or genus is unnecessary for a particular study would reduce the time, expertise, and expense required. A hierarchical diversity index would be of more ecological value if it were based on trophic relationships rather than taxonomy. Osborne, et al. (1980) presented the following hierarchical trophic diversity index (HTDI):

$$HTDI = H'(T_1) + H'_{T1}(T_2) + H'_{T1T2}(T_3)$$

in which $H'(T_1)$ is the general trophic level component of the total trophic diversity, $H'_{T1}(T_2)$ is the functional group component of the total trophic diversity, and $H'_{T1T2}(T_3)$ is the lowest taxonomic unit component of the total trophic diversity. The classifications used in the hierarchical trophic-based diversity index of Osborne et al. (1980) are listed in Table IV-2-4A. Two classification systems were investigated by Kaesler et al. (1978): the trophic classifications appear in Table IV-2-4B. and the functional morphological classifications are shown in Table IV-2-4C. All of these hierarchical diversity indices used benthic macroinvertebrates as their group of study. Hierarchical diversity indices based on trophic level and functional morphology are relatively new and their utility will improve as more experience is gained. These indices are of potentially great ecological value because of their functional (rather than structural, e.g. taxonomic) approach to community analysis.

Evenness and Redundancy

When using dominance diversity indices, it is desirable to distinguish between the two concepts of diversity incorporated into them, since it is theoretically possible for a community with a few, evenly-represented species to have the same index value as a community with many, unevenly-represented species. For this reasons, relative diversity expressions (equations 11 through 14 in Table IV-2-2, plus others) such as eveness and redundancy are often used in conjunction with dominance diversity indicies. Redundancy is an expression of the dominance of one or more species and is inversely proportional to the wealth of species (Wilhm and Dorris, 1968). To use the redundancy expression in conjunction with the Shannon-Wiener index, the theoretical maximum diversity (d_{max}) and minimum diversity (d_{min}) are calculated by the equations:

$$\bar{d}_{max} = \left(\frac{1}{n}\right) [\log_2 n! - s \log_2 (n/s)!]$$

TABLE IV-2-4. FUNCTIONALLY-BASED HIERARCHICAL CLASSIFICATION SYSTEMS

A. Hierarchical trophic classification used for HTDI calculations

H_{T1} (Trophic level)	H_{T2} (Functional group)	H_{T3} (Number of individuals)
Omnivore	Filter Feeders	Number of individuals of each taxon within each functional group.
	Collector-Gatherer-Shredder-Engulfer	
	Engulfer-Shredder	
	Collector-Filterer-Engulfer	
	Engulfer-Grazer	
	Engulfer-Collector-Grazer	
Carnivore	Engulfer	
	Piercer	
Herbivore	Scraper-Collector-Gatherer	
	Collector-Gatherer-Shredder	
	Collector-Filterer-Gatherer	
	Collector-Gatherer	
	Collector-Filterer	
	Shredder	
Detritivore	Shredder	
	Collector-Gatherer	

B. Trophic classification of macrobenthic invertebrates. For any specific application, not all possible combinations are likely to be realized.

Level of Hierarchy	Name	Subdivisions
I	Functional group	shredders (vascular plant tissues)
		collectors (detrital materials)
		grazers (Aufwuchs)
		predators
		parasites
II	Feeding mechanism	chewers and miners
		filters (suspension feeders)
		gatherers (sediment or deposit feeders)
		scrapers
		chewers and suckers
		swallowers and chewers
		piercers
		attachers
III	Dependence	obligate
		facultative
IV	Food habit	herbivory
		detritivory
		carnivory
		omnivory
V	Species	number of individuals

TABLE IV-2-4 . FUNCTIONALLY-BASED HIERARCHICAL CLASSIFICATION SYSTEMS (Cont'd)

C. HBR (head, body, respiratory organ) classification of macrobenthic invertebrates according to functional morphology: head position, body shape, and respiratory organs.

Level of Hierarchy	Name	Subdivisions
I	Head position (feeding category)	hypognathous
		prognathous
		opisthorhynchous
		vestigial or other
II	Body shape (current of stream)	flattened irregular
		flattened oval
		flattened elongate
		compressed laterally
		cylindrical
		elongate
		short, compact
		fusiform
		irregular
		hemicylindrical or subtriangular
III	Respiratory organs (substratum)	simple filamentous gills
		compound filamentous gills
		platelike gills
		operculate gills
		leaflike gills or organs
		respiratory dish
		respiratory tube
		spiracular gills
		caudal chamber
		plastron
		body integument
		tracheal respiration
IV	Species	number of individuals

$$\bar{d}_{min} = \left(\frac{1}{n}\right) \{\log_2 n! - \log_2 [n - (s-1)!]\}$$

Then the location of $\bar{d}$ between the theoretical extremes can be computed by the redundancy formula:

$$r = \frac{\bar{d}_{max} - \bar{d}}{\bar{d}_{max} - \bar{d}_{min}}$$

Table IV-2-5 illustrates the expression of redundancy.

TABLE IV-2-5. THE SHANNON-WIENER INDEX AND CORRESPONDING REDUNDANCY VALUES FOR 11 HYPOTHETICAL COMMUNITIES. (after Patten, 1962).

	Communities (N = 6)										
Species	A	B	C	D	E	F	G	H	I	J	K
S_1	1	2	2	3	2	3	4	3	4	5	6
S_2	1	1	2	1	2	2	1	3	2	1	-
S_3	1	1	1	1	2	1	1	-	-	-	-
S_4	1	1	1	1	-	-	-	-	-	-	-
S_5	1	1	-	-	-	-	-	-	-	-	-
S_6	1	-	-	-	-	-	-	-	-	-	-
$\bar{d}$(bits)	2.58	2.25	1.93	1.79	1.61	1.47	1.25	1.00	0.92	0.65	0.00
R	0.00	0.13	0.25	0.30	0.38	0.43	0.52	0.61	0.64	0.75	1.00

Expressions have also been developed to describe the evenness of apportionment of individuals among species in a community. Evenness measures have historically taken two forms. One is the ratio of diversity to the maximum possible diversity, where d_{max} is defined as the community in which all species are equally distributed:

$$J' = \bar{d}/\bar{d}_{max} \cong \bar{d}/\log s$$

Where the logarithm is to the same base as used in the corresponding diversity index calculation. However, log s is only an approximation of d_{max} because all species in the community generally will not be sampled. A measure of evenness that does not depend on s is shown below:

$$V = \frac{\bar{d} - \bar{d}_{min}}{\bar{d}_{max} - \bar{d}_{min}}$$

It was from this measure of evenness that the expression for redundancy (shown above) was derived by the relationship $r = 1-V$; thus, redundancy may also be thought of as a measure of the unevenness of apportionment of individuals among species.

Sequential Comparison Index

The sequential comparison index (SCI) is probably the most widely used index of diversity because of its extensive worldwide use in industrial (non-academic) studies. The SCI is a simplified, rapid method for estimating relative differences in biological diversity and has been used mainly for assessing the biological consequences of pollution. Use of the SCI requires no taxonomic expertise on the part of the investigator. Although it has been used with microorganisms, the SCI is predominately used to evaluate diversity in benthic macroinvertebrate communities. The collected specimens are randomly poured into a white enamel pan with parallel lines drawn on the bottom. Only two specimens are compared at a time. Comparisons are based on differences in shape, color, and size of the organisms. If the imminent specimen is apparently the same as the previous one, it is part of the same "run"; if it is not, it is part of a new run. An easy way of recording runs is to use a series of X's and O's. For example, the specimens shown in line one of Figure IV-2-2 would be recorded, from left to right as X 0 X 0 X 0 X, or seven runs. The specimens in line two would be tabulated by X X X 0 X X X. Sample two only contains three runs and is obviously less diverse. Ultimately, it will be necessary to know the total number of taxa in the collection. This can either be counted after determining the number of runs or determined simultaneously by underlining the symbol of each new taxon as shown above.

Cairns, et al. (1971) described the following stepwise procedure for calculating the Sequential Comparison Diversity Index:

1. Gently randomize specimens in a jar by swirling.
2. Pour specimens out on a lined white enamel pan.
3. Disperse clumps of specimens by pouring preservative or water on clumps.
4. If the sample has fewer than 250 specimens, determine the number of runs for entire sample and go to Step 12.
5. If sample has more than 250 specimens, determine the number of runs for the first 50 specimens.
6. Calculate DI_1 where DI_1 = numbers of runs/50.
7. Plot DI_1 against the number of specimens examined as in Figure IV-2-3.
8. Calculate the SCI for the next 50 specimens.
9. Determine the total number of runs for the 100 specimens examined.
10. Calculate a new DI_1 for 100 specimens as in Step 6 and plot the value obtained on the graph made in Step 7, where DI_1 = number of runs/100.
11. Repeat this procedure in increments of 50 until the curve obtained becomes asymptotic. At this point enough specimens have been examined so that continued work will produce an insignificant change in the final DI_1 value.
12. Calculate final DI_1 where

$$DI_1 = \frac{\text{number of runs}}{\text{number of specimens}}$$

13. Record the number of different taxa observed in the entire sample. This can be done after deriving the final DI_1 or simultaneously by simply noting each new taxon as it is examined in the determination of runs.

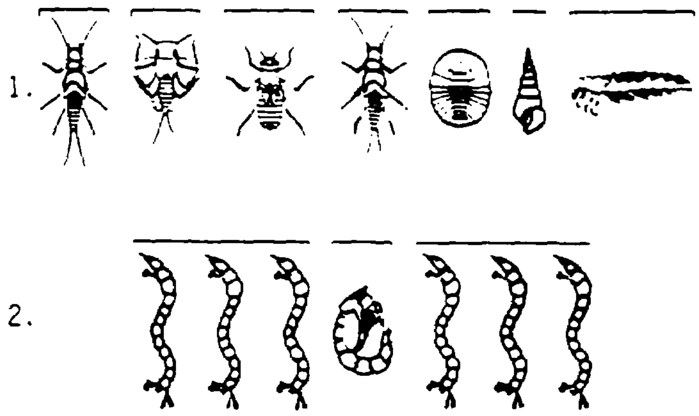


Figure IV-2-2. Determination of runs in SCI technique (from Cairns and Dickson, 1971).

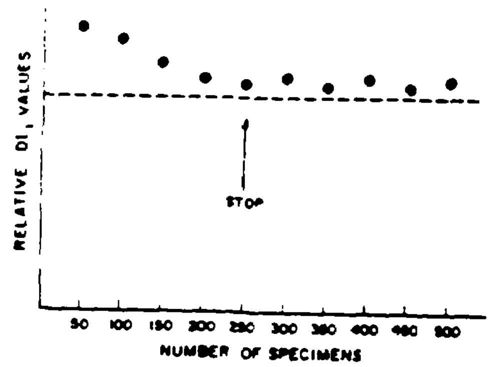


Figure IV-2-3. DI_1 and sample size (from Cairns and Dickson, 1971).

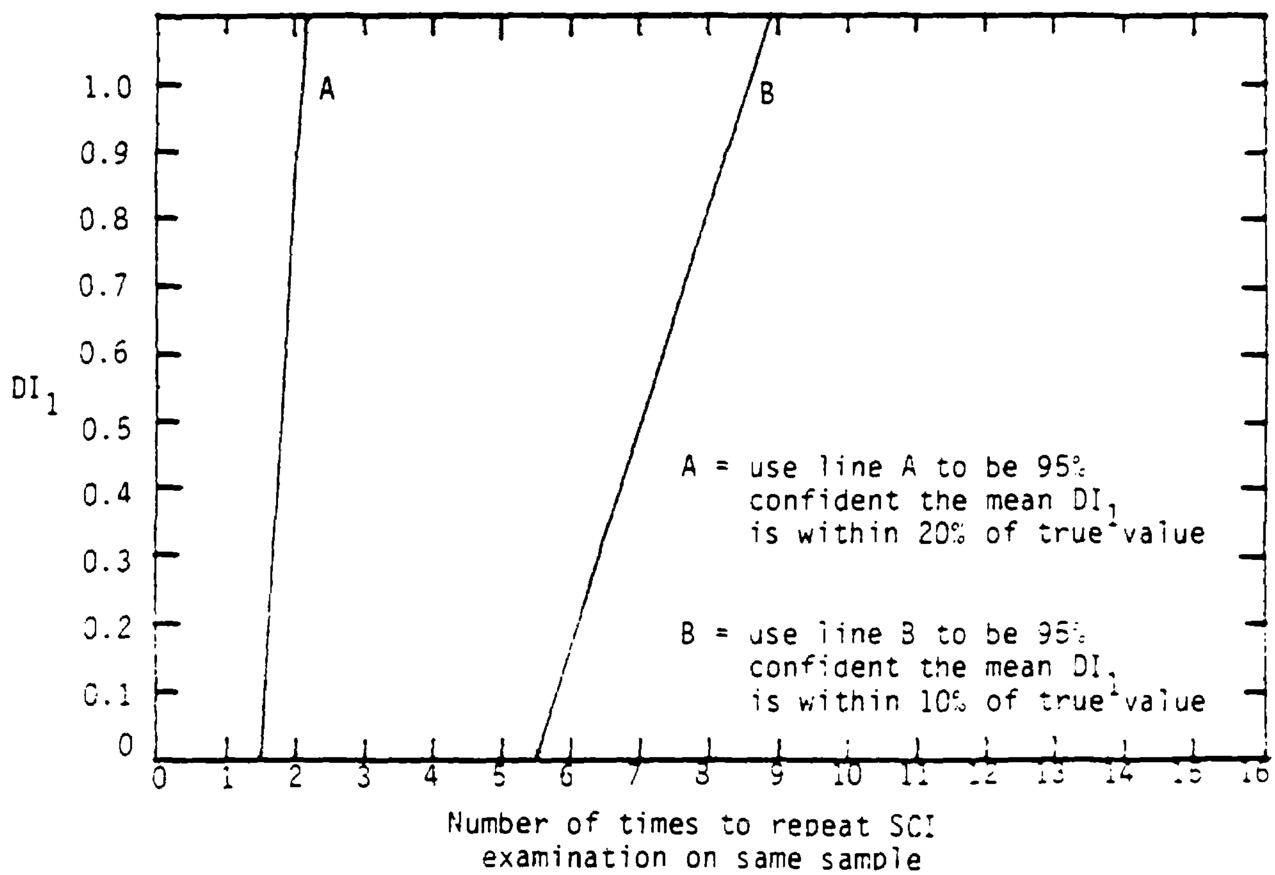


Figure IV-2-4. Confidence limits for DI_1 values (from Cairns and Dickson, 1971).

14. Determine from Figure IV-2-4 the number of times the SCI examination must be repeated on the same sample to be 95 percent confident that the mean DI_1 is within a chosen percentage of the true value for DI_1. In most pollution work involving gross differences between sampling areas, Line A of Figure IV-2-4 should be used. For example, suppose DI_1 were 0.60. Using Line A of Figure IV-2-4 the SCI should be performed twice to be 95 percent confident that the mean DI_1 is within 20 percent of the true value.
15. After determining N, rerandomize the sample and repeat the SCI examination on the same number of specimens as determined in Step 11. Repeat this procedure N - 1 times.
16. Calculate DI_1 by the following equation:

$$DI_T = \overline{DI}_1 \times \text{(number of taxa)}$$

17. Calculate DI_T by the following equation:

$$DI_T = (\overline{DI}_1) \times \text{(number of taxa)}$$

18. Repeat the above procedure for each bottom fauna collection.
19. After determining the DI_T for each bottom fauna collection at each sampling station, there is a simple technique for determining if the community structures of the bottom fauna as evaluated by the SCI (DI_T) value are significantly different within a station or between stations. Calculate the 95 percent confidence intervals around each DI_T value. If the 95 percent confidence intervals do not overlap, then the community structures of the bottom fauna as reflected by the DI_T values are significantly different. For example, suppose the DI_T value for Station 1 were 45 and for Station 2 were 28. In the determination of DI_T a decision was made to use Line A in Figure IV-2-4, which means that the DI_T is within 20 percent of the true value 95 times out of 100. Therefore the 95 percent confidence interval for the DI_T value at Station 1 would be from 49.5 to 40.5, or 10 percent of the DI_T value on either side of the determined DI_T. Station 2 would have a 95 percent confidence interval for the DI_T value of from 30.8 to 25.2. The bottom fauna communities at the two stations as evaluatd by the DI_T index are significantly different.

The SCI permits rapid evaluation of the diversity of benthic macroinvertebrates. Some insight into the integrity of the bottom community can be gained from DI_T values. Cairns and Dickson (1971) reported that healthy streams with high diversity and a balanced density seem to have DI_T values above 12.0, while polluted communities with skewed population structures have given values for DI_T of 8.0 or less, and intermediate values have been found in semipolluted situations.

SPECIAL INDICES

Several expressions that are not diversity indices *per se* but which incorporate the concept of diversity have been formulated. These include numerous biotic indices (Pantle and Buck, 1955; Beck, 1955; Beak, 1964; Chutter 1971, Howmiller and Scott 1977, Hilsenhoff 1977, Winget and Mangum 1979), a composite index of "well-being" (Gammon 1976), and Karr's index (Karr 1981). These indices are designed to evaluate the biotic integrity, or health, of biological communities and ecosystems.

Biotic Indices

Beck (1955) developed a biotic index for evaluating the health of streams using aquatic macroinvertebrates. In the equation

$$\text{Biotic index} = 2(n\ \text{Class I}) + (n\ \text{Class II})$$

where n represents the number of macroinvertebrate species, more weight is assigned to Class I organisms (those tolerant of little organic pollution) than to Class II organisms (those tolerant of moderate organic pollution but not of anaerobic conditions). A stream nearing septic conditions will have a biotic index value of zero; whereas streams receiving moderate amounts of organic wastes will have values from 1 to 6, and streams receiving little or no waste will have values usually over 10 (Gaufin 1973).

The biotic index proposed by Hilsenhoff uses the arthropod community (specifically insects, amphipods, and isopods) to evaluate the integrity of aquatic ecosystems via the formula:

$$BI = \sum n_i a_i / n$$

where n_i is the total number of individuals of the ith species (or genus), a_i is the tolerance value assigned to that species (or genus), and n is the total number of individuals in the sample (Hilsenhoff, 1977; Hilsenhoff, 1982). Pollution tolerance values of zero to five are assigned to species (or genera when species cannot be identified) on the basis of previous field studies. A zero value is assigned to species found only in unaltered streams of very high water quality, a value of 5 is assigned to species known to occur in severely polluted or disturbed streams, and intermediate values are assigned to species occurring in intermediate situations. Calculation of this and other biotic indices are methods of biologically assessing water quality.

Index of Well-Being

Utilizing fish communities, Gammon developed a composite index of well-being (I_{WB}) as a tool for measuring the effect of various human activities on aquatic communities (Gammon, 1976; Gammon and Reidy, 1981; Gammon et al., 1981). This index was calculated by:

$$I_{WB} = 0.5 \ln n + 0.5 \ln w + \bar{d}_{no} + \bar{d}_{wt}$$

in which n is the number of individuals captured per kilometer, w is the weight in kilograms captured per km, $\bar{d}_{no}$ is the Shannon index based on numbers, and $\bar{d}_{wt}$ is the Shannon index based on weights. (The Shannon index was calculated using natural logarithms).

Karr's Index of Biotic Integrity (IBI)

Karr (1981) presented a procedure for classifying water resources by evaluating their biotic integrity using fish communities. Use of the system involves three assumptions: (1) the fish sample is a balanced representation of the fish community at the sample site; (2) the sample site is representative of the larger geographic area of interest; and (3) the scientist charged with data analysis and the final classification is a trained, competent biologist with considerable familiarity with the local fish fauna. For each of the twelve criteria listed in Table IV-2-6, the evaluator subjectively assigns a minus (-), zero (0), or plus (+) value to the sample. The grades are assigned numerical values - (-)=1, (0)=3, (+)=5 - which are summed over all twelve criteria to produce an index of community quality. The sampled community is then placed in one of the biotic integrity classes described in Table IV-2-7 based on numerical boundaries such as those tentatively suggested by Karr (1981) and shown in Table IV-2-8.

TABLE IV-2-6. PARAMETERS USED IN ASSESSMENT OF FISH COMMUNITIES. (SEE ARTICLE TEXT FOR DISCUSSION.)

Species Composition and Richness
- Number of Species
- Presence of Intolerant Species
- Species Richness and Composition of Darters
- Species Richness and Composition of Suckers
- Species Richness and Composition of Sunfish (except Green Sunfish)
- Proportion of Green Sunfish
- Proportion on Hybrid Individuals

Ecological Factors
- Number of Individuals in Sample
- Proportion of Omnivores (Individuals)
- Proportion of Insectivorous Cyprinids
- Proportion of Top Carnivores
- Proportion with Disease, Tumors, Fin Damage, and Other Anomalies

BIOLOGICAL POLLUTION SURVEY DESIGN

The first step in planning any survey of water quality is to identify specific objectives and clearly define what information is sought. For instance, the objective of a use attainability analysis might be to evaluate the water quality or degree of degradation of a body of water, in general, in order to ascertain the accuracy of the current use designation. Alternately, the analysis objective might be to determine the extent of damage caused by a discharge or series of discharges. From such information, the potential attainable use can be identified; judgments must then be made regarding the benefits/costs of improving the degree of waste treatment.

TABLE IV-2-7: BIOTIC INTEGRITY CLASSES USED IN ASSESSMENT OF FISH COMMUNITIES ALONG WITH GENERAL DESCRIPTIONS OF THEIR ATTRIBUTES

Class	Attributes
Excellent	Comparable to the best situations without influence of man; all regionally expected species for the habitat and stream size, including the most intolerant forms, are present with full array of age and sex classes; balanced trophic structure.
Good	Species richness somewhat below expectation especially due to loss of most intolerant forms; some species with less than optimal abundances or size distribution; trophic structure shows some signs of stress.
Fair	Signs of additional deterioration include fewer intolerant forms, more skewed trophic structure (e.g., increasing frequency of omnivores); older age classes of top predators may be rare.
Poor	Dominated by omnivores, pollution-tolerant forms, and habitat generalists; few top carnivores; growth rates and condition factors commonly depressed; hybrids and diseased fish often present.
Very Poor	Few fish present, mostly introduced or very tolerant forms; hybrids common; disease, parasites, fin damage, and other anomalies regular.
No Fish	Repetitive sampling fails to turn up any fish.

TABLE IV-2-8: TENTATIVE RANGES FOR THE BIOTIC INTEGRITY CLASSES.

Class	Index Number
Excellent (E)	57-60
E-G	53-56
Good (G)	48-52
G-F	45-47
Fair (F)	39-44
F-P	36-38
Poor (P)	28-35
P-VP	24-27
Very Poor (VP)	$\leq$ 23

The next steps in planning the survey are to review all available reports and records concerning the waste effluents and receiving waters, and to make a field reconnaissance of the waterway, noting all sources of pollution, tributaries, and uses made of the water.

Sampling Stations

There is no set number of sampling stations that will be sufficient to monitor all types of waste discharges; however, some basic rules for a sound survey design are listed below (Cairns and Dickson 1971). The following describes an "upstream-downstream" study. The reader should also consult Section IV-6 on the reference reach approach to see an alternative method.

1. Always have a reference station or stations above all possible discharge points. Because the usual purpose of a survey is to determine the damage that pollution causes to aquatic life, there must be some basis for comparison between areas above and below the point or points of discharge. In practice, it is usually advisable to have at least two reference stations. One should be well upstream from the discharge and one directly above the effluent discharge, but out of any possible influence from the discharge.

2. Have a station directly below each discharge.

3. If the discharge does not completely mix on entering the waterway but channels on one side, stations must be subdivided into left-bank, midchannel, and right-bank substations. All data collected - biological, chemical, and physical - should be kept separate by substations.

4. Have stations at various distances downstream from the last discharge to determine the linear extent of damage to the river.

5. All sampling stations must be ecologically similar before the bottom fauna communities found at each station can be compared. For example, the stations should be similar with respect to bottom substrate (sand, gravel, rock, or mud), depth, presence of riffles and pools, stream width, flow velocity, and bank cover.

6. Biological sampling stations should be located close to those sampling stations selected for chemical and physical analyses to assure the correlation of findings.

7. Sampling stations for bottom fauna organisms should be located in an area of the stream that is not influenced by atypical habitats, such as those created by road bridges.

8. In order to make comparisons among sampling stations, it is essential that all stations be sampled approximately at the same time. Not more than 2 weeks should elapse between sampling at the first and last stations.

For a long-term biological monitoring program, bottom organisms should be collected at each station at least once during each of the annual seasons. More frequent sampling may be necessary if water quality of any discharge changes or if spills occur. The most critical period for bottom fauna organisms is usually during periods of high temperature and low flow of the waterway. Therefore, if time and funds available limit the sampling frequency, then at least one survey during this time will produce useful information.

Sampling Equipment

Commonly used devices for sampling benthic macroinvertebrate communities include the Peterson dredge, the surber square foot sampler, aquatic bottom nets, and artificial substrate samplers. Proper use of the first three pieces of equipment requires that the operator exert the same amount of effort at each station before comparisons can be made. This subjectivity can cause error, but can be minimized by an experienced operator. Artificial substrates standardize sampling to some extent by providing the same type of habitat for colonization when placed in ecologically similar conditions. A simple type of artificial substrate sampler is a wire basket containing rocks and debris. Others consist of masonite plates or plastic webs which can be floated or submerged. Additional advantages of artificial substrate samplers are quickness and ease of use.

Fish sampling equipment includes electrofishing gear, encircling gear (haul seine, purse seine), towed nets (otter trawl), gill nets, maze gear, and chemical toxicants (rotenone, antimycin). As discussed above, the same sampling effort must be put forth at each station when using this equipment. Also, measures should be taken to reduce the selectivity of fish sampling.

Number of Samples

If comparisons are to be made between stations in a pollution survey, each station must be sampled equally. Either an equal number of samples must be taken at each station or an equal amount of time and effort must be expended.

Organisms are not randomly distributed in nature, but tend to occur in clusters. Because of this, it is necessary to take replicate samples in order to obtain a composite sample that is representative of that station.

There is no "cookbook recipe" which defines the number of samples to take in a given situation. Cairns and Dickson (1971) have found practical experience to show that not less than three artificial substrate samplers, 3 to 10 dredge hauls, and at least three Surber square foot samples represent the minimum number of samples required to describe the bottom fauna of a particular station. Naturally, increasing the number of replicate samples increases the reliability of the data. The data of replicate samples taken at a given station are combined to form a pooled sample. It has been found that a plot of the pooled diversity index versus cumulative sample units becomes asymptotic, and that once this asymptotic diversity index value is found, little is gained by additional sampling. Ideally, a base line study would be conducted to determine the optimum number of samples for a pollution survey.

STATISTICAL ANALYSES

This section describes some of the statistical methods of comparing the diversity indices calculated for different sampling stations.

Hutcheson's t-test

Hutcheson (1970) proposed a t-test for testing for difference between two diversity indices:

$$t = \frac{H_1 - H_2}{S_{H_1 - H_2}}$$

Where H_1-H_2 is simply the difference between the two diversity indices, and

$$S_{H_1 - H_2} = (S^2_{H_1} - S^2_{H_2})^{1/2}$$

The variance of H may be approximated by:

$$S^2_H = \frac{\sum f_i \log^2 f_i - (\sum f_i \log f_i)^2/n}{n^2}$$

Where f_i is the frequency of occurrence of species i and n is the total number of individuals in the sample. The degrees of freedom (df) associated with the preceding t are approximated by:

$$df = (S^2_{H_1} + S^2_{H_2})^2 / \left[\frac{(S^2_{H_1})^2}{n_1} + \frac{(S^2_{H_2})^2}{n_2}\right]$$

Convenient tables of $f_i \log^2 f_i$ are provided by Lloyd, et al. (1968), and t-distribution tables can be found in any statistics textbook (such as Dixon and Massey, 1969; Zar, 1974; etc.). Example IV-2-1 demonstrates the calculation of the Shannon-Wiener index (H) for two sets of hypothetical sampling station data, and then tests for significant difference between them using Hutcheson's t-test.

Example IV-2-1. Comparing Two Indices of Diversity (adapted from Zar 1974).

H_0: The diversity index of station 1 is the same as the diversity index of station 2.

H_A: The diversity indices of stations 1 and 2 are not the same. The level of significance (α) = 0.05

Station 1

Species	number of individuals(n_i)	percentage(f_i)	$f_i \log f_i$	$f_i \log^2 f_i$	$\frac{n_i}{n} \log \frac{n_i}{n}$
1	47	47	78.5886	131.4078	-0.1541
2	35	35	54.0424	83.4452	-0.1596
3	7	7	5.9157	4.9994	-0.0808
4	5	5	3.4949	2.4429	-0.0651
5	3	3	1.4314	0.6830	-0.0457
6	3	3	1.4314	0.6830	-0.0457
6	100	100	144.9044	223.6613	-0.5510

Station 2

Species	number of individuals(n_i)	percentage(f_i)	$f_i \log f_i$	$f_i \log^2 f_i$	$\frac{n_i}{n} \log \frac{n_i}{n}$
1	48	48	80.6996	135.6755	-0.1530
2	23	23	31.3197	42.6489	-0.1468
3	11	11	11.4553	11.9294	-0.1054
4	13	13	14.4813	16.1313	-0.1152
5	3	3	1.4314	0.6830	-0.0457
6	2	2	0.6021	0.1813	-0.0340
6	100	100	139.9894	207.2494	-0.6001

$H_1 = 0.5510$ $\qquad$ $H_2 = 0.6001$

$S^2_{H_1} = 0.00136884$ $\qquad$ $S^2_{H_2} = 0.00112791$

$S_{H_1 - H_2} = 0.0499$

$t = -0.98$

$df = 198.2 = 200$

From a t-distribution table: $t_{0.05(2),200} = 1.972$

Therefore, since the t value is not as great as the critical value for the 95 percent level of significance ($\alpha = 0.05$), the null hypothesis (H_0) is not rejected.

Analysis of Variance

Analysis of variance (ANOVA) can be used to test the null hypothesis that all means are equal, e.g. $H_0: u_1 = u_2 = \ldots = u_k$, where k is the number of experimental groups. "Single factor" or "one-way" ANOVA is used to test the effect of one factor (sampling site) on the variable in question (diversity) in Example IV-2-2. Two-way ANOVA can be used for comparison of spacial and temporal data.

In Example IV-2-2, each datum (X_{ij}) represents a diversity index that has been calculated for j replicate samples at each of i stations. Also, $\bar{x}_i$ represents the mean of station i, n_i represents the number of replicates in sample i, and $N(=\sum n_i)$ represents the total number of indices calculated in the survey.

After computing the mathematical summations, the ANOVA results are typically summarized in a table as shown. The equality of means is determined by the F test.

$$F_{\alpha,\ \text{groups df},\ \text{error df}} = \frac{\text{group MS}}{\text{error MS}}$$

The critical value for this test is obtained from an F-distribution table based on the degrees of freedom of both the numerator and denominator. Since the computed F is at least as large as the critical value, H_0 is rejected, e.g. the diversity index means at all stations are not equal.

Example IV-2-2. A Single Factor Analysis of Variance (adapted from Zar 1974).

H_0: $u_1 = u_2 = u_3 = u_4 = u_5$

H_A: The mean diversity indices of the five stations are not the same

$\alpha = 0.05$

Station 1	Station 2	Station 3	Station 4	Station 5
2.82	3.96	4.10	4.63	5.63
3.32	4.08	4.41	4.21	5.41
3.64	3.79	4.64	4.35	5.94
3.46	3.71	4.02	4.88	6.27
2.91	4.36	3.86	4.37	6.00
3.10	4.24	3.63	4.01	5.73

Station	1	2	3	4	5
$\bar{x}_i$	3.21	4.02	4.11	4.41	5.83
n_i	6	6	6	6	6
$\sum_{j=1}^{n_i} x_{ij}$	19.25	24.14	24.67	26.45	34.98
$\left[\sum_{j=1}^{n_i} x_{ij}\right]^2 / n_i$	61.76	97.12	101.43	116.60	203.93

$$\sum_{i=1}^{k} \left[\sum_{j=1}^{n_i} x_{ij}\right]^2 / n_i = 580.84 \qquad \sum_i \sum_j x_{ij}^2 = 583.21$$

$$\sum_i \sum_j x_{ij} = 129.49 \qquad C = \frac{\left(\sum_i \sum_j x_{ij}\right)^2}{N} = 558.92$$

$$\text{total sum of squares} = \sum_i \sum_j x_{ij}^2 - C = 24.29$$

$$\text{groups sum of squares} = \sum_{i=j}^{k}\left(\sum_{j=1}^{n_i} x_{ij}\right)^2 / n_i - C = 21.92$$

error sum of squares = total ss - groups ss = 2.37
total degrees of freedom = N - 1 = 29
groups degrees of freedom = k - 1 = 4
error degrees of freedom = total df - groups df = 25
mean squared deviations from the mean (MS) = ss/df
groups MS = 21.92/4 = 5.48 error MS = 2.37/25 = 0.09

Summary of the Analysis of Variance

Source of Variation	SS	df	MS
total	24.29	29	
groups	21.92	4	5.480
error	2.37	25	0.095

$$F = \frac{\text{groups MS}}{\text{error MS}} = \frac{5.480}{0.095} = 57.68$$

$$F_{0.05(1),4,25} = 2.76$$

Therefore, Reject $H_0 : u_1=u_2=u_3=u_4=u_5$

Multiple Range Testing

The single factor analysis of variance tests whether or not all of the mean diversity indices are the same, but gives no insight into the location of the differences among stations. To determine between which stations the equalities or inequalities lie, one must resort to multiple comparison tests (also known as multiple range tests). The most commonly used methods are the Student-Newman-Keuls test (Newman 1939, Keuls 1952) and the Duncan's test (Duncan 1955).

Student-Newman-Keuls Test

Example IV-2-3 demonstrates the Student-Newman-Keuls (SNK) procedure for the data presented in Example 2. Since the ANOVA in Example IV-2-2 rejected the null hypothesis that all means are equal, the SNK test may be applied. First, the diversity index means are ranked in increasing order.

Then, pairwise differences ($\bar{x}_B-\bar{x}_A$) are tabulated as shown in Example IV-2-2 . The value of p is determined by the number of means in the range of means being tested. Using the p value and the error degrees of freedom from the ANOVA, "studentized ranges," abbreviated q $q_{\alpha,df,p}$ are obtained from a table of q-distribution critical values. The standard error is calculated by:

$$SE = (S^2/n)^{1/2} = (\text{error MS}/n)^{1/2}$$

If the k group sizes are not equal, a slight modification is necessary. For each comparison involving unequal n, the standard error is approximated by:

$$SE = \left[\frac{S^2}{2}\left(\frac{1}{n_A} + \frac{1}{n_B}\right)\right]^{1/2}$$

Example IV-2-3. Student-Newman-Keuls Multiple Range Test with Equal Sample Sizes. This example utilizes the raw data and analysis of variance presented in Example IV-2-2.

Ranks of sample means (i)	1	2	3	4	5
Ranked sample means ($\bar{x}_i$)	3.21	4.02	4.11	4.41	5.83

$$SE = (\text{error MS}/n)^{1/2} = (0.095/6)^{1/2} = 0.126$$

Comparison (B vs. A)	Difference ($\bar{X}_B - \bar{X}_A$)	SE	q	P	$q_{0.05,24,p}$*	Conclusion	
5 vs. 1	5.83-3.21=2.62	0.126	20.79	5	4.166	Reject	$H_0: u_5=u_1$
5 vs. 2	5.83-4.02=1.81	0.126	14.37	4	3.901	Reject	$H_0: u_5=u_2$
5 vs. 3	5.83-4.11=1.72	0.126	13.65	3	3.532	Reject	$H_0: u_5=u_3$
5 vs. 4	5.83-4.41=1.42	0.126	11.27	2	2.919	Reject	$H_0: u_5=u_4$
4 vs. 1	4.41-3.21=1.20	0.126	9.52	4	3.901	Reject	$H_0: u_4=u_1$
4 vs. 2	4.41-4.02=0.39	0.126	3.10	3	3.532	Accept	$H_0: u_4=u_2$
4 vs. 3	Do Not Test						
3 vs. 1	4.11-3.21=0.90	0.126	7.14	3	3.532	Reject	$H_0: u_3=u_1$
3 vs. 2	Do Not Test						
2 vs. 1	4.02-3.21=0.81	0.126	6.43	2	2.919	Reject	$H_0: u_2=u_1$

*Since $q_{0.05,25,p}$ does not appear in the q-distribution table, $q_{0.05,24,p}$ is used.

Overall conclusion: $u_1 \neq u_2 = u_3 = u_4 \neq u_5$

The q value is computed by:

$$q = (\bar{x}_B - \bar{x}_A)/SE$$

If the computed q value is greater than or equal to the critical value, then H_0: $u_B = u_A$ is rejected.

In Example 3, after accepting $H_0: u_4 = u_2$ there is no need to test 4 vs. 3 or 3 vs. 2. The conclusions drawn in the example are that the community at Station 1 has a significantly different mean diversity index from all other sampled communities; likewise, the Station 5 mean is different from the others. However, the communities at Stations 2, 3, and 4 have statistically equal diversity index means. These conclusions can be visually represented by underlining the means that are not significantly different with a common line as shown below:

station	1	2	3	4	5
mean diversity index	<u>3.21</u>	<u>4.02</u>	<u>4.11</u>	<u>4.41</u>	<u>5.83</u>

Conversely, any two means not underscored by the same line are significantly different.

Duncan's Multiple Range Test

The theoretical basis of the Duncan's test is somewhat different from the Student-Newman-Keul test, although the procedures and conclusions are quite similar. Duncan's test makes use of the concept of Least Significant

Difference (LSD) which is related to the t-test, a form of which was discussed previously. The LSD is calculated by:

$$LSD_\alpha = t_\alpha \, (2s^2/n)^{1/2}$$

where s^2 is the mean square for error, n is the number of replications, and t is the tabulated t value for the error degrees of freedom (MS and df for error are calculated in the analysis of variance). After determining p as in the SNK procedure, R values are obtained from a table dependent on the level of significance, error df, and p. The shortest significant difference (SSD) is computed by the equation:

$$SSD = R(LSD)$$

Example IV-2-4 demonstrates Duncan's procedure for hypothetical data. As before, the difference between means is calculated for every possible pairwise comparison of means. This difference is then compared to the corresponding SSD value and conclusions are drawn. If the difference is at least as large as the SSD, then the null hypothesis - that the two means are equal - is rejected; if the difference is less than SSD, H_0 is accepted. The results are visually represented as described for the SNK test.

<u>Example IV-2-4. Duncan's Multiple Range Test.</u>

H_0: $u_1=u_2=u_3=u_4$
H_A: The mean diversity indices of the four sampling stations are not the same

$\alpha = 0.05$ n = 4 error MS = 0.078 error df=9

Ranks of sample means (i)	1	2	3	4
Ranked sample means ($\bar{x}_i$)	5.3	5.7	5.9	6.3

$$LSD_{0.05} = t_{0.05} \, (2s^2/n)^{1/2} = 0.447$$

Comparison (B vs. A)	Difference ($\bar{X}_B - \bar{X}_A$)	P	$R_{\alpha,df,\rho}$	SSD =R(LSD)	Conclusion
4 vs. 1	6.3-5.3=1.0	4	1.07	0.48	reject $H_0:u_4=u_1$
	6.3-5.7=0.6	3	1.04	0.46	reject $H_0:u_4=u_2$
4 vs. 3	6.3-5.9=0.4	2	1.00	0.45	accept $H_0:u_4=u_3$
3 vs. 1	5.9-5.3=0.6	3	1.04	0.46	reject $H_0:u_3=u_1$
3 vs. 2	5.9-5.7=0.2	2	1.00	0.45	accept $H_0:u_3=u_2$
2 vs. 1	5.7-5.3=0.4	2	1.00	0.45	accept $H_0:u_2=u_1$

station	1	2	3	4
mean diversity index	5.3	5.7	5.9	6.3
visual representation				

COMMUNITY COMPARISON INDICES

Introduction

Whereas the statistical analyses discussed above can discern significant differences between diversity indices calculated at two or more sampling stations, community comparison indices have been developed to measure the degree of similarity or dissimilarity between communities. These indices can detect spatial or temporal changes in community structure. Polluted communities presumably will have different species occurrences and abundances than relatively non-polluted communities, given that all other factors are equal. Hence, community comparison indices can be used to assess the impact of pollution on aquatic biological communities.

There are two basic types of community comparison indices: qualitative and quantitative. Qualitative indices use binary data: in ecological studies, the two possible attribute states are that a species is present or is not present in the collection. This type of community similarity index is used when the sampling data consists of species lists. Kaesler and Cairns (1972) considered the use of presence-absence data to be the only justifiable (and defensible) approach when comparing a variety of organism groups (e.g. algae and aquatic insects). Also, qualitative similarity coefficients are simple to calculate. When data on species abundance are available, quantitative similarity indices can be used. Quantitative coefficients incorporate species abundance as well as occurrence in their formulas, and thus, retain more information than indices using binary data. An annotated list of community comparison indices of both types appears in Table IV-2-9.

Qualitative Similarity Indices

Although the terminology used in the literature varies considerably, the qualitative similarity indices in Table IV-2-9 (1 - 6) are represented using the symbolism of the 2X2 contingency table shown in Figure IV-2-5. In the form of the contingency table shown, collections A and B are entities and all of the species represented in a collection are the attributes of that entity.

Indices 1 through 4 in Table IV-2-9 are constrained between values of 0 and 1, while equation 6 has a potential range of -1 to 1. The minimum value represents two collections with no species in common and the maximum value indicates structurally identical communities.

According to Boesch (1977), the Jaccard, Dice, and Ochiai coefficients are the most attractive qualitative similarity measures for biological assessment studies. The Jaccard coefficient (1) is superior for discriminating between highly similar collections. The Dice (2) and Ochiai (4) indices place more emphasis on common attributes and are better at discriminating between highly dissimilar collections (Clifford and Stehpenson, 1975; Boesch, 1977; Herricks and Cairns, 1982). Thus, the nature of the data determines which index is most suitable. The Jaccard coefficient has been widely used by some workers in stream pollution investigations (Cairns and Kaesler, 1969; Cairns et al., 1970; Cairns and Kaesler, 1971; Kaesler at al., 1971; Kaesler and Cairns, 1972; Johnson and Brinkhurst, 1971; Foerster et al., 1974). Peters (1968) has written BASIC computer programs for calculating Jaccard, Dice, and Ochiai indices.

TABLE IV-2-9. SUMMARY OF COMMUNITY COMPARISON INDICES

Descriptive Name	Formula
1. Jaccard Coefficient of Community	$S = \frac{a}{a+b+c}$
2. Dice Index (Czekanowski, Sorenson)	$S = \frac{2a}{2a+b+c}$
3. Sokal and Michener Simple Matching Index	$S = \frac{a+b}{a+b+c+d}$
4. Ochiai Index (Otsuka)	$S = \frac{a}{[(a+b)(a+c)]^{1/2}}$
5. Fager Index	$S = \frac{a}{[(a+b)(a+c)]^{1/2}} - \frac{1}{2(a+b)^{1/2}}$
6. Point Correlation Coefficient (Kendall Coefficient of Association)	$S = \frac{ab-bc}{[(a+b)(c+d)(a+c)(b+d)]^{1/2}}$
7. Bray-Curtis Similarity Coefficient	$S_{ab} = \frac{2 \sum \min(x_{ia}, x_{ib})}{\sum (x_{ia} + x_{ib})}$
Bray-Curtis Dissimilarity Coefficient	$D_{ab} = \frac{\sum \lvert x_{ia} - x_{ib} \rvert}{\sum (x_{ia} + x_{ib})}$
Percentage Similarity of Community	$S_{ab} = 1 - 0.5 \sum \lvert p_{ia} - p_{ib} \rvert = \sum \min (p_{ia}, p_{ib})$

TABLE IV-2-9 (continued)

8.	Pinkham and Pearson Index of Similarity	$S_{ab} = \frac{1}{n} \sum \frac{\min (x_{ia}, x_{ib})}{\max (x_{ia}, x_{ib})}$	Pir
9.	Morisita Index of Affinity	$S_{ab} = \frac{2 \sum x_{ia} x_{ib}}{(\lambda_a + \lambda_b) X_a X_b}$	Mor
10.	Horn Index of Overlap	$S_{ab} = \frac{H_{max} - H_{ab}}{H_{max} - H_{min}}$	Hor
11.	Distance	$D_{ab} = [\sum (x_{ia} - x_{ib})^2]^{1/2}$	Boe
		$\bar{D}_{ab} = [\frac{1}{n} \sum (x_{ia} - x_{ib})^2]^{1/2}$	Sok
12.	Product-Moment Correlation Coefficient (Pearson)	$S_{ab} = \frac{\sum (x_{ia} - \bar{x}_a)(x_{ib} - \bar{x}_b)}{[\sum (x_{ia} - \bar{x}_a)^2 (x_{ib} - \bar{x}_b)^2]^{1/2}}$	Sne

TABLE IV-2-9 (continued)

Key: S = similarity between samples.
D = dissimilarity between samples.
a,b,c,d = (see Figure IV-2-5).
x_{ia}, x_{ib} = number of individuals of species i at Station A or B.
p_{ia}, p_{ib} = relative abundance of species i at Station A or B.
X_a, X_b = total number of individuals at Station A or B.
n = total number of different taxa.
λ_a, λ_b = Simpson diversity index for Station A or B.
H_{ab} = Shannon-Wiener diversity index of Station A and B combined.
H_{max} = maximum possible value of H_{ab}.
H_{min} = minimum possible value of H_{ab}.

$$H_{ab} = - \sum \frac{x_{ia} + x_{ib}}{X_a + X_b} \log \frac{x_{ia} + x_{ib}}{X_a + X_b}$$

$$H_{max} = \frac{(X_a + X_b) \log (X_a + X_b) - \sum x_{ia} \log x_{ia} - \sum x_{ib} \log x_{ib}}{X_a + X_b}$$

$$H_{min} = \frac{(X_a \sum \frac{x_{ia}}{X_a} \log \frac{x_{ia}}{X_a} + X_b \sum \frac{x_{ib}}{X_b} \log \frac{x_{ib}}{X_b})}{X_a + X_b}$$

COLLECTION B \ COLLECTION A	present	absent
present	a number of species common to both collections	b number of species present in B but not in A
absent	c number of species present in A but not in B	d number of spcies not represented in either collection

Figure IV-2-5. 2 x 2 contingency table defining variables a, b, c, and d.

The Fager coefficient (5) is simply a modification of the Ochiai index. Because a correction factor is subtracted from the Ochiai index, the Fager coefficient may range from slightly less than zero to slightly less than one; this makes it less desirable. The Fager index has been used a great deal in marine ecology.

Both the Sokal and Michener index (3) and the Point Correlation Coefficient (6) include the double-absent term d. A number of authors (Kaesler and Cairns, 1972; Clifford and Stephenson, 1975; Boesch, 1977) have criticized the approach of considering two collections similar on the basis of species being absent from both.

Pinkham and Pearson (1976) illustrated the weaknesses of qualitative comparison indices. The basic shortcoming is that two communities having completely different species abundances but the same species occurrence will produce the maximum index value, indicating that the two collections are identical.

Quantitative Comparison Indices

Quantitative indices (7 - 12) consider species abundance in addition to mere presence-absence. Incorporating species abundance precludes the over-emphasis of rare species, which has been a criticism of the Jaccard coefficient (Whittaker and Fairbanks, 1958). Quantitative measures are not as sensitive to rare species as qualitative indices and emphasize dominant species to a greater extent. Distance (11), information (9, 10), and correlation (12) coefficients weight dominance even more than other quantitative indices. Quantitative indices also avoid the loss of information involved in considering only presence-absence data when species abundance data are available. However, data transformations (e.g., to logarithms, roots, or percentages) may be desirable or necessary for the use of some quantitive comparison indices. Calculation of quantitative indices is more complicated than qualitative coefficients, but can be facilitated by computer application.

The Bray-Curtis index (7) is one of the most widely used quantitive comparison measures. Forms of this index have been referred to as "index of associaton" (Whittaker, 1952), as "dominance affinity" (Sanders, 1960), and as "percentage similarity of community" (Johnson and Brinkhurst, 1971; Pinkham and Pearson, 1976; Brock, 1977). The simplest and probably most commonly used form of the Bray-Curtis index is the Percent Similarity equation:

$$S_{ab} = \sum \min(p_{ia}, p_{ib})$$

where the attributes have been standardized into a proportion or percent of the total for that entity (collection). The shortcoming of the Percent Similarity coefficient was illustrated by Pinkham and Pearson (1976) as shown below.

	TAXA				
	A	B	C	D	E
Station A	40	20	10	10	10
Station B	20	10	5	5	5

In this hypothetical comparison, all species are twice as abundant at Station A as at Station B but their relative abundance is identical; therefore, the maximum similarity value of 1.0 is registered. The authors felt that this situation is germane to pollution assessment surveys in which the only difference between two sampling stations is the relative degree of cultural eutrophication.

In Table IV-2-9, the Bray-Curtis index is displayed as both a measure of similarity and dissimilarity. Any community similarity index can be converted to a dissimilarity measure by the simple equality:

$$D = 1 - S$$

Of course, values obtained by a dissimilarity expression are inversely related to similarity values; they increase with decreasing similarity.

Pinkham and Pearson (1976) presented a community similarity index (8) that would overcome the shortcomings of other indices (e.g. 1,3,7,12) that were discussed in the article. Their similarity coefficient can be calculated using either actual or relative (percent) species abundance, although they suggested using actual abundance whenever possible. The authors also offered a modified formula that includes a weighting factor for assigning more significance to dominant species:

$$S_{ab} = \sum \frac{\min(x_{ia}, x_{ib})}{\max(x_{ia}, x_{ib})} \left[\frac{x_{ia}}{X_a} \cdot \frac{x_{ib}}{X_b} \ / \ 2 \right]$$

Two community comparison indices that employ diversity indices in their formulas are the Morisita Index of Affinity (9) and the Horn Index of Overlap (10). The Morisita comparison measure incorporates the Simpson (1949) diversity index, and the Horn coefficient uses the Shannon-Wiener (1948) diversity index. Horn (1966) described the Morisita index as the probability that two individuals drawn randomly from communities A and B will both belong to the same species, relative to the probability of randomly drawing two individuals of the same species from A or B alone. Because the numerator of the Morisita index is a product rather than a difference (or minimum value) it tends to be affected by abundant species to a greater extent than the Bray-Curtis or Pinkham and Pearson indices. Like those similarity measures, the Morisita index ranges from zero for no resemblance to one for identical collections. The Horn Index of Overlap is a manipulation of Shannon's information theory equation that closely resembles the expression of community redundancy developed by Margalef:

$$R = (H_{max} - H)/ (H_{max} - H_{min})$$

The observed value in Horn's index (Hab) is the Shannon index calculated for the sum of the two collections being considered. The maximum diversity value (Hmax) would occur if the two collections contained no species in common, and the minimum diversity value (Hmin) would be attained if the two collections contained the same species in the same proportions. It should be noted that the equations given for Hab, Hmax, and Hmin in the key to Table IV-2-9 are adapted from those given by Perkins (1983) since those appearing in the original article (Horn, 1966) are apparently inconsistent with the Shannon index. The Morisita and the Horn indices have been used in aquatic ecology studies (Kohn, 1968; Bloom et al., 1972; Livingston, 1975; Heck, 1976).

If two entities (i.e. communities) are thought of as points in an n-dimensional space whose dimensions are determined by their attributes (i.e. species occurrence and abundance), then the linear distance between the two points in the hyperspace can be construed as a measure of dissimilarity between the two entities. The two distance formulas shown in Table IV-2-9 (11) are simply forms of the familiar geometrical distance formula,

$$d = \left[(x_1-x_2)^2 + (y_1-y_2)^2\right]^{1/2}$$

which has been expanded to accomodate n dimensions. Sokal (1961) divided the distance by n to produce a mean squared difference, which he felt was an appropriate measure of taxonomic distance. Values computed by the distance formulas may range from zero for identical collections to infinity; the greater the distance the less similar the two comunities are. Because the difference in species abundance is squared in the numerator, the distance formulas are heavily influenced by abundant species and may over-emphasize dominance. The similarity of disparate communities with low species abundances may be overstated, while the resemblance of generally similar communities with a few disproportionately high species abundances may be understated. To avoid indicating misleading resemblance, it may be necessary to transform data (e.g. to squared or cubed roots) before computing taxonomic distance.

The Product-Moment Correlation Coeffficient (12) is a popular resemblance measure that ranges from -1 (completely dissimilar) to +1 (entirely similar). Several undersirable characteristics of this measure have been cited (Sneath and Sokal, 1973; Clifford and Stephenson, 1975; Boesch, 1977). Deceptive resemblance values can result from outstandingly high species abundances or the presence of many species absences, and non-identical communities can register perfect correlation scores. Pinkham and Pearson (1976) demonstrated how the Product-Moment Correlation Coefficient, like the Percent Community Similarity Index, indicates maximum similarity for two communities having the same relative species composition but different actual species abundances.

Experimental Evaluation of Comparison Indices

Brock (1977) compared the Percent Community Similarity Index (7) and the Pinkham and Pearson Similarity Index (8) for their ability to detect changes in the zooplankton community of Lake Lyndon B. Johnson, Texas, due to a thermal effluent. For this study, the Pinkham and Pearson index was considered too sensitive to rare species and not sensitive enough to dominant forms, whereas the Percent Similarity coefficient was more responsive to variation in dominant species and relationships between dominant and semi-dominant forms. Linking dominance to function, the author concluded that the later index may better indicate structural-funcitonal similarity between communities.

Perkins (1983) evalutaed the responsiveness of eight diversity indices and five community comparison indices to increasing copper concentrations. The indices were calculated for bioassays conducted using benthic macroinvertebrates and artificial streams. The indices evaluated by Perkins correspond to equations presented in Tables IV-2-2 and IV-2-9 except: Perkins tested the Bray-Curtis dissimilarity index; Perkins' Biosim index is Pinkham and Pearson's index, and the distance formula tested by Perkins (not included in this report) is shown below.

$$D = \left[\frac{1}{n} \sum \left(\frac{x_{ia}-x_{ib}}{x_{ia}+x_{ib}}\right)^2\right]^{1/2}$$

The results of the study appear in Figure IV-2-6; the diversity index results are presented for comparison.

The diversity indices did not clearly demonstrate the perturbation caused by increasing copper concentrations. The Shannon and Brillouin formulas increased initially, in spite of a decreasing number of species, because of increasing evenness of species distribution. Other than the increasing diversity indicated at the lower copper concentrations, these two indices reflected perturbation effectively by decreasing rapidly with increasing pollutant concentration. The McIntosh, Simpson, and Pielou (evenness) indices (not shown for 28 days in Figure IV-2-6) resembled the trends demonstrated by the Shannon and Brillouin formulas albeit less dramatically. Because the results obtained for those three indices were less pronounced, they were more difficult to interpret than the Shannon and Brillouin findings.

The community comparison indices were found to be good indicators of the perturbation of macroinvertebrate communities caused by copper pollution. Although the Bray-Curtis index was considered the most accurate after 14 days, all of the comparison indices tested effectively reflected community response after 28 days (see Figure IV-2-6). Note that by definition the Biosim, Morisita, and Percent Community Similarity indices decrease as similarity decreases, while the Distance and Bray-Curtis dissimilarity indices increase. It has frequently been suggested that it may be desirable to apply several indices in a pollution assessment study (Peters, 1968; Brock, 1977; Perkins, 1983).

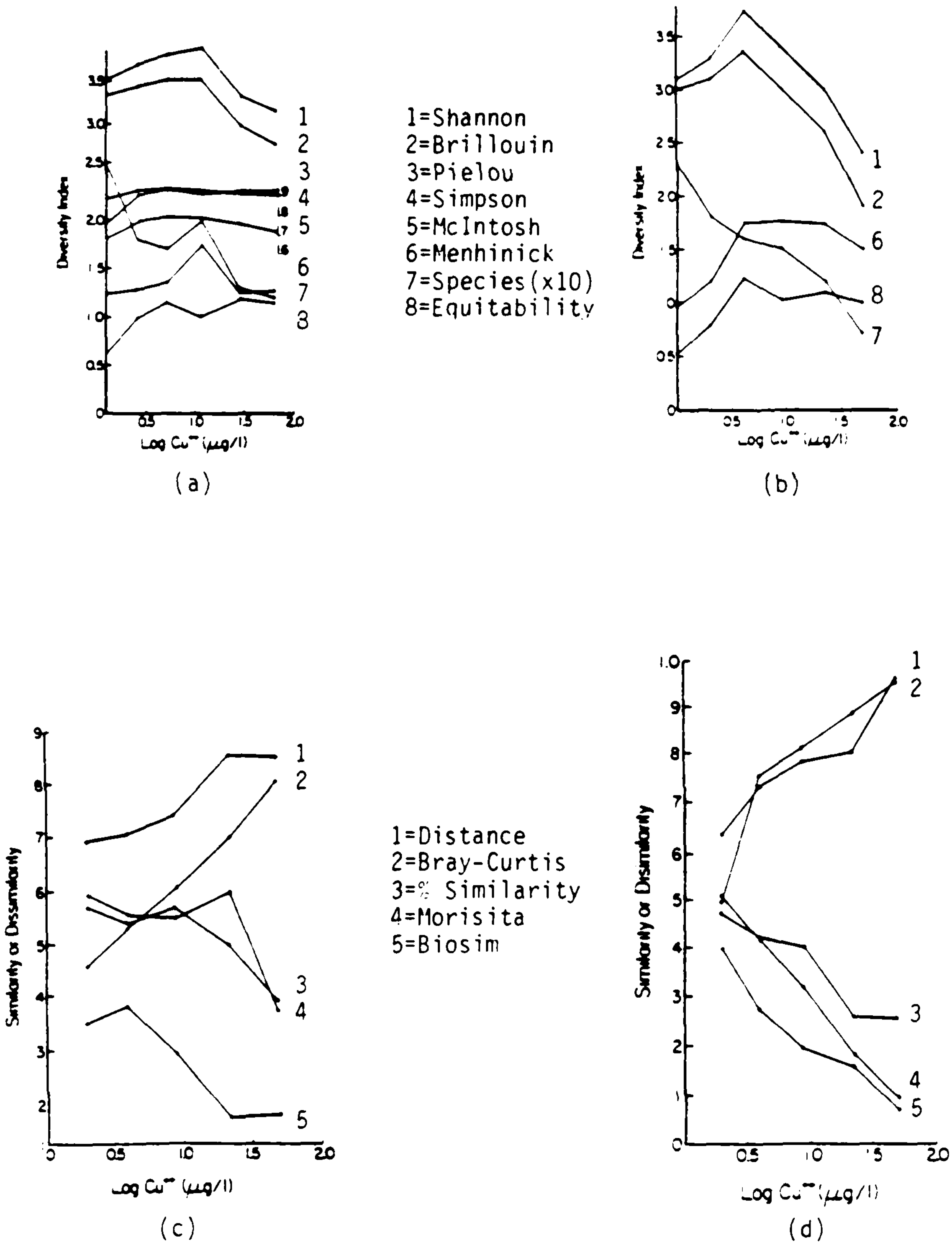


Figure IV-2-6. Evaluation of diversity indices and community comparison indices using bioassay data: a,c=after 14 days; b,d=after 28 davs (from Perkins, 1983).

Numerical Classification or Cluster Analysis

A common use of similarity indices is in numerical clasification of biological communities. Numerical classification, or cluster analysis, is a technique for grouping similar entities on the basis of the rsemblance of their attributes. In instances where subjective classification of communities is not clear-cut, cluster analysis allows incorporation of large amounts of attribute data into an objective classification procedure. Kaesler and Cairns (1972) outlined five steps involved in normal cluster analysis. First, a community similarity index is chosen based on pre-determined criteria and objectives. Second, a matrix of similarity coefficients is generated by pairwise comparison of all possible combinations of stations. The third step is the actual clustering based on the resemblance coefficients. A number of clustering procedures are discussed in the literature (Williams, 1971; Sneath and Sokal, 1973; Hartigan, 1975; Boesch, 1977). In the fourth step, the clustered stations are graphically displayed in a dendogram. Because multi-dimensional resemblance patterns are displayed in two dimensions and because the similarity coefficients are averaged, a significant amount of distortion can occur. For this reason, a distortion measure should be evaluated and presented as the fifth step in the cluster analysis. The Cophenetic Correlation Coefficient (Sokal and Rohlf, 1962) is a popular metric of display accuracy. An additional step in any cluster analysis application should be interpretation of the numerical classification results since the technique is designed to simplify complex data and not to produce ecological interpretation.

SUMMARY

The ability of a water resource to sustain a balanced biotic community is one of the best indicators of its potential for beneficial use. This ability is essential to the community's health. Although several papers have criticized the use of diversity indices (Hurlbert,1971; Peet,1975; Godfrey,1978), Cairns (1977) stated that "the diversity index is probably the best single means of assessing biological integrity in freshwater streams and rivers". Cairns concluded that no single method will adequately assess biological integrity, but rather its quantification requires a mix of assessment methods suited for a specific site and problem. The index of diversity is an integral part of that mix. Community comparison indices are also useful in assessing the biological health of aquatic systems. By measuring the simiarity (or dissimilarity) between sampling stations, community comparison indices indicate relative impairment of the aquatic resource.

CHAPTER IV-3
RECOVERY INDEX

It is important to examine the ability of an ecosystem to recover from displacement due to pollutional stress in order to evaluate the potential uses of a water body. Cairns (1975) developed an index which gives an indication of the ability of the system to recover after displacement. The factors and rating system for each factor are:

(a) Existence of nearby epicenters (e.g., for rivers these might be tributaries) for providing organisms to reinvade a damaged system.
Rating System : 1=poor, 2=moderate, 3=good

(b) Transportability or mobility of disseminules (the disseminules might be spores, eggs, larvae, flying adults which might lay eggs, or other stages in the life history of an organism which permit it to move to a new area).

Rating System : 1=poor, 2-moderate, 3=good

(c) Condition of the habitat following pollutional stress (including physical habitat and chemical quality).
Rating system : 1=poor, 2=moderate, 3=good

(d) Presence of residual toxicants following pollutional stress.
Rating System : 1=large amounts, 2=moderate amounts, 3=none

(e) Chemical-physical environmental quality after pollutional stress.
Rating System : 1=in severe disequilibrium, 2=partially restored, 3=normal

(f) Management or organizational capabilities for control of damaged area.
Rating system : 1=none, 2-some, 3=strong enforcement possible.

Using the characteristics listed above, and their respective rating systems, a recovery index can be developed. The equation for the recovery index follows:

Recovery Index = a x b x c x d x e x f
400+ = chances of rapid recovery excellent
55-399 = chances of rapid recovery fair to good
less than 55 = chances of rapid recovery poor

This index and the rating system was developed by Cairns based on his experience with the Clinch River. For a full description of the rationale for the rating factor, the reader should refer to Cairns (1975).

CHAPTER IV-4

INTOLERANT SPECIES ANALYSIS

NICHE CONCEPT

The ecological niche of a species is its position and role in the biological community. Hutchinson (1957) described niche as a multidimensional space, or hypervolume, that is delineated by the species' environmental requirements and tolerances. Physical, chemical, and biological conditions and relationships constitute the dimensions of the hypervolume, and the magnitude of each dimension is defined by the upper and lower limits of each environmental variable within which a species can persist. If any one of the variables is outside of this range the organism will die, regardless of other environmental conditions.

TOLERANCE

The "Law of Toleration" proposed by Shelford (1911) is illustrated in Figure IV-4-1. For each species and environmental variable there is a range in the variable intensity over which the organism functions at or near its optimum level. Outside the maximum and minimum extremes of the optimum range there are zones of physiological stress, and, beyond, there are zones of intolerance in which the functions of the organism are inhibited. The upper and lower tolerance limits (also called incipient lethal levels) are intensity levels of the environmental variable that will eventually cause the death of a stated fraction of test organisms, usually 50 percent.

VARIABILITY OF TOLERANCE

The tolerance of an organism for a lethal condition is dependent on its genetic constitution - both its species and its individual genetic makeup - and its early and recent environmental history (Warren 1971). Acclimation has a marked effect on the tolerance of environmental factors such as temperature, dissolved oxygen, and some toxic substances (see Figure IV-4-2). Tolerance is also a function of the developmental stage of the organism and it may change with age throughout the life of the animal. Because of this variability, no two organisms have exactly the same tolerance for a lethal condition and tolerance limits must be expressed in terms of an "average" organism.

INTERACTIONS INFLUENCING TOXICITY

An organism's tolerance for a particular lethal agent is dependent not only on its own characteristics but also on the environmental conditions. The interactions between lethal and nonlethal factors are well documented and are addressed elsewhere in this handbook (Chapters II-5 and III-2). Briefly, these nonlethal effects include:

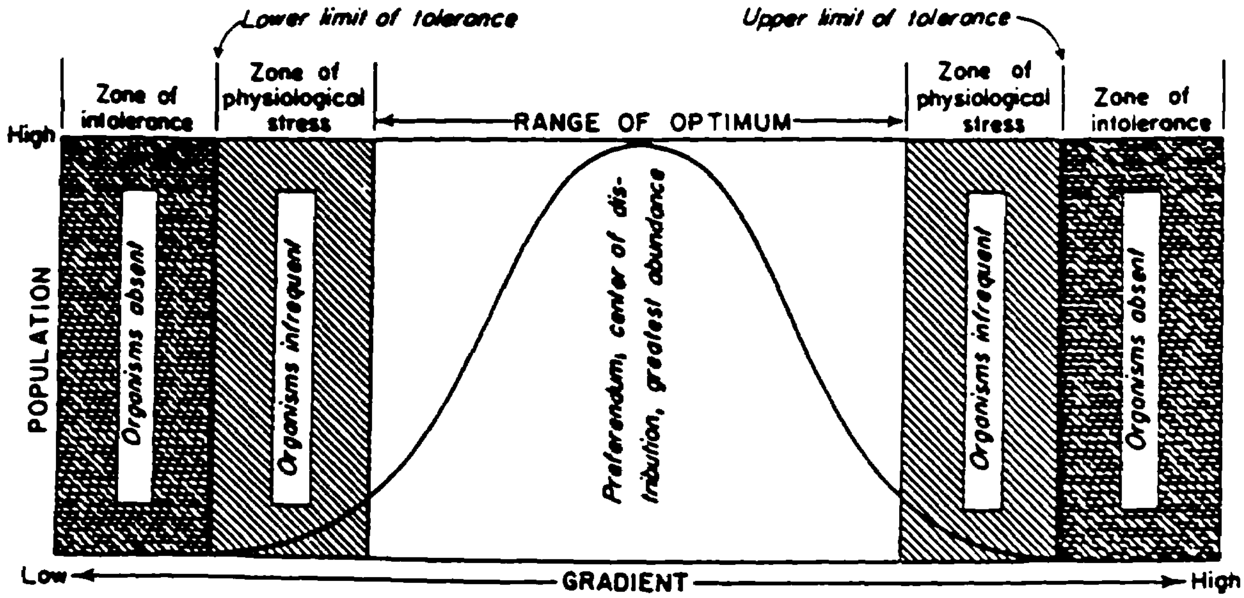


Figure IV-4-1. Law of toleration in relation to distribution and population level--often a normal curve (modified by Kendeigh (1974) from Shelford (1911)).

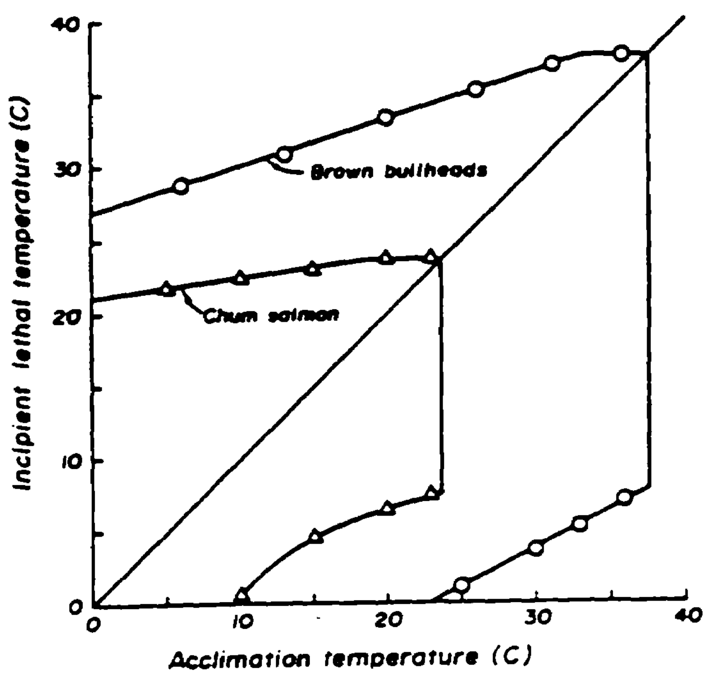


Figure IV-4-2. The zones of tolerance of brown bullheads (Ictalurus nebulosus) and chum salmon (Oncorhynchus keta) as delimited by incipient lethal temperature and influenced by acclimation temperature (after Brett 1956).

CHAPTER IV-4

INTOLERANT SPECIES ANALYSIS

NICHE CONCEPT

The ecological niche of a species is its position and role in the biological community. Hutchinson (1957) described niche as a multidimensional space, or hypervolume, that is delineated by the species' environmental requirements and tolerances. Physical, chemical, and biological conditions and relationships constitute the dimensions of the hypervolume, and the magnitude of each dimension is defined by the upper and lower limits of each environmental variable within which a species can persist. If any one of the variables is outside of this range the organism will die, regardless of other environmental conditions.

TOLERANCE

The "Law of Toleration" proposed by Shelford (1911) is illustrated in Figure IV-4-1. For each species and environmental variable there is a range in the variable intensity over which the organism functions at or near its optimum level. Outside the maximum and minimum extremes of the optimum range there are zones of physiological stress, and, beyond, there are zones of intolerance in which the functions of the organism are inhibited. The upper and lower tolerance limits (also called incipient lethal levels) are intensity levels of the environmental variable that will eventually cause the death of a stated fraction of test organisms, usually 50 percent.

VARIABILITY OF TOLERANCE

The tolerance of an organism for a lethal condition is dependent on its genetic constitution - both its species and its individual genetic makeup - and its early and recent environmental history (Warren 1971). Acclimation has a marked effect on the tolerance of environmental factors such as temperature, dissolved oxygen, and some toxic substances (see Figure IV-4-2). Tolerance is also a function of the developmental stage of the organism and it may change with age throughout the life of the animal. Because of this variability, no two organisms have exactly the same tolerance for a lethal condition and tolerance limits must be expressed in terms of an "average" organism.

INTERACTIONS INFLUENCING TOXICITY

An organism's tolerance for a particular lethal agent is dependent not only on its own characteristics but also on the environmental conditions. The interactions between lethal and nonlethal factors are well documented and are addressed elsewhere in this handbook (Chapters II-5 and III-2). Briefly, these nonlethal effects include:

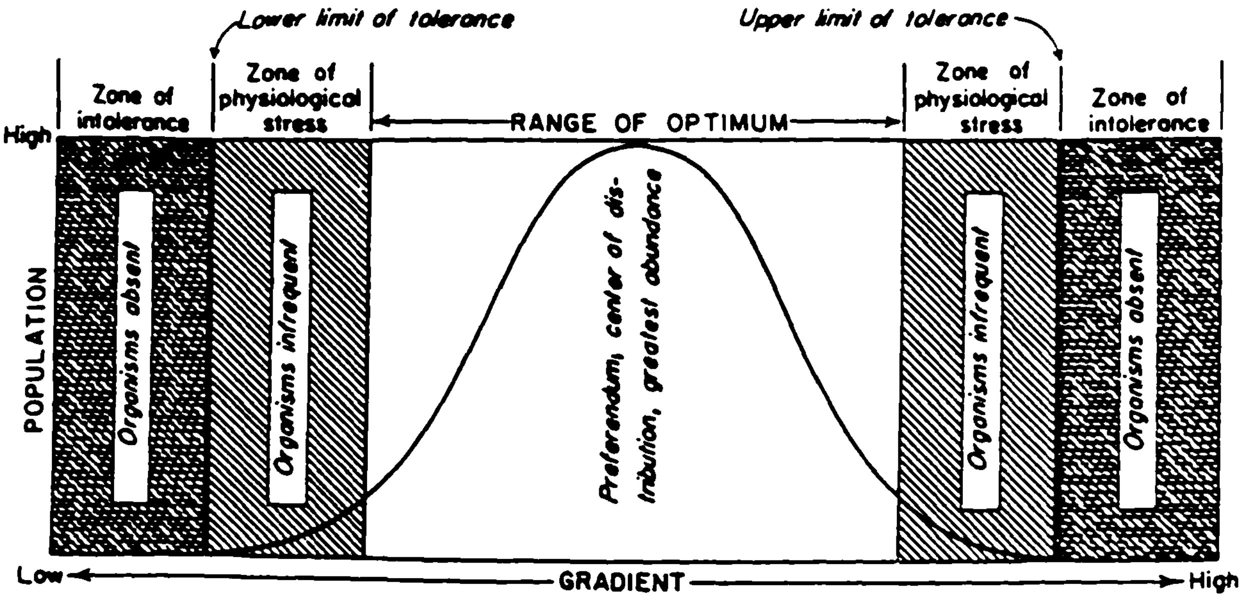


Figure IV-4-1. Law of toleration in relation to distribution and population level--often a normal curve (modified by Kendeigh (1974) from Shelford (1911)).

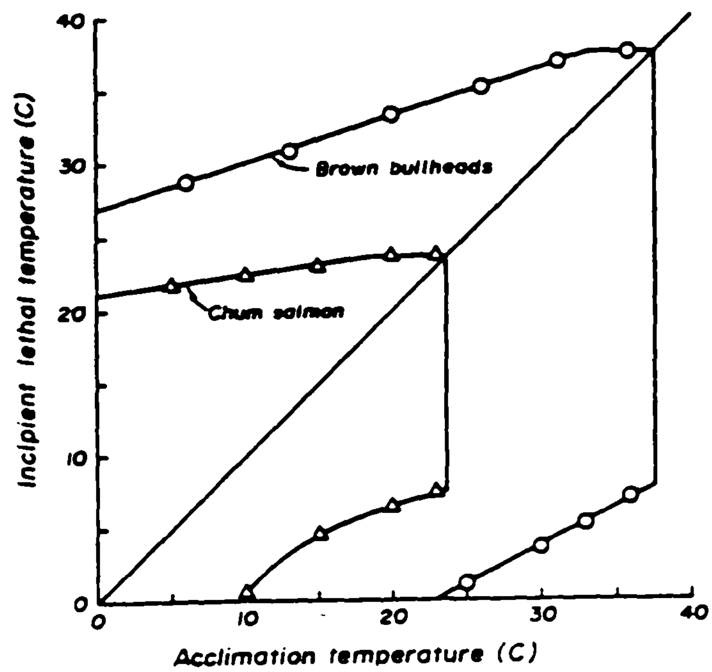


Figure IV-4-2. The zones of tolerance of brown bullheads (Ictalurus nebulosus) and chum salmon (Oncorhynchus keta) as delimited by incipient lethal temperature and influenced by acclimation temperature (after Brett 1956).

- Hardness. Increasing hardness decreases the effect of toxic metals on aquatic organisms by forming less-toxic complexes.
- pH. The dissociation of weak acids and bases is controlled by pH and either the molecular or ionic form may be more toxic.
- Alkalinity and Acidity. These modify pH by constituting the buffering capacity of the system.
- Temperature. Increasing temperature enhances the effect of toxicants by increasing the rates of metabolic processes.
- Dissolved Oxygen. Decreasing dissolved oxygen concentration augments the exposure and absorption of toxicants by increasing the necessary irrigation rate of respiratory organs.

When two or more lethal agents are present, several types of interactions are possible: synergistic, additive, antagonistic, or no interaction.

INTOLERANT SPECIES ANALYSIS

The tolerance ranges for environmental variables differ widely between species. Thus, the range of conditions under which an organism can survive (its niche) is broader for some species than it is for others. Fish species with narrow tolerance ranges are relatively sensitive to degradation of water quality and other habitat modifications, and their populations decline or disappear under those circumstances before more tolerant organisms are affected. In general, intolerant species can be identified and used in evaluating environmental quality. The presence of typically intolerant species in a fish sampling survey indicates that the site has relatively high quality; while the absence of intolerant species that, it is judged, would be there if the environment was unaltered indicates that the habitat is degraded.

LISTS OF INTOLERANT FISH SPECIES

While the tolerance limits of a fish species for a particular environmental factor can be defined relatively precisely by toxicity bioassays, its degree of tolerance may vary considerably over the range of physical, chemical, and biological variables that may be encountered in the environment. The variables that are the object of intolerant species analysis are intentionally left vague in order to accommodate the variety of situations precipitated by man's activities. A species may be intolerant of alterations in water quality or in habitat structure, such as those listed below.

Water Quality Changes	Habitat Alterations
increased turbidity	substrate disruption
increased siltation	cover removal
increased water temperature	changes in velocity and discharge
increased dissolved solids	removal of instream and streamside vegetation
organic enrichment	water level fluctuation
lowered dissolved oxygen	impoundment and channelization
	blockage or hinderance of migration

Many species can be identified that are relatively intolerant of anthropogenic alterations of the aquatic environment compared to other fish. Appendix C contains a list of fish species, nationally, which are relatively intolerant to one or more of the environmental changes shown above. The information in Appendix C is based on literature sources (Wallen 1951; Trautman 1957; Carlander 1969, 1977; Scott and Crossman 1973; Pflieger 1975; Moyle 1976; Timbol and Maciolek 1978; Smith 1979; Muncy et al. 1979; Lee et al. 1980; Morrow 1980; Johnson and Finley 1980; U.S. EPA 1980; Karr 1981; Haines 1981; and Ball 1982) and on the professional judgment of State and University biologists.

The darters and sculpins are listed only by genus in Appendix C. Identification of those taxa to species would have been inconvenient (together, *Ammocrypta*, *Etheostoma*, *Percina*, and *Cottus* contain 150 species in the United States) and largely unnecessary because, with a few possible exceptions, all of the species of darters and sculpins can be considered intolerant. Karr (1981) recognized the johnny darter (*Etheostoma nigrum*) as the most tolerant darter species in Illinois and Ball (1982) did not categorize the johnny darter as an intolerant forage fish. Other darter species that appear to be relatively more tolerant of turbidity, silt, and detritus than others in their genus are listed below:

mud darter	*Etheostoma asprigene*
bluntnose darter	*E. chlorosomum*
slough darter	*E. gracile*
cypress darter	*E. proeliare*
orangethroat darter	*E. spectabile*
swamp darter	*E. fusiforme*
river darter	*Percina shumardi*

The list in Appendix C is intended to be used by knowledgeable biologists as a rough guide to the relatively intolerant fish species in their state. Site-specific editing is left to persons familiar with the local fish fauna and environmental conditions. Local editing of the provided data should produce a workable list for intolerant species analyses of the streams in that area.

CHAPTER IV-5

OMNIVORE-CARNIVORE (TROPHIC STRUCTURE) ANALYSIS

INTRODUCTION

Water pollution problems nearly always involve changes in the pathways by which aquatic populations obtain energy and materials (Warren 1971). These changes lead to differential success of constituent populations which affects the composition of the aquatic community. Anthropogenic introduction of organic substances or mineral nutrients directly increases the energy and material resources of the system, but other pollution problems - such as pH or temperature changes, toxic materials, low dissolved oxygen, turbidity, siltation, et cetera - also lead to changes in trophic pathways. Thus, the health of a system can be evaluated through a study of its trophic structure. The following material concentrates on stream and river systems. Lakes will have different structural aspects.

TROPHIC STRUCTURE

The ecosystem has been described as the entire complex of interacting physicochemical and biological activities operating in a relatively self-supporting community (Reid and Wood 1976). The biological operations of an ecosystem can be viewed as a series of compartments which are described by three general categories: producers, consumers, and decomposers. The producers include all autotrophic plants and bacteria (both photosynthetic and chemosynthetic) which, by definition, are capable of synthesizing organic matter from inorganic substrates. The consumers are heterotrophic organisms that feed on other organisms, and are typically divided into herbivores and carnivores. Herbivores (primary consumers) feed principally on living plants while carnivores (secondary, tertiary, and quarternary consumers) feed principally on animals that they kill. Another type of consumer, the omnivore, feeds nearly equally on plants and animals, and occupies two or more trophic levels. The decomposers include all organisms that release enzymes which break down dead organisms.

Food chains are sometimes used to simply represent feeding relationships between trophic levels (e.g., plant > herbivore > carnivore). Ecosystems commonly contain three to five links in their food chains. Diagramming all of the pathways of energy and material transfer in a community entails many interconnecting food chains, forming a complex food web.

The concept of trophic structure, first formally discussed by Lindeman (1942), is a method of dealing with the pathways of energy and material transfer which focuses on functional compartments without considering the specific feeding relationships. The pathways between functional compartments are illustrated in Figure IV-5-1. Trophic structure is commonly represented by trophic or ecological pyramids. An ecological pyramid is a diagramatic representation of the relationships between trophic levels arranged with the producers making up the base and the terminal or top carnivore at the apex. An ecological pyramid may

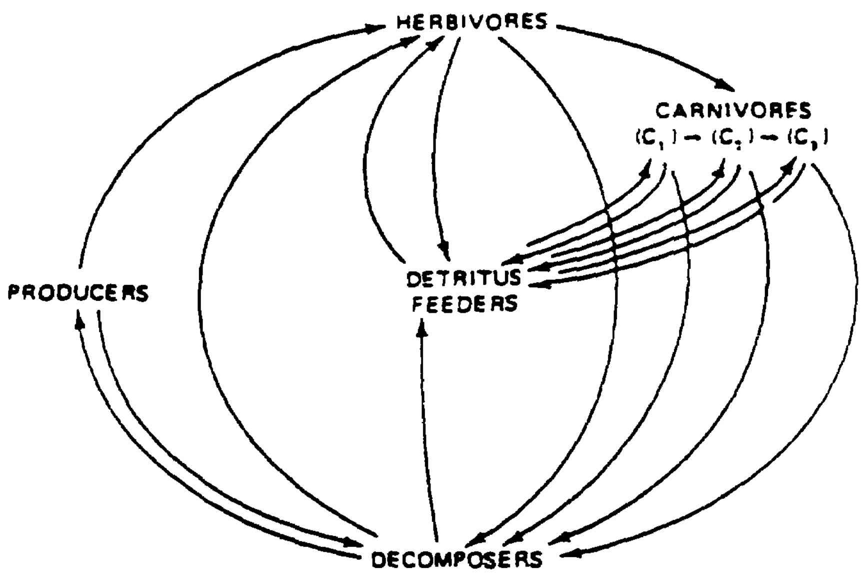


Figure IV-5-1. Trophic pathways of an ecosystem (after Reid and Wood 1976).

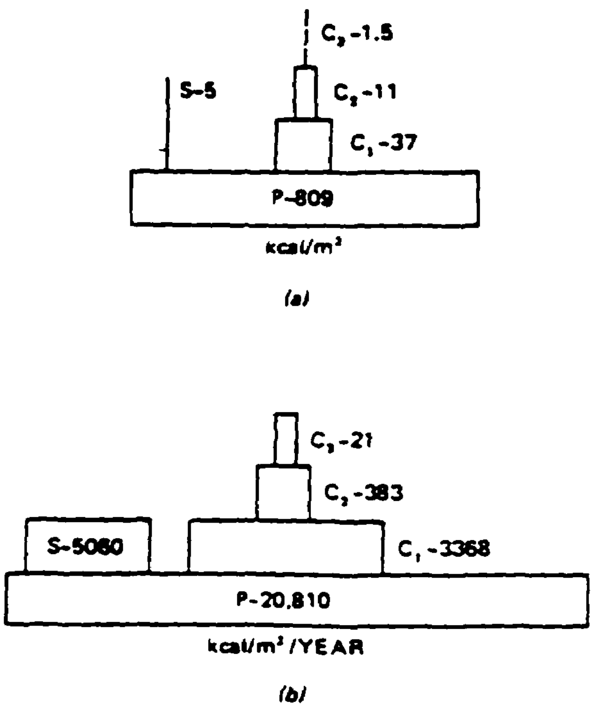


Figure IV-5-2. Ecological pyramids for Silver Springs, Florida, indicating (a) biomass and (b) productivity. P=producers; C=consumers; S=saprophytes or heterotrophs (after Odum 1957).

represent the number of individuals that compose each trophic level, or, of more ecological significance, the biomass or productivity of each level (Figure IV-5-2). Because energy transfer between trophic levels is less than 100 percent efficient the pyramid of productivity must always be regular in shape, while pyramids of numbers and biomass may be partially inverted in some instances (Richardson 1977).

TROPHIC STRUCTURE OF FISH COMMUNITIES

Fish communities generally include a range of species that represent a variety of trophic levels. The trophic classification system shown below was used in the assessment of fish fauna of the Illinois and Maumee River basins (Karr and Dudley 1978, Karr et al. 1983).

(1) Invertivore - food predominantly (>75%) invertebrates.
(2) Invertivore/Piscivore - food a mixture of invertebrates and fish; relative proportions often a function of age.
(3) Planktivore - food dominated by microorganisms extracted from the water column.
(4) Omnivore - two or more major (>25% each) food types consumed.
(5) Herbivore - feed mostly by scraping algae and diatoms from rocks, and other stream substrates.
(6) Piscivore - feed on other fish.

Schlosser (1981, 1982a, 1982b) used the trophic structure of fish communities to investigate differences in Illinois stream ecosystems. His categorization scheme appears in Table 1.

In addition to representing a range of trophic levels, fish utilize foods of both aquatic and terrestrial origin, and occupy a position at the top of the aquatic food web in relation to plants and invertebrates. These facts enhance the ability of fish communities to provide an integrative view of the watershed environment (Karr 1981).

BIOLOGICAL HEALTH

Degradation of water quality and habitat affects the availability of many food resources, resulting in changes in the structure and functions, and, thus, the health of the aquatic community. Structural characteristics include the numbers and kinds of species and the number of individuals per species. These parameters can be evaluated relatively quickly via compilation of species lists, calculation of diversity indices, and identification of indicator species. The importance of evaluating the impact of pollution on community functions - such as production, respiration, energy flow, degradation, nutrient cycling, and other rate processes - is becoming increasingly evident, and, ideally, any study of community health should include both structural and functional assessment. However, use of functional methods has been hindered because they are often expensive, time-consuming, and not well understood.

TABLE IV-5-1. TROPHIC GUILDS USED BY SCHLOSSER (1981, 1982A, 1982B) TO CATEGORIZE FISH SPECIES

Herbivore - detritivores (HD)	HD species fed almost entirely on diatoms or detritus.
Omnivores (OMN)	OMN species consumed plant and animal material. They differed from GI species in that, subjectively, greater than 25 percent of their diet was composed of plant or detritus material.
Generalized Insectivores (GI)	GI species fed on a range of animal and plant material including terrestrial and aquatic insects, algae, and small fish. Subjectively, less than 25 percent of their diet was plant material.
Surface and Water Column Insectivores (SWI)	SWI species fed on water column drift or terrestrial insects at the water surface.
Benthic insectivores (BI)	BI species fed predominantly on immature forms of benthic insects.
Insectivore - Piscivores (IP)	IP species fed on aquatic invertebrates and small fish. Their diets ranged from predominantly fish to predominantly invertebrates.

Examining the trophic structure of a community can provide insight into its production and consumption dynamics. A trophic-structure approach to the study of the functional processes of stream ecosystems has been proposed by Cummins and his colleagues (Cummins 1974, 1975; Vannote et al. 1980). Their concept assumes that a continuous gradient of physical conditions in a stream, from its headwaters to its mouth, will illicit a series of consistent and predictable responses within the constituent populations. The River Continuum Concept identifies structural and functional attributes that will occur at different reaches of natural (unperturbed) stream ecosystems. These attributes (summarized in Table IV-5-2) can serve as a reference for comparison to measured stream data. Measured data which are commensurate with those predicted by the river continuum model indicate that the studied system is unperturbed, while disagreement between actual and expected data indicates that modification of the ecosystem has occurred (Karr and Dudley 1978).

EVALUATION OF BIOLOGICAL HEALTH USING FISH TROPHIC STRUCTURE

Karr (1981) developed a system for assessing biotic integrity using fish communities, which is discussed in Chapter IV-2: Diversity Indices. Three empirical trophic metrics are incorporated into Karr's index of biotic integrity (IBI). They are:

(1) the proportion of individuals that are omnivores,
(2) the proportion of insectivorous individuals of the Cyprinidae family, and
(3) the presence of top carnivore populations.

Karr (1981) observed that the proportion of omnivores in a community increases as the quality of the aquatic environment declines. Nearly all major consumer species are omnivorous to a degree (Darnell 1961), so populations are considered to be truly omnivorous only if they feed on plants and animals in nearly equal amounts or indiscriminately (Kendeigh 1974). Recall that Karr and Schlosser used 25 percent of plant material ingested as the level for distinguishing between omnivores and other trophic guilds. Presumably, changes in the food base due to pollutional stress allow the euryphagic omnivores to become dominant because their opportunistic foraging ecology makes them more successful than more specific feeders. Omnivores are often virtually absent from unmodified streams. Even in moderately - altered streams omnivorous species usually constitute a minor portion of the community. For this reason, the biologist responsible for assessment must be familiar with the local fish fauna and aquatic habitats in order to be able to interpret subtle disproportions in trophic structure. In general, Karr (1981) has found samples with fewer than 20 percent of individuals as omnivores to be representative of good environmental quality, while those with greater than 45 percent omnivores represent badly degraded sites.

Karr (1981) reported that a strong inverse correlation exists between the abundance of insectivorous cyprinids and omnivores. Thus, communities containing a large proportion of insectivorous members of the minnow family (>45%) tends to indicate relatively high environmental quality.

TABLE IV-5-2. GENERAL CHARACTERISTICS OF RUNNING WATER ECOSYSTEMS ACCORDING TO SIZE OF STREAM. (From Karr and Dudley 1978, modified from Cummins 1975)

Stream size	Primary energy source	Production (trophic) state	Light and temperature regimes	Trophic status of dominant	
				Insects	Fish
*Small headwater streams (stream order 1-3)	Coarse particulate organic matter (CPOM) from the terrestrial environment Little primary production	Heterotrophic P/R <1	Heavily shaded Stable temperatures	Shredders Collectors	Invertivores
*Medium sized streams (4-6)	Fine particulate organic matter (FPOM), mostly Considerable primary production	Autotrophic P/R >1	Little shading High daily temperature variation	Collectors Scrapers (grazers)	Invertivores Piscivores
*Large rivers (7-12)	FPOM from upstream	Heterotrophic P/R <1	Little shading Stable temperatures	Planktonic collectors	Planktivores

* Streams are typically subdivided into these three size classes based on the stream order classification system of Kuehne (1962).

Faucsh et al. (unpublished manuscript) investigated the regional applicability of the IBI. Results from the two least disturbed watersheds in the study -- the Embaras River, Illinois and the Red River, Kentucky -- confirmed the fixed scoring criteria proposed by Karr (1981) for omnivores and insectivorous cyprinids. At most of the undisturbed sites in each stream, omnivores constituted 20 percent or less of all individuals and at least 45 percent of individuals were insectivorous cyprinids.

The presence of viable, vigorous populations of top carnivores is another indicator of a relatively healthy, trophically diverse community used in Karr's index. As described earlier, top carnivores constitute the peak of the ecological pyramid, and, therefore, occupy the highest trophic level in that particular community. Degradation of environmental quality causes top carnivore populations to decline and disappear. Theoretically, since top carnivore populations are supported (directly or indirectly) by all of the other (lower) trophic levels, they serve as a natural monitor of the overall health of the community. Because of their position atop the food chain, terminal carnivores are most vulnerable to detrimental effects of biomagnified toxicants. Also, predation by top carnivores keeps the populations of forage and rough fish in check, thereby functioning to maintain biotic integrity. As always, it is assumed that the project biologist will use considerable personal knowledge of local ichthyology and ecology in adjusting expectations of top carnivore species to stream size. The top carnivore populations must be evaluated in relation to what would be there if the habitat were not modified. Defining the baseline is a major problem in any study of pollutional stress. In determining the baseline community, the biologist may rely on the faunas of similar, unaltered habitats in the area, literature information, and personal experience -- remembering the concepts of the river continuum model.

The results of research conducted throughout the midwest tend to support the theoretical basis of the omnivore and top carnivore metric approaches to assessing biotic integrity (Larimore and Smith 1963, Cross and Collins 1975, Menzel and Fierstine 1976, Karr and Dudley 1978, Schlosser 1982a, Karr et al. 1983). Fausch et al. (unpublished manuscript) evaluated five watersheds in Illinois, Michigan, Kentucky, Nebraska, and North and South Dakota using the IBI, and found that scores accurately reflected watershed and stream conditions.

However, experts in the field recognize that the omnivore - top carnivore analysis may not be applicable in every situation on a nationwide basis. Reservations over use of this approach seem to be based on three variables.

(1) Type of pollutional stress - e.g., the trophic metrics proposed by Karr (1981) were largely derived from agricultural watersheds in which sedimentation and nutrient enrichment are the predominant forms of anthropogenic stress; other pollution problems such as toxic waste discharge could conceivably have a different impact on fish trophic structure.

(2) Type of aquatic habitat - e.g., headwater streams, large rivers, and flowing swamps represent very different environments which are characterized by a variety of trophic pathways and food sources.

(3) Type of ambient fish fauna - e.g., no or very tolerant top carnivores might be present naturally, or no or very intolerant omnivores.

LIST OF OMNIVORES AND TOP CARNIVORES

Examples of resident omnivore and top carnivore fish species are listed nationally in Appendices B-1 and B-2, respectively. These tables were compiled based on information found in the literature (Morita, 1963; Carlander, 1969, 1977; Pflieger, 1975; Moyle, 1976; Timbol and Maciolek, 1978; Smith, 1979; Morrow, 1980; Lee et al.,1980; Karr et al., 1983). The purpose of the lists is to provide a framework for assessing omnivore and top carnivore populations. However, because of the geographic variability in feeding habits, the gaps in available foraging data, and the dynamic nature of range boundaries, some members of the list may not occupy the specified trophic compartment in a particular area, while other species that belong on the list may have been overlooked. The list is intended to be used by knowledgeable biologists who are capable of adding and deleting species where necessary to produce a list which is appropriate for the particular area of study.

CHAPTER IV-6
REFERENCE SITES

Introduction

The goal of this section is to suggest an objective, ecological approach that should aid States in determining the ecological potential of priority aquatic ecosystems, evaluating and refining standards, prioritizing ecosystems for improvements, and comprehensively evaluating the ecological quality of aquatic ecosystems. The objectives of this section are to demonstrate the need for regional reference sites and to demonstrate how they can be determined. To do this the need for some type of control or reference sites will be discussed and alternate types will be outlined, the concept of ecological regions and methods for determining them will be described, aspects that should be considered when selecting reference sites will be listed, and the limitations of the regionalization method will be discussed.

Although correlation between a disturbance and the resulting functional or structural disorder can stimulate considerable insight, the disorder that results from disturbing a water body can be demonstrated scientifically only by comparing it with control or reference sites. To scientifically test for functional or structural disorder, data must be collected when the disturbances are present and when the disturbances are absent but everything else is the same. Disorders that are unique to the disturbed areas must be related to the disturbances but separated from natural variability. This requires carefully selected reference sites, but it is difficult or impossible to find pristine control or reference sites in most of the conterminous United States. Also, it is unlikely that pristine reference sites would be appropriate for most disturbed sites because they would differ in ways besides the distrubance, as will be discussed later.

The most commonly used reference sites are upstream and downstream of the recovery zone of a point source. However, these sites provide little value where diffuse pollution is a problem, where channel modifications are extensive, where point sources occur all along the stream, where the stream's morphology or flow changes considerably among sites, or where various combinations of these disturbances occur. Hughes et al. (1983) suggest a different approach, which reduces the problems of upstream-downstream reference sites. Their approach is based on first determining large, relatively-homogeneous, ecological regions (areas with similar land-surface form, climate, vegetation, etc.) followed by selection of a series of reference sites within each region. These sites could possibly serve as references for a number of polluted sites on a number of streams thereby economizing on and simplifying concurrent or future studies. A modification of Hughes et al.'s approach has been tested on two polluted streams in Montana (Hughes MS) and the approach is being rigorously tested on 110 sites in Ohio (Omernik and Hughes 1983).

The logical basis for Omernik and Hughes' approach was developed from ailey (1976), Green (1979), Hall et al. (1978), and arren (1979). Their logic fits well with the proposed water quality standards regulation (Federal Register 1982) that suggests grouping of streams wherever possible. Bailey stressed that heterogeneous lands, such as those managed by the U.S. Forest Service, must be hierarchically classified by their capabilities. He added that classification should be objective, synthesized from present mapped knowledge, and based on the spatial relationships of several environmental characteristics rather than on one characteristic or on the similarity of the characteristics alone.

One of Green's ten principles for optimizing environmental assessments is that wherever there are broad environmental patterns, the area should be broken into relatively homogeneous subareas. Clearly, this principle applies to most States. Hall et al. found that studies that incorporate several variously-impacted sites were more useful than separate intensive studies of one or two sites and more practical than long-term pre- and post- impact studies.

Warren proposed that a watershed/stream classification should integrate climate, topography, substrate, biota, and culture at all levels, as opposed to considering them separately. He also stated that the integration and classification should be hierarchical and be determined from the potentials of the lands and waters of interest, rather than from their present conditions. Streams within Warren's proposed classification would have increasingly similar ecological potentials as one moved down through the hierarchy to ever smaller watersheds or ecological regions.

The Concept of Ecological Regions

The ecological potential of a reference or disturbed site is considered to be the range of ecological conditions present in a number of typical, but relatively-undisturbed sites within an ecological region. Such relatively-undisturbed sites, can be found even in the channelized streams of the Midwest Corn Belt (Marsh and Luey 1982). One should not suppose that such sites represent pristine or undisturbed controls, only that they are the best that exist given the prevalent land use patterns in an ecological region. Because of the major economic and political strains required, we do not believe that resource managers or even knowledgeable and concerned citizens will change those general land use patterns much. But such persons will need to know the best conditions they can expect in a water body in order to decide whether the economic and noneconomic benefits of a particular water body standard are worth their economic and noneconomic costs. To make such determinations rationally, the reference sites must also be typical of a region. That is, their watersheds must wholly reflect the predominant climate, land-surface form, soil, potential natural vegetation, land use, and other environmental characteristics defining that region, and the site itself must contain no anomalous feature. For example, a cobble-bottomed stream in an entirely forested, highly dissected watershed would not be typical of the sand and gravel-bottomed streams in the agricultural prairies of the Midwest, nor could it be a useful predictor of such an agricultural stream's ecological potential, even though such a watershed and stream might be found in such a region.

Although all aquatic ecosystems differ to some degree, the basis of ecological regions is that there also is considerable similarity among aquatic ecosystem characteristics and that these similarities occur in definable geographic patterns. Also, the variabilities in the present and potential conditions of the chemical and physical environment and the biota are believed to be less within an area than among different areas. For example, streams in the Appalachian Mountains, are more similar to each other than to those in the Corn Belt or those on the Atlantic Coastal Plain. It is assumed that streams acquire their similarities from similarities in their watersheds and that streams draining watersheds with similar characteristics will be more similar to each other than to those draining watersheds with dissimilar characteristics. Thus, an ecological region is defined as a large area where the homogeneity in climate, land-surface form, soil, vegetation, land use, and other environmental characteristics is sufficient to produce relative homogeneity in stream ecosystems.

The concept of an ecological region is an out-growth of the work of vegetation ecologists, climatologists, physiographers, and soil taxonomists, all of whom have sought to display national patterns by mapping classes of individual environmental characteristics (USDI - Geological Survey 1970). James (1952) discusses the value of integrating or regionalizing such environmental characteristics and Warren (1979) provides an excellent rationale for classifying ecological regions, but Bailey's ecoregion map (1976) comes the closest to actually doing so. However, Bailey's map incorporates a hierarchical approach, concentrating on an individual environmental characteristic at each level, and does not yet incorporate land-surface form or land use. Hughes and Omernik (1981b) agree with Warren that it is most useful to integrate these features at every level in the hierarchy of ecological regions. Such an approach facilitates the mapping of ecological regions at a national, state, or county level with increasing resolution (but decreasing generality) at each lower level.

Ecological regions should improve States' abilities to manage aquatic ecosystems in at least four ways (Hughes and Omernik 1981b): (1) They should provide ecologically-meaningful management units. Such units allow objective and logical synthesis of existing data from ecologically-similar aquatic ecosystems and, using that synthesis, extrapolation to other unstudied ecosystems in the same ecological region. (2) They should provide an objective, ecological basis to refine use classifications and to evaluate the attainment of uses for aquatic ecosystems. This is because they provide an ecological basis for determining typical and potential states of aquatic ecosystems located in similar watersheds. (3) They should provide an objective ecological basis to prioritize aquatic ecosystems for improvements or for attainability analyses. Given knowledge of the typical and potential conditions of aquatic ecosystems in the separate ecological regions of a State, that State can rationally determine what to expect from improvements and thereby know where it will get the greatest ecological returns for its investments. (4) They should simplify setting site-specific criteria on site-specific biota, as allowed by the proposed water quality regulation. Rather than set separate criteria for a large number of sites at enormous expense, a State could use criteria obtained from a series of sites that typify potential conditions in each ecological region of that state or neighboring states.

The process of selecting reference sites can be broken into two major phases with most of the work done in an office. First, the ecological regions, and most-typical area(s) of interest are determined. Second, various sizes of candidate watersheds and reaches are evaluated for typicalness and level of disturbance in order to select reference sites.

Determining Ecological Regions

There are several methods for determining ecological regions. Trautman (1981) suggested that one factor, physiography, could be used to determine patterns of stream types and fish assemblages in Ohio. Lotspeich and Platts (1982) believed regions should be determined from two factors, climate and geology. Bailey (1976) used three factors, climate, soil, and potential natural vegetation, in his ecoregion map of the United States but suggested adding land-surface form and lithology if smaller ecoregions are mapped. Warren (1979) proposed that five factors, climate, topography, substrate, biota and culture, should all be incorporated in watershed classification. Hughes and Omernik (1981b), Omernik et al. (1982), and Omernik and Hughes (1983) overlaid maps of land-surface form, soil suborders, land use, and potential natural vegetation in studies of the Corn Belt and Ohio, but suggest using precipitation, temperature, and lithology if major differences in these factors are suspected. Lotspeich and Platts, Bailey, and Warren all emphasized the use of hierarchical ecoregions, moving from broad national regions thousands of square kilometers in size to small watersheds a few square kilometers in area. A much different approach to determining ecological regions is the stream habitat classification of Pflieger et al. (1981). They used cluster analysis of fish collections from throughout Missouri to group localities having similar fish faunas. Where States have computerized fish collection data from a thousand or more sites, cluster analysis is a useful approach, however only a handful of States have such data.

Because of the diversity of methods for determining ecological regions, the limited testing of their applicability to aquatic ecosystems, and the limited number of large computerized data files, States are encouraged to select a method that allows the greatest potential for later modification. The method of Hughes and Omernik requires no prior collection data and appears to allow more modification than the others. The greater number of characteristics used to determine regions increases the opportunity that those regions will have a variety of uses by several agencies and greater value in predicting impacts of managment actions. Therefore, their method is outlined by the following steps:

1. Select the area and aquatic characteristics of interest. In many cases the area of interest will be a State, but wherever major environmental characteristics or watersheds do not coincide with state borders, States may find it useful and economical to work cooperatively and incorporate portions of neighboring States. Aquatic characteristics of interest may include fish and macro-invertebrate assemblages and various aspects of the chemical and physical environment affecting those assemblages.

2. Select broad environmental characteristics most likely to control the aquatic characteristics of interest. Environmental characteristics to consider are climate (especially mean annual precipitation and summer and winter temperature extremes), land-surface form (types of plains, hills, or mountains), surficial geology (types of soil parent material), soils (whether wet or dry, hot or cold, shallow or deep, or low or high in nutrients), potential natural vegetation (grassland, shrubland, or forestland, and dominant species), major river basins (especially important in unglaciated areas for limiting fish and mollusk distribution), and land use (especially cropland, grazing land, forest, or various mixes of these). National maps of most of these characteristics are available in USDI-Geological Survey (1970), but, often, larger-scale State maps can be obtained from State agencies or university departments.

3. Examine maps of the selected environmental characteristics for classes of characteristics that occur in regional patterns. When original maps differ in scale or when finer resolution is required, a mechanical enlarger/reducer, photocopy machine, photo-enlarger, or slide projector can be used to produce maps of the desired scale. Select those classes of characteristics that best represent tentative ecological regions. For example, is the predominant class of land-surface form flat plains or high hills; is the predominant potential natural vegetation oak forest or ash forest? List the predominant class of all the characteristics considered for each tentative ecological region.

4. Overlay the selected environmental characteristics mapped at the same scale and outline the most-typical areas in each tentative ecological region. The maps are examined in combination on a light table and lines are drawn on a sheet of clear plastic or transparent paper (e.g. albanene). Most-typical areas are those areas in each tentative ecological region where all the predominant classes of environmental characteristics in that region are present. These can be considered as most-typical areas because they contain all the classes of characteristics that will be used to determine that ecological region. For example, if the predominant classes of land use, potential natural vegetation, and land-surface form in an ecological region are cropland, grassland, and plains, respectively, only the portion of that region where cropland, grassland, and plains all occur together would be most-typical. This overlay approach and some of the environmental characteristics are similar to those used by McHarg (1969) in his examination of the values of various land uses in the Potomac River Basin.

5. Determine which environmental characteristics best distinguish between regions. Where the major characteristics abruptly differ at the same place (e.g. hilly forestlands vs. prairie croplands) this is easily done, but where there are gradual transitions (e.g. from flat to smooth and irregular plains with decreasing amounts of croplands and increasing forestlands) it is more difficult and the boundries are less precise. At one boundary the distinguising characteristic may be land-surface form and surficial geology, at another it may be land use or a river

basin divide. Thus, this boundary determination is a subjective - not a mechanical or McHargian - process and it requires considerable judgment and knowledge of the key environmental characteristics along the tentative boundary. See Figure IV-7-1 for an example of a final product. Fianlly, the regional lines are transferred to a base map of the area of interest. On a State level, most of this work should be done using map scales of 1:500,000 to 1:7,500,000. The base map should then be circulated among knowledgeable professionals to evaluate the significance of the ecological regions as drawn.

For cases where top-priority aquatic ecosystems are anomalies, or where the State is interested in only a few sites, it may be more appropriate to use a slightly different approach based only on the watershed characteristics of the sites in question. For such cases, rather than analyze the entire State, researchers can determine the climate, land-surface form, soils, potential natural vegetation, land use, river basin, etc. of the watershed upstream of the site of interest. The same classes of characteristics elsewhere in the State or neighboring States can then be determined from maps. The rest of the regionalization process is the same as described above. The major difference in this approach is that, because of the spatially-narrower objective, fewer ecological regions will be determined, consequently, the product would have only local application.

Determining Candidate References Reaches

The most-typical areas are considered the most-logical places to locate reference reaches for several reasons: (1) Such areas should contain a narrower range of land use or disturbance potentials compared to the entire region or other regions. Hence, there should be a narrower range of aquatic ecosystem conditions in these most-typical areas compared to the entire region or other regions. (2) Such areas are more likely to be free of major anomalies that might produce undisturbed sites that are also atypical, such as an entirely forested, mountainous watershed in a region typified by shrublands and plains. (3) Such areas can potentially represent the greatest number of streams in the ecological region because they drain watersheds having all the predominant classes of environmental characteristics that were used to identify the region. (4) Such areas best represent the prevailing land use of the ecological region and the best background conditions likely. For example, there is little likelihood of transforming an area dominated by rangeland into forestland, therefore, the predominant land use in the watershed of a reference reach in such an area should be grazing.

For the above reasons, if watersheds of reference or benchmark reaches are to have the broadest possible applicability, they should fall entirely within the most-typical areas of ecological regions. Thus, the size of the most-typical area will determine the maximum size of such watersheds. The smallest watersheds should include the smallest intermittent or permanent streams and ponds that support spawning or rearing or valued populations. Valued populations may include sport, commercial, rare, threatened, endangered, forage, or intolerant species of any phylum.

Refining the Number of Candidate Reference Reaches

Regardless of how candidates for reference watersheds are determined there are several important aspects to consider when selecting reference reaches:

1. Human Disturbances. Obviously, watersheds that contain dense human populations, concentrations of mines or industry, several or important point sources, or major and atypical problems with diffuse pollution (e.g. acidification, soil erosion, overgrazing, mine wastes, landslides) should be eliminated from consideration as reference watersheds. Intentional stocking of sport fishes and incidental releases of aquarium and bait organisms have extended the ranges of many aquatic species. If these introductions are only local, knowledge of such populations should be considered when selecting least-disturbed watersheds because introduced stocks of species are one of the most detrimental changes that humans initiate in aquatic ecosystems. Where human disturbances are mapped this step should be done for the entire State.

2. Size: Because of the gradual change in many stream characteristics from headwaters to rivers (Vannote et al. 1980), plus application of MacArthur and Wilson's (1967) theory of island biogeography to lakes (Barbour and Brown 1974), it is important to consider the size of the reference reaches when they are to be compared with a priority water body. Although stream order (Strahler 1957) has often been used by biologists to approximate stream size, Hughes and Omernik (1981a, 1983) give several reasons why watershed area and mean annual discharge are preferable measures. Limnologists typically use surface area and volume to estimate lake size. Although regional differences make any generalizations difficult, the stream order of priority and reference reaches should not differ by more than one order in most cases and the watershed areas usually should differ by less than one order of magnitude.

3. Surface water hydrology. While determining size, the researcher should also briefly examine the types of the watersheds, streams, or lakes for anomalies. Large scale topographic maps will usually reveal whether the streams are effluent or influent, i.e., whether the net movement of water if from the streams to the ground water or the reverse. The same maps reveal drainage lakes, lake type (kettle, solution, oxbow, etc.), amount of ditching or canalization, and drainage pattern (dendritic, trellis, aimless, etc.).

4. Refugia. Parks, monuments, wildlife refuges, natural areas, preserves, state and federal forests, and woodlots are often indicated on large scale topographic maps and locations of others can be obtained from state agencies charged with their administration. Such refugia are often excellent places to locate reference sites and reference watersheds.

5. Groundwater hydrology. Reports from the State water resource agency and the State office of the U.S. Geological Survey reveal whether lakes are influent or effluent. The direction of water movement in lakes is extremely important in determining their nutrient balance, causes of eutrophication, and possible results of lake restoration efforts. For example, in shallow effluent lakes with small watersheds the major source of nutrients is the atmosphere and hence uncontrollable.

6. Runoff per unit area. This is extremely important in estimating stream size. The summarized runoff data are published in U.S. Geological Survey reports for each State. These data can be used to estimate isolines of runoff per unit area or existing runoff maps produced by State water resource agencies can be used. For a national example, see USDI - Geological Survey (1970).

7. Water chemistry. These data can be used to estimate background or typical conditions. Most are not summarized, but they can be located using NAWDEX and are available from computerized data bases such as WATSTORE and STORET and from State water reports of the U.S. Geological Survey and State water resource agencies.

8. Geoclimatic history. The historical geomorphology and climate determine the basin divides and historical connections among water bodies and basins. The absence of such connections and the locations of basin divides and major gradient changes determine centers of origin or endemism. Regionally, continental glaciation, ocean subsidence, and pluvial flooding, and locally, stream capture, canals, and headwater flooding all provided passages across apparent barriers that allowed range extension, and, in large part, determine the present ranges of primary freshwater fish and mollusks. This information is usually available from university geology departments and often from the state geologist.

9. Known zoogeographic patterns. These are best revealed by maps in books and articles on the biota of the state, e.g. Smith (1983), Trautman (1981), or Pflieger (1975). Such patterns may also be predicted by present river basins where the basin divides are substantial and the river mouths distant.

After considering the broad watershed and regional aspects of the candidate watersheds, the highly-degraded or unusual watersheds should be easily rejected. Candidate reaches can then be selected and ranked or clustered by expected level of disturbance. At this level of resolution, the researcher should study air photo mosaics and large-scale (1:24,000-1:250,000) maps of the candidate reaches. Stream gradient, distance from other refugia, barriers (falls, dams) between reference reaches and other refugia, distance from the major receiving water, number of mines, and buildings, amount of channelization, and presence of established monitoring or gaging sites should all be considered. The list of candidate reaches should be distributed to other professionals to query them about their knowledge of disturbance levels, previous or concurrent studies, fish stocking schedules, fish catch per unit effort, spawning or hatching pulses, valued species, etc.

Selecting Actual Reference Sites

All the preceding research can, and should, be done in an office. It is then useful to view and photograph the reduced number of candidate reaches from the air. A small wing-over airplane flying 300-1500 meters above the ground is ideal for this or recent stereo pairs of air photos can suffice. The candidate reach should be examined at several access points to assess typical and least-disturbed conditions, i.e., the absence of farm yards, feed lots, livestock grazing, irrigation diversions, row crops, channelization, mines, housing developments, clearcuts, or other small scale disturbances should be rejected, though the candidate reaches may be moved upstream of them. The main reasons for this aerial view are to determine what the candidate watersheds and reaches typically look like, to characterize relatively undisturbed conditions, and to help select actual reference sites. The photographs are also useful as visual aids in briefings and public meetings. This phase is not essential if the chief state ecologist has developed this knowledge of present conditions through years of experience statewide.

Finally, the remaining candidate reaches can be assessed and ranked for disturbance from the ground. Three to four candidate reference sites in each reach should be examined for typical natural features, least-disturbed channel and riparian characteristics, and ease of access. The concept of typicalness of natural features is similar to that of typicalness of watershed features; for example, riffle-pool morphology and swift current would not be typical of coastal plain or swamp streams and such anomolous sites should not be included as reference sites.

One of the best indicators of least-disturbed sites is extensive, old, riparian forest (see Section II-6). Another is relatively-high heterogeneity in channel width and depth (shallow riffles, deep pools, runs, secondary channels, flooded backwaters, sand bars, etc.). Abundant large woody debris (snags, root wads, log jams, brush piles), coarse bottom substrate (gravel, cobble, boulders), overhanging vegetation, undercut banks, and aquatic vascular macrophytes and additional substrate heterogeneity and concealment for biota. Relatively high discharges; clear, colorless, and odorless waters; visually-abundant diatom, insect, and fish assemblages; and the presence of beavers and piscivorous birds also indicate relatively-undisturbed sites.

In order to confidently ascertain whether a designated biotic use of a priority aquatic ecosystem is attainable it is necessary to (1) clearly define that use in objective, measurable, biotic conditions and (2) examine those conditions in at least three least-disturbed reference sites. We have described a process to locate and rank a number of least-disturbed reference sites. However, there are several limitations to that approach. To date this process has only been tested on streams with watersheds less than 1600 km^2. Major lakes and rivers can be examined in the same manner, but a multistate or national analysis will be needed and greater allowances for variability in the level of disturbance and the degree of typicalness may be necessary because large ecosystems encompass more variability, they are more likely to receive major point sources, and they are rarer to begin with.

Where priority aquatic ecosystems are unique it will be more difficult to find reference sites. For example, if the priority system is a forested watershed with a high-gradient stream in Iowa, where such a system is rare, it would be necessary to seek reference sites in neighboring States. Where a stream passes through extremely dissimilar ecological regions, reference streams should do likewise. For example, the Yampa River of Northwestern Colorado passes from spruce-forested mountains through sagebrush tablelands and should not be compared with a river that flows through only one of those regions.

Stream reaches above barriers, such as the falls on the Cumberland River or the relatively steep gradients of the Watauga River at the North Carolina-Tennessee border, should not be compared with those below because few purely aquatic species have passed those historical barriers. Streams that had glacial or pluvial connections (such as the Susquehanna and James Rivers) may have more species in common than neighboring rivers of either, the neighboring rivers have similar environmental conditions. Gilbert (1980) provides a clear discussion of these possible zoogeographic anomalies using examples from the eastern United States. Decisions about reference sites must also take such knowledge into consideration.

Finally, ecological regions and reference sites as described herein are believed most useful for making comparisons between broad assemblage-level patterns or patterns between widely-ranging and common species of importance, not between the presence or absence of specific uncommon or localized species viewed separately. That is, multivariate approaches such as ordination and classification or biotic indices such as Karr's (1981) are most applicable and researchers should not expect to discriminate among sites that vary only slightly.

Summary

The final product of this approach is a map like that of Figure IV-7-1. Data from the reference sites in each ecological region can be compared with those from disturbed sites in that region. For aquatic ecosystems that cross boundaries between ecological regions, state ecologists ought to examine data from the reference sites in those respective regions. Comparisons should be limited to ecosystems of similar size.

Rather than an ad hoc, best - biological judgment approach, a regionalization approach as described provides a rational, objective means to compare similarities and differences over large areas. The regions provide ecologically-meaningful management units and they would help in the organization and interpretation of State water quality and NPS reports. Data from the reference sites provide an objective, ecological basis to refine use classifications and, when compared with more disturbed sites, to evaluate the attainment of uses. Knowledge of potential conditions in a region provides an objective, ecological basis to predict effects of land use changes and pollution controls, to prioritize aquatic ecosystems for improvements, and to set site-specific criteria. Regular monitoring of the reference sites and comparisons with historical information will provide a useful assessment of temporal changes, not only in those aquatic ecosystems, but in the ecological regions that they model.

I NORTHWEST FLAT PLAINS

II WESTERN ROLLING PLAINS

III NE and SW IRREGULAR

IV DISSECTED SOUTHEAST

Most Typical Areas

Generally Typical Areas

Study Watersheds

° SECTION V: INTERPRETATION

CHAPTER V

INTERPRETATION

INTRODUCTION

There are many use classifications which might be assigned to a water body, such as navigation, recreation, water supply or the protection of aquatic life. These need not be mutually exclusive. The water body survey as discussed in this manual is concerned only with aquatic life uses and the protection of aquatic life in a water body.

The water body survey may also be referred to as a use attainability analysis. The objectives in conducting a water body survey are to identify:

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

2. What the causes are of any impairment to attaining the designated aquatic protection uses, and

3. What the aquatic protection uses are that could be attained, based on the physical, chemical and biological characteristics of the water body.

The types of analyses that might be employed to address these three points are summarized in Table V-1. Most of these are discussed in detail elsewhere in this manual.

CURRENT AQUATIC PROTECTION USES

The actual aquatic protection use of a water body is defined by the resident biota. The prevailing chemical and physical attributes will determine what biota may be present, but little need be known of these attributes to describe current uses. The raw findings of a biological survey may be subjected to various measurments and assessments, as discussed in Chapters IV-2, IV-4, and IV-5. After performing a biological inventory, omnivore-carnivore analysis, and intolerant species analysis, and calculating a diversity index and other indices of biological health, one should be able adequately to describe the condition of the aquatic life in the water body.

It will be helpful to digress at this juncture briefly to discuss water body use classification systems and their relationship to the water body survey. Classification systems vary widely from state to state. Some consist of as few as three broad categories, while others include a number of more sharply-defined categories. Also, the use classes may be based on geography, salinity, recreation, navigation, water supply (municipal, agricultural, or industrial), or aquatic life. Often an aquatic protection use must be categorized as either

TABLE V-1. SUMMARY OF TYPICAL WATER BODY EVALUATIONS (from EPA,1983, Water Quality Standards Handbook).

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

a warmwater or coldwater fishery. Clearly, little information is required to place a water body into one of these two categories. Far more information may be gathered in a water body survey than is needed to assign a classification, based on existing classes, but the additional data may be necessary to evaluate management alternatives and refine use classification systems for the protection of aquatic life in the water body.

Since there may not be a spectrum of aquatic protection use categories available against which to compare the findings of the biological survey; and since the objective of the survey is to compare existing uses with designated uses, and existing uses with potential uses, as seen in the three points listed above, the investigators may need to develop their own system of ranking the biological health of a water body (whether qualitative or quantitative) in order to satisfy the intent of the water body survey. Implicit to the water body survey is the development of management strategies or alternatives which might result in enhancement of the biological health of the water body. To do this it would be necessary to distinguish the predicted results of one strategy from another, where the strategies are defined in terms of aquatic life. The existing State use classifications will probably not be helpful at this stage, for one may very well be seeking to define use levels within an existing use category, rather than describing a shift from one use classification to another. To conclude, it may be helpful to develop an internal use classification system to serve as a yardstick during the course of the water body survey, which may later be referenced to the legally constituted use categories of the state. Sample scales of aquatic life classes are presented in Table V-2 and V-3.

CAUSES OF IMPAIRMENT OF AQUATIC PROTECTION USES

If the biological evaluations indicate that the biological health of the system is impaired relative to a "healthy" or least disturbed control station or reference aquatic ecosystem (e.g., as determined by reference reach comparisons), then the physical and chemical evaluations can be used to pinpoint the causes of that impairment. Figure V-1 shows some of the physical and chemical parameters that may be affected by various causes of change in a water body. The analysis of such parameters will help clarify the magnitude of impairments to attaining other uses, and will also be important to the third step in which potential uses are examined.

ATTAINABLE AQUATIC PROTECTION USES

The third element to be considered is the assessment of potential uses of the water body. This assessment would be based on the findings of the physical, chemical and biological information which has been gathered, but additional study may also be necessary. Procedures which might be particularly helpful in this stage include the Habitat Suitability Index Models of the Fish and Wildlife Service, that may indicate which fish species could potentially occupy a given habitat; and the Recovery Index of Cairns et al. (1977) which estimates the ability of a system to recover following stress. A reference reach comparison will be particularly important. In addition to establishing a comparative

TABLE V-2. BIOLOGICAL HEALTH CLASSES WHICH COULD BE USED IN WATER BODY ASSESSMENT (Modified from Karr, 1981)

Class	Attributes
Excellent	Comparable to the best situations unaltered by man; all regionally expected species for the habitat and stream size, including the most intolerant forms, are present with full array of age and sex clases; balanced trophic structure.
Good	Fish and macroinvertebrate species richness somewhat less than the best expected situation, especially due to loss of most intolerant forms; some species with less than optimal abundances or size distribution (fish); trophic structure shows some signs of stress.
Fair	Fewer intolerant forms of fish and macroinvertebrates are present. Trophic structure of the fish community is more skewed toward an increasing frequency of omnivores; older age classes of top carnivores may be rare.
Poor	Fish community is dominated by omnivores; pollution-tolerant forms and habitat generalists; few top carnivores; growth rates and condition factors commonly depressed; hybrids and diseased fish may be present. Tolerant macroinvertebrates are often abundant.
Very Poor	Few fish present, mostly introduced or very tolerant forms; hybrids common; disease, parasites, fin damage, and other anomalies regular. Only tolerant forms of macroinvertebrates are present.
Extremely Poor	No fish, very tolerant macroinvertebrates, or no aquatic life.

Table V-3: Aquatic Life Survey Rating System (EPA, 1983 Draft)

A reach that is rated a five has:

- -A fish community that is well balanced among the different levels of the food chain.
- -An age structure for the most species that is stable, neither progressive (leading to an increase in population) or regressive (leading to a decrease in population).
- -A sensitive sport fish species or species of special concern always present.
- -Habitat which will support all fish species at every stage of their life cycle.
- -Individuals that are reaching their potential for growth.
- -Fewer individuals of each species.
- -All available niches filled.

A reach that is rated a four has:

- -Many of the above characteristics but some of them are not exhibited to the full potential. For example, the reach has a well balanced fish community; the age structure is good, sensitive species are present; but the fish are not up to their full growth potential and may be present in higher numbers; an aspect of the habitat is less than perfect (i.e. occasional high temperatures that do not have an acute effect on the fish); and not all food organisms are available or they are available in fewer numbers.

A reach that is a three has:

- -A community is not well balanced, one or two tropic levels dominate.
- -The age structure for many species is not stable, exhibiting regressive or progressive charisteristics.
- -Total number of fish is high, but individuals are small.
- -A sensitive species may be present, but is not flourishing.
- -Other less sensitive species make up the majority of the biomass.
- -Anadromous sport fish infrequently use these water as a migration route.

A reach that is rated a two has:

- -Few sensitive sport fish are present, nonsport fish species are more common than sport fish species.
- -Species are more common than abundant.
- -Age structures may be very unstable for any species.
- -The composition of the fish population and dominate species is very changeble.
- -Anadromous fish rarely use these waters as a migration route.
- -A small percent of the reach provides sport fish habitat.

A reach that is a one has:

- -The ability to support only nonsport fish. A occasional sport fish may be found as a transient.

A reach that is rated a zero has:

- -No ability to support a fish of any sort, an occasional fish may be found as a transient.

SOURCE OF MODIFICATION

STREAM PARAMETER	Acid Mine Drainage or Acid Precipitation	Sewage Treatment Plant Discharge (primary or secondary)	Agricultural Runoff (pasture or cropland)	Urban Runoff	Channelization	(Industries) Pulp and Paper	Textile	Metal Finishing and Electroplating	Petroleum	Iron and Steel	Paint and Ink	Dairy and Meat Products	Fertilizer Production and Lime Crushing	Plastics and Synthetics
pH	D					C	I	C		D	C		D,I	C
Alkalinity	D						I						D,I	
Hardness	I						I						I	
Chlorides		I		I								I		
Sulfates	I								I	I			I	I
TDS	I						I		I		I		I	
TKN		I	I	I					I	I		I	I	I
NH_3-N		I							I	I			I	I
Total-P		I	I	I				I				I	I	
Ortho-P		I		I				I					I	
BOD_5		I				I	I		I	I	I	I		I
COD	I	I		I		I	I			I	I	I		I
TOC		I	I	I		I			I	I	I	I		I
COD/BOD_5	I			I		I	I		D	I				
D.O.		D				D						D		
Aromatic Compounds			I	I		I			I					I
Fluoride								I					I	I
Cr				I		I	I	I	I		I			I
Cu	I			I		I		I			I			I
Pb				I				I	I	I	I			
Zn	I			I		I		I	I	I	I			I
Cd		I		I				I			I			I
Fe	I			I				I		I	I		I	I
Cyanide										I				I
Oil and Grease						I	I	I	I	I	I	I		I
Coliforms	D	I	I	I				D		D	D	I		
Chlorophyll	D	I	I		I	D		D	D	D	D	I	I	D
Diversity	D	D		D	D	D	D	D	D	D	D			D
Biomass	D	I	I		I		I	D	D	D	D	I	I	D
Riparian Characteristics					C									
Temperature					I									
TSS			I	I	I	I	I	I			I		I	I
VSS				I		I								
Color						I	I				I	I		I
Conductivity	I												I	
Channel Characteristics					C									

Figure V-1. Potential Effects of Some Sources of Alteration on Stream Parameters; D = decrease, I = increase, C = change.

baseline community, defining a reference reach can also provide insight to the aquatic life that could potentially occur if the sources of impairment were mitigated.

The analysis of all information that has been assembled may lead to the definition of alternative strategies for the management of the water body at hand. Each such strategy corresponds to a unique level of protection of aquatic life, or aquatic life protection use. If it is determined that an array of uses are attainable, further analysis which is beyond the scope of the water body survey would be required to select a management program for the water body.

A number of factors which contribute to the health of the aquatic life will have been evaluated during the course of the water body survey. These may be divided into two groups: those which can be controlled or manipulated, and those which cannot. The factors which cannot be regulated may be attributable to natural phenemona or may be attributable to irrevocable anthropogenic (cultural) activities. The potential for enhancing the aquatic life of a water body essentially lies in those factors over which some control may be exerted.

Whether or not a factor can be controlled may itself be a subject of controversy for there may be a number of economic judgments or institutional considerations which are implicit to a definition of control. For example, there are many cases in the West where a wastewater discharge may be the only flow to what would otherwise be an intermittent stream. If water rights have been established for that discharge then the discharge cannot be diverted elsewhere, applied to the land for example, in order to reduce the pollutant load to the stream. If a stream does not support an anadromous fishery because of dams and diversions which have been built for water supply and recreational purposes, it is unlikely that a concensus could be reached to restore the fishery by removing the physical barriers - the dams - which impede the migration of fish. However, it may be practical to build fish ladders and by-passes to allow upstream and downstream migration. In a practical sense these dams represent anthropogenic activity which cannot be reversed. A third example might be a situation in which dredging to remove toxic sediments in a river may pose a much greater threat to aquatic life than to do nothing. In doing nothing the toxics may remain in the sediment in a biologically-unavailable form, whereas dredging might resuspend the toxic fraction, making it biologically available and also facilitating wider distribution in the water body.

The points touched upon above are presented to suggest some of the phenomena which may be of importance in a water body survey, and to suggest the need to recognize whether or not they may realistically be manipulated. Those which cannot be manipulated essentially define the limits of the highest potential use that might be realized in the water body. Those that can be manipulated define the levels of improvement that are attainable, ranging from the current aquatic life uses to those that are possible within the limitations imposed by factors that cannot be manipulated.

• SECTION VI: REFERENCES

CHAPTER VI

REFERENCES

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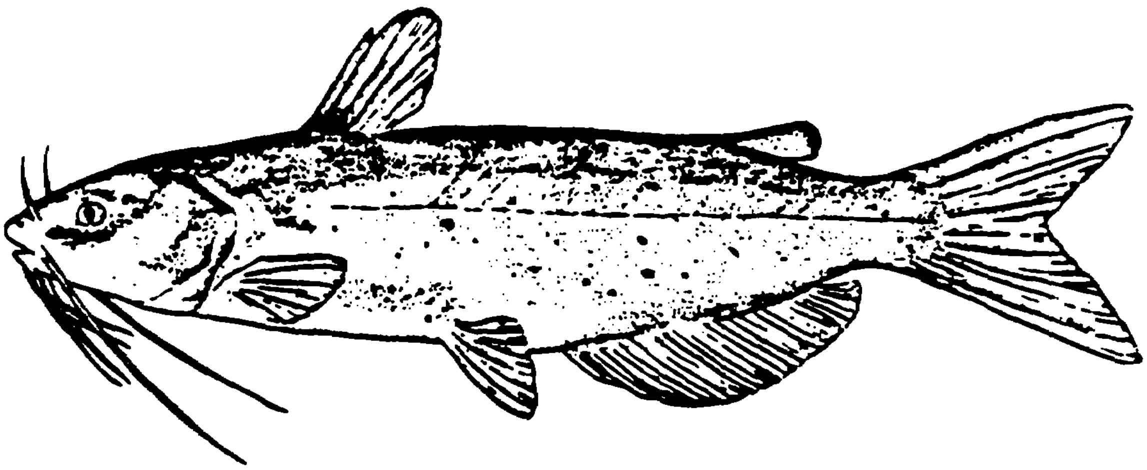
APPENDIX A-1:

SAMPLE HABITAT SUITABILITY INDEX
(Channel Catfish)

Biological Services Program

FWS/OBS-82/10.2
FEBRUARY 1982

HABITAT SUITABILITY INDEX MODELS: CHANNEL CATFISH



Fish and Wildlife Service

U.S. Department of the Interior

FWS/OBS-82/10.2
February 1982

HABITAT SUITABILITY INDEX MODELS: CHANNEL CATFISH

by

Thomas E. McMahon
and
James W. Terrell
Habitat Evaluation Procedures Group
Western Energy and Land Use Team
U.S. Fish and Wildlife Service
Drake Creekside Building One
2625 Redwing Road
Fort Collins, Colorado 80526

Western Energy and Land Use Team
Office of Biological Services
Fish and Wildlife Service
U.S. Department of the Interior
Washington, D.C. 20240

PREFACE

The habitat use information and Habitat Suitability Index (HSI) models presented in this document are an aid for impact assessment and habitat management activities. Literature concerning a species' habitat requirements and preferences is reviewed and then synthesized into HSI models, which are scaled to produce an index between 0 (unsuitable habitat) and 1 (optimal habitat). Assumptions used to transform habitat use information into these mathematical models are noted, and guidelines for model application are described. Any models found in the literature which may also be used to calculate an HSI are cited, and simplified HSI models, based on what the authors believe to be the most important habitat characteristics for this species, are presented.

Use of the models presented in this publication for impact assessment requires the setting of clear study objectives and may require modification of the models to meet those objectives. Methods for reducing model complexity and recommended measurement techniques for model variables are presented in Appendix A.

The HSI models presented herein are complex hypotheses of species-habitat relationships, <u>not</u> statements of proven cause and effect relationships. Results of model performance tests, when available, are referenced; however, models that have demonstrated reliability in specific situations may prove unreliable in others. For this reason, the FWS encourages model users to convey comments and suggestions that may help us increase the utility and effectiveness of this habitat-based approach to fish and wildlife planning. Please send comments to:

Habitat Evaluation Procedures Group
Western Energy and Land Use Team
U.S. Fish and Wildlife Service
2625 Redwing Road
Ft. Collins, CO 80526

CONTENTS

CHANNEL CATFISH (*Ictalurus punctatus*)

HABITAT USE INFORMATION

General

The native range of channel catfish (*Ictalurus punctatus*) extends from the southern portions of the Canadian prairie provinces south to the Gulf states, west to the Rocky Mountains, and east to the Appalachian Mountains (Trautman 1957; Miller 1966; Scott and Crossman 1973). They have been widely introduced outside this range and occur in essentially all of the Pacific and Atlantic drainages in the 48 contiguous states (Moore 1968; Scott and Crossman 1973). The greatest abundance of channel catfish generally occurs in the open (unleveed) floodplains of the Mississippi and Missouri River drainages (Walden 1964).

Age, Growth, and Food

Age at maturity in channel catfish is variable. Catfish from southern areas with longer growing seasons mature earlier and at smaller sizes than those from northern areas (Davis and Posey 1958; Scott and Crossman 1973). Southern catfish mature at age V or less (Scott and Crossman 1973; Pflieger 1975) while northern catfish mature at age VI or greater for males and at age VIII or greater for females (Starostka and Nelson 1974).

Young-of-the-year (age 0) catfish feed predominantly on plankton and aquatic insects (Bailey and Harrison 1948; Walburg 1975). Adults are opportunistic feeders with an extremely varied diet, including terrestrial and aquatic insects, detrital and plant material, crayfish, and molluscs (Bailey and Harrison 1948; Miller 1966; Starostka and Nelson 1974). Fish may form a major part of the diet of catfish > 50 cm in length (Starostka and Nelson 1974). Channel catfish diets in rivers and reservoirs do not appear to be significantly different (see Bailey and Harrison 1948; Starostka and Nelson 1974). Feeding is done by both vision and chemosenses (Davis 1959) and occurs primarily at night (Pflieger 1975). Bottom feeding is more characteristic but food is also taken throughout the water column (Scott and Crossman 1973). Additional information on the composition of adult and juvenile diets is provided in Leidy and Jenkins (1977).

Reproduction

Channel catfish spawn in late spring and early summer (generally late May through mid-July) when temperatures reach about 21° C (Clemens and Sneed 1957; Marzolf 1957; Pflieger 1975). Spawning requirements appear to be a major factor in determining habitat suitability for channel catfish (Clemens and Sneed 1957). Spawning is greatly inhibited if suitable nesting cover is unavailable (Marzolf 1957).

Specific Habitat Requirements

Channel catfish populations occur over a broad range of environmental conditions (Sigler and Miller 1963; Scott and Crossman 1973). Optimum riverine habitat is characterized by warm temperatures (Clemens and Sneed 1957; Andrews et al. 1972; Biesinger et al. 1979) and a diversity of velocities, depths, and structural features that provide cover and food (Bailey and Harrison 1948). Optimum lacustrine habitat is characterized by large surface area, warm temperatures, high productivity, low to moderate turbidity, and abundant cover (Davis 1959; Pflieger 1975).

Fry, juvenile, and adult channel catfish concentrate in the warmest sections of rivers and reservoirs (Ziebell 1973; Stauffer et al. 1975; McCall 1977). They strongly seek cover, but quantitative data on cover requirements of channel catfish in rivers and reservoirs are not available. Debris, logs, cavities, boulders, and cutbanks in lakes and in low velocity (< 15 cm/sec) areas of deep pools and backwaters of rivers will provide cover for channel catfish (Bailey and Harrison 1948). Cover consisting of boulders and debris in deep water is important as overwintering habitat (Miller 1966; Jester 1971; Cross and Collins 1975). Deep pools and littoral areas (≤ 5 m deep) with ≥ 40% suitable cover are assumed to be optimum. Turbidities > 25 ppm but < 100 ppm may somewhat moderate the need for fixed cover (Bryan et al. 1975).

Riffle and run areas with rubble substrate and pools (< 15 cm/sec) and areas with debris and aquatic vegetation are conditions associated with high production of aquatic insects (Hynes 1970) consumed by channel catfish in rivers (Bailey and Harrison 1948). Channel catfish are most abundant in river sections with a diversity of velocities and structural features. Therefore, it is assumed that a riverine habitat with 40-60% pools would be optimum for providing riffle habitat for food production and feeding and pool habitat for spawning and resting cover (Bailey and Harrison 1948). It also is assumed that at least 20% of lake or reservoir surface area should consist of littoral areas (≤ 5 m deep) to provide adequate area for spawning, fry and juvenile rearing, and feeding habitat for channel catfish.

High standing crops of warmwater fishes are associated with total dissolved solids (TDS) levels of 100 to 350 ppm for reservoirs in which the concentrations of carbonate-bicarbonate exceed those of sulfate-chloride (Jenkins 1976). It is assumed that high standing crops of channel catfish in lakes or reservoirs will, on the average, correspond to this TDS level.

Turbidity in rivers and reservoirs and reservoir size are other factors that may influence habitat suitability for channel catfish populations. Channel catfish are abundant in rivers and reservoirs with varying levels of turbidity and siltation (Cross and Collins 1975). However, low to moderate turbidities (< 100 ppm) are probably optimal for both survival and growth (Finnell and Jenkins 1954; Buck 1956; Marzolf 1957). Larger reservoirs (> 200 ha) are probably more suitable reservoir habitat for channel catfish populations because survival and growth are better than in smaller reservoirs (Finnell and Jenkins 1954; Marzolf 1957). Other factors that may affect reservoir habitat suitability for channel catfish are mean depth, storage ratio (SR), and length of agricultural growing season. Jenkins (1974) found that high mean depths were negatively correlated with standing crop of channel catfish. Mean depths are an inverse correlate of shoreline development (Ryder et al. 1974), thus higher mean depths may mean less littoral area would be available. Jenkins (1976) also reported that standing crops of catfishes (Ictaluridae) peaked at an SR of 0.75. Standing crops of channel catfish were postively correlated to growing season length (Jenkins 1970). However, harvest of channel catfish reported in reservoirs was not correlated with growing season length (Jenkins and Morais 1971).

Dissolved oxygen (DO) levels of 5 mg/l are adequate for growth and survival of channel catfish, but D.O. levels of ≥ 7 mg/l are optimum (Andrews et al. 1973; Carlson et al. 1974). Dissolved oxygen levels < 3 mg/l retard growth (Simco and Cross 1966), and feeding is reduced at D.O. levels < 5 mg/l (Randolph and Clemens 1976).

Adult. Adults in rivers are found in large, deep pools with cover. They move to riffles and runs at night to feed (McCammon 1956; Davis 1959; Pflieger 1971; 1975). Adults in reservoirs and lakes favor reefs and deep, protected areas with rocky substrates or other cover. They often move to the shoreline or tributaries at night to feed (Davis 1959; Jester 1971; Scott and Crossman 1973).

The optimal temperature range for growth of adult channel catfish is 26-29° C (Shrable et al. 1969; Chen 1976). Growth is poor at temperatures < 21° C (McCammon and LaFaunce 1961; Macklin and Soule 1964; Andrews and Stickney 1972) and ceases at < 18° C (Starostka and Nelson 1974). An upper lethal temperature of 33.5° C has been reported for catfish acclimated at 25° C (Carlander 1969).

Adult channel catfish were most abundant in habitats with salinities < 1.7 ppt in Louisiana, although they occurred in areas with salinities up to 11.4 ppt (Perry 1973). Salinities ≤ 8 ppt are tolerated with little or no effect, but growth slows above this level and does not occur at salinities > 11 ppt (Perry and Avault 1968).

Embryo. Dark and secluded areas are required for nesting (Marzolf 1957). Males build and guard nests in cavities, burrows, under rocks, and in other protected sites (Davis 1959; Pflieger 1975). Nests in large impoundments generally occur among rubble and boulders along protected shorelines at depths of about 2-4 m (Jester 1971). Catfish in large rivers are likely to move into shallow, flooded areas to spawn (Bryan et al. 1975). Lawler (1960) reported that spawning in Utah Lake, Utah, was concentrated in sections of the lake with abundant spawning sites of rocky outcrops, trees, and crevices. The male catfish fans embryos for water exchange and guards the nest from predators (Miller 1966; Minckley 1973). Embryos can develop in the temperature range of 15.5 to 29.5° C, with the optimum about 27° C (Brown 1942; Clemens and Sneed 1957). They do not develop at temperatures < 15.5° C (Brown 1942). Embryos hatch in 6-7 days at 27° C (Clemens and Sneed 1957).

Laboratory studies indicate that embryos three days old and older can tolerate salinities up to 16 ppt until hatching, when tolerance drops to 8 ppt (Allen and Avault 1970). However, 2 ppt salinity is the highest level in which successful spawning in ponds has been observed (Perry 1973). Embryo survival and production in reservoirs will probably be high in areas that are not subject to disturbance by heavy wave action or rapid water drawdown.

Fry. The optimal temperature range for growth of channel catfish fry is 29-30° C (West 1966). Some growth does occur down to temperatures of 18° C (Starostka and Nelson 1974), but growth generally is poor in cool waters with average summer temperatures < 21° C (McCammon and LaFaunce 1961; Macklin and

Soule 1964; Andrews et al. 1972) and in areas with short agricultural growing seasons (Starostka and Nelson 1974). Upper incipient lethal levels for fry are about 35-38° C, depending on acclimation temperature (Moss and Scott 1961; Allen and Strawn 1968). Optimum salinities for fry range from 0-5 ppt; salinities ≥ 10 ppt are marginal as growth is greatly reduced (Allen and Avault 1970).

Fry habitat suitability in reservoirs is related to flushing rate of reservoirs in midsummer. Walburg (1971) found abundance and survival of fry greatly decreased at flushing rates < 6 days in July and August.

Channel catfish fry have strong shelter-seeking tendencies (Brown et al. 1970), and cover availability will be important in determining habitat suitability. Newly hatched fry remain in the nest for 7-8 days (Marzolf 1957) and then disperse to shallow water areas with cover (Cross and Collins 1975). Fry are commonly found aggregated near cover in protected, slow-flowing (velocity < 15 cm/sec) areas of rocky riffles, debris-covered gravel, or sand bars in clear streams (Davis 1959; Cross and Collins 1975), and in very shallow (< 0.5 m) mud or sand substrate edges of flowing channels along turbid rivers and bayous (Bryan et al. 1975). Dense aquatic vegetation generally does not provide optimum cover because predation on fry by centrarchids is high under these conditions, especially in clear water (Marzolf 1957; Cross and Collins 1975). Fry overwinter under boulders in riffles (Miller 1966) or move to cover in deeper water (Cross and Collins 1975).

Juvenile. Optimal habitat for juveniles is assumed to be similar to that for fry. The temperature range most suitable for juvenile growth is reported to be 28-30° C (Andrews et al. 1972; Andrews and Stickney 1972). Upper lethal temperatures are assumed to be similar to those for fry.

HABITAT SUITABILITY INDEX (HSI) MODELS

Model Applicability

Geographic area. The model is applicable throughout the 48 conterminous States. The standard of comparison for each individual variable suitability index is the optimum value of the variable that occurs anywhere within the 48 conterminous States. Therefore, the model will never provide an HSI of 1.0 when applied to water bodies in the Northern States where temperature-related variables do not reach the optimum values for channel catfish found in the Southern States.

Season. The model provides a rating for a water body based on its ability to support a self-sustaining population of channel catfish through all seasons of the year.

Cover types. The model is applicable in riverine, lacustrine, palustrine, and estuarine habitats, as described by Cowardin et al. (1979).

Minimum habitat area. Minimum habitat area is defined as the minimum area of contiguous suitable habitat that is required for a species to succesfully live and reproduce. No attempt has been made to establish a minimum habitat size for channel catfish, although this species is most abundant in larger water bodies.

Verification level. The acceptable output of these models is an index between 0 and 1 which the authors believe has a positive relationship to carrying capacity. In order to verify that the model output was acceptable, sample data sets were developed for calculating HSI's from the models.

The sample data sets and their relationship to model verification are discussed in greater detail following the presentation of the models.

Model Description

It is assumed that channel catfish habitat quality is based primarily on their food, cover, water quality, and reproduction requirements. Variables that have been shown to have an impact on the growth, survival, distribution, abundance, or other measure of well-being of channel catfish are placed in the appropriate component and a component rating derived from the individual variable suitability indices (Figs. 1 and 2). Variables that affect habitat quality for channel catfish, but which do not easily fit into these four major components, are combined under the "other component" heading. Levels of a variable that are near lethal or result in no growth cannot be offset by other variables.

Model Description - Riverine

Food component. Percent cover (V_2) is assumed to be important for rating the food component because if cover is available, fish would be more likely to occupy an area and utilize the food resources. Substrate (V_4) is included because stream production potential of aquatic insects (consumed directly by both channel catfish and their prey species) is related to amount and type of substrate.

Cover component. Percent pools (V_1) is included because channel catfish utilize pools as cover. Percent cover (V_2) is an index of all types of objects, including logs and debris, used for cover in rivers. Average current velocity in cover areas (V_{18}) is important because the usable habitat near a cover object decreases if cover objects are surrounded by high velocities.

Water quality component. The water quality component is limited to temperature, oxygen, turbidity, and salinity measurements. These parameters have been shown to effect growth or survival, or have been correlated with changes in standing crop. Variables related to temperature, oxygen, and salinity are assumed to be limiting when they approach lethal levels. Toxic substances are not considered.

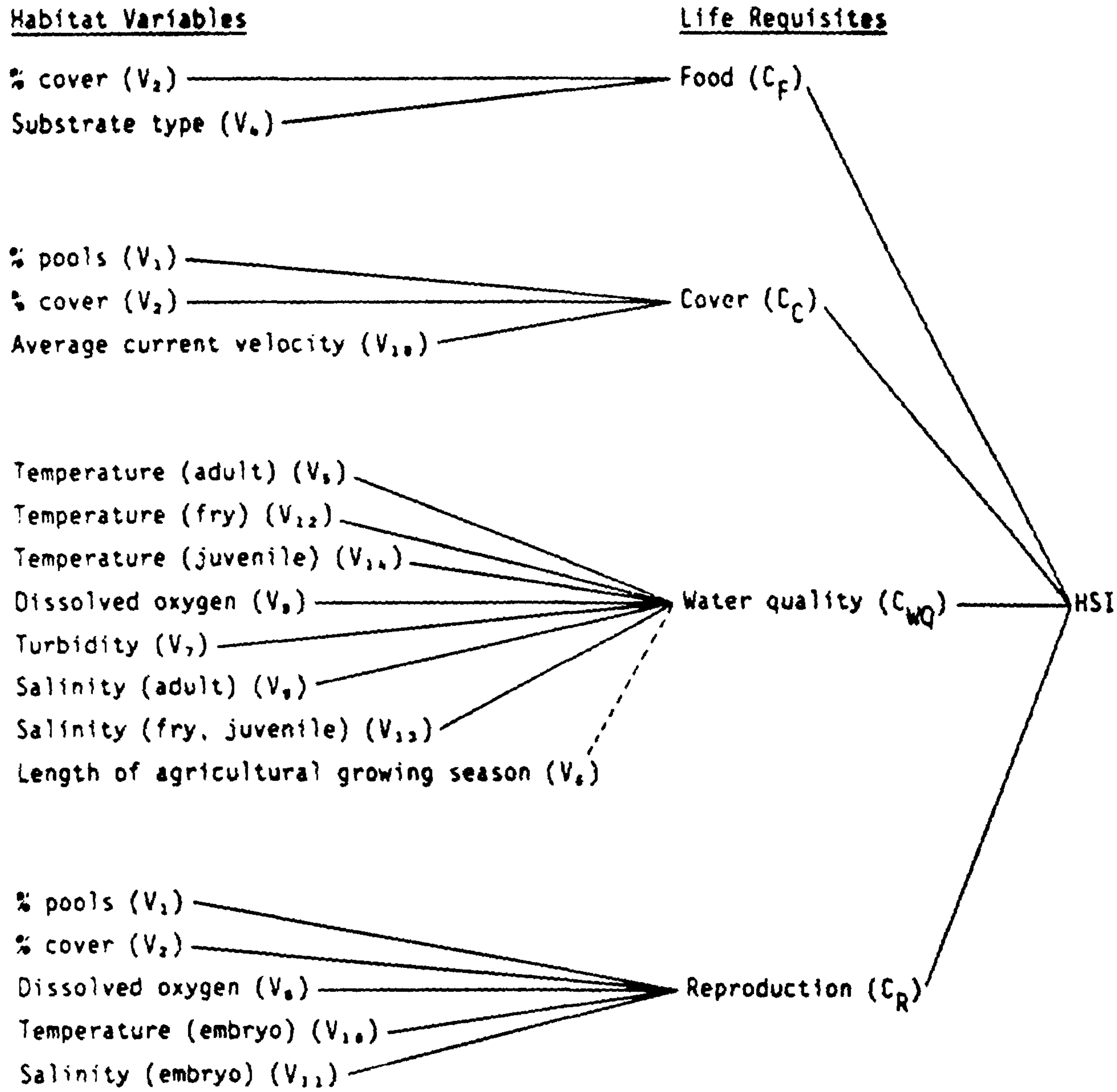


Figure 1. Tree diagram illustrating relationship of habitat variables and life requisites in the riverine model for the channel catfish. Dashed lines indicate optional variables in the model.

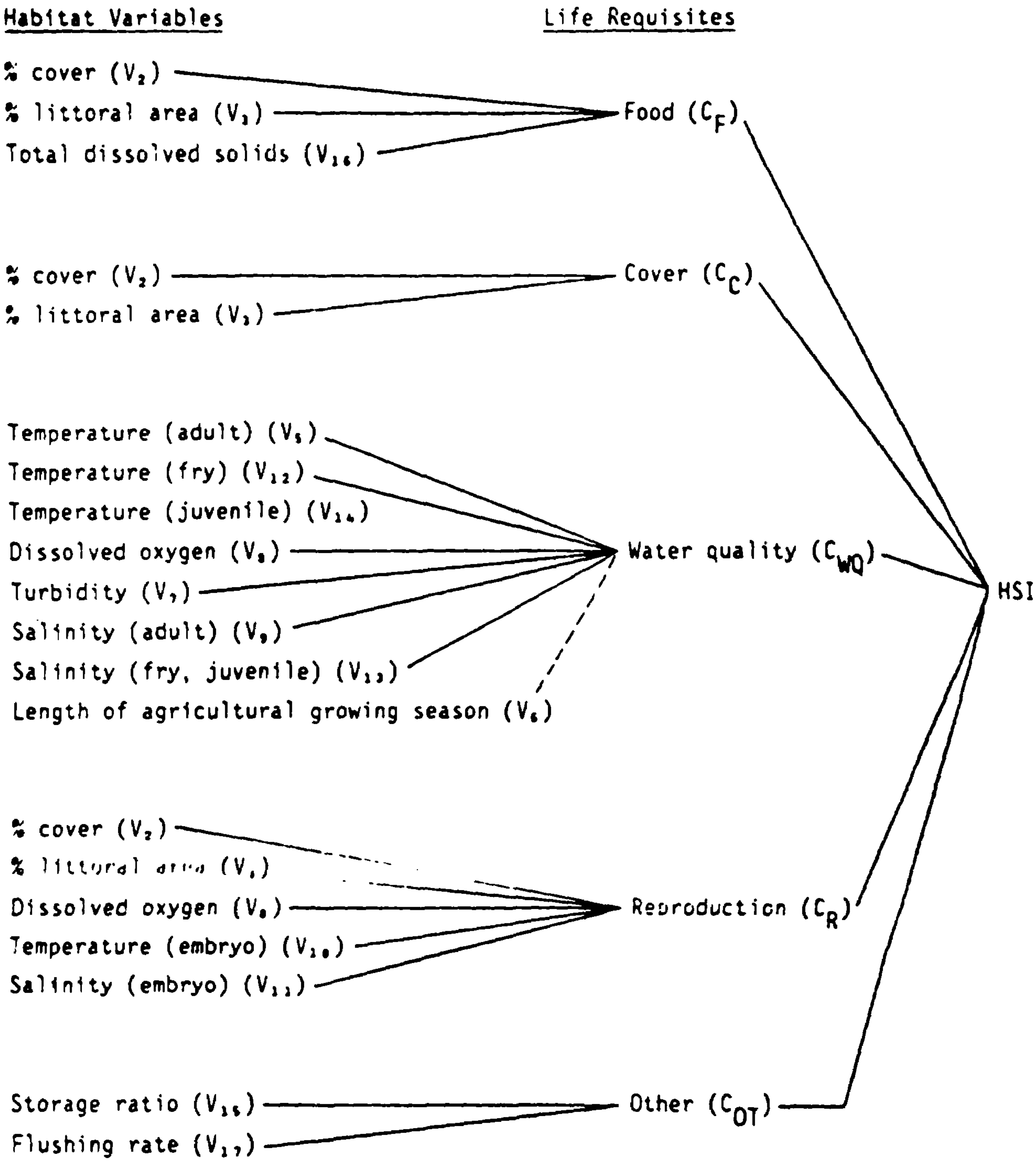


Figure 2. Tree diagram illustrating relationship of habitat variables and life requisites in the lacustrine model for the channel catfish. Dashed lines indicate optional variables in the model.

Reproduction component. Percent pools (V_1) is in the reproductive component because channel catfish spawn in low velocity areas in rivers. Percent cover (V_2) is in this component since channel catfish require cover for spawning. If minimum dissolved oxygen (DO) levels within pools and backwaters during midsummer (V_8) are adequate, they should be adequate during spawning, which occurs earlier in the year. DO levels measured during spawning and embryo development could be substituted for V_8. Two additional variables, average water temperatures within pools and backwaters during spawning and embryo development (V_{10}) and maximum salinity during spawning and embryo development (V_{11}) are included because these water quality conditions affect embryo survival and development.

Model Description - Lacustrine

Food component. Percent cover (V_2) is included since it is assumed that if cover is available, channel catfish would be more likely to utilize an area for feeding. Percent littoral area (V_3) is included because littoral areas generally produce the greatest amount of food and feeding habitat for catfish. Total dissolved solids (TDS) (V_{16}) is included because adult channel catfish eat fish, and fish production in lakes and reservoirs is correlated with TDS.

Cover component. Percent cover (V_2) is included since channel catfish strongly seek structural features of logs, debris, brush, and other objects for shelter. Percent littoral area (V_3) is included because all life stage predominantly utilize cover found in littoral areas of a lake.

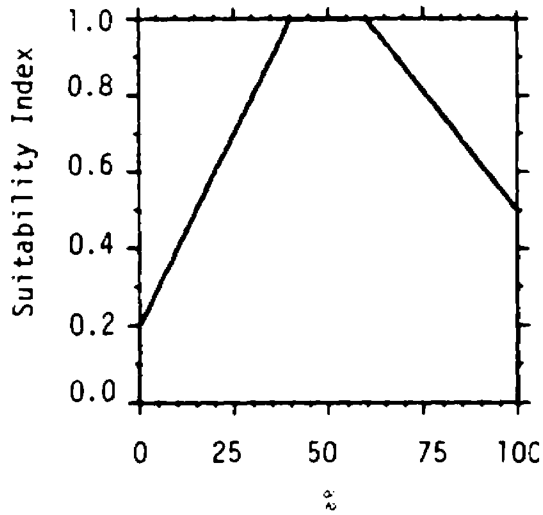
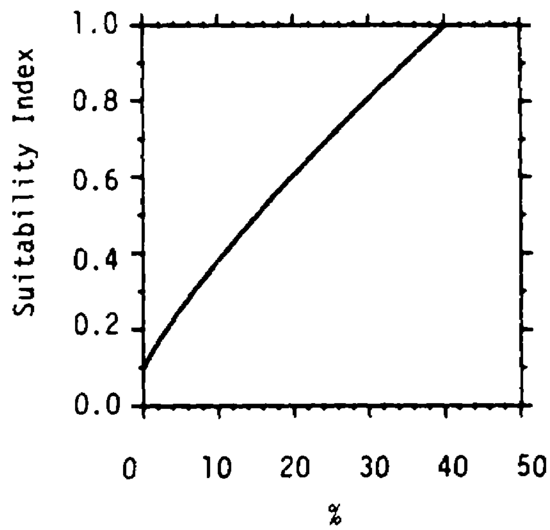
Water quality component. Refer to riverine model description.

Reproduction component. Percent cover (V_2) is included since catfish build nests in dark and secluded areas; spawning is not observed if suitable cover is unavailable. Percent littoral area (V_3) is included since catfish spawning is concentrated along the shoreline. DO (V_8), temperature (V_{10}) and salinity (V_{11}) are included because these water quality parameters affect embryo survival and development.

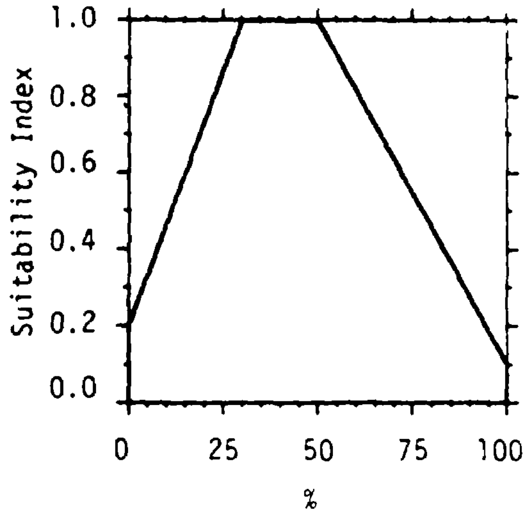
Other component. For reservoirs, storage ratio (V_{18}) and maximum flushing rate when fry are present (V_{17}) are included in this component because storage ratio may affect standing crop and the flushing of fry from a reservoir outlet can reduce the abundance of fry.

Suitability Index (SI) Graphs for Model Variables

This section contains suitability index graphs for the 18 variables described above, and equations for combining selected variables into a species HSI using the component approach. Variables pertain to a riverine (R) habitat, lacustrine (L) habitat, or both (R, L).

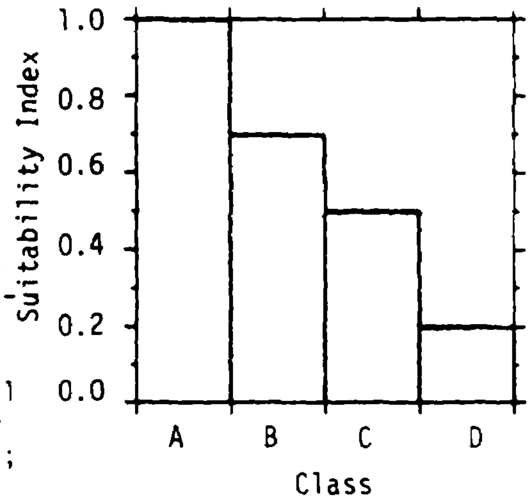
Habitat	Variable		Suitability Graph
R	(V_1)	Percent pools during average summer flow.	
R,L	(V_2)	Percent cover (logs, boulders, cavities, brush, debris, or standing timber) during summer within pools, backwater areas, and littoral areas.	

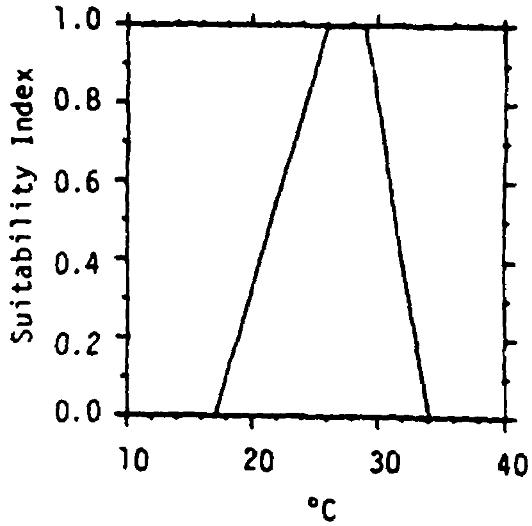
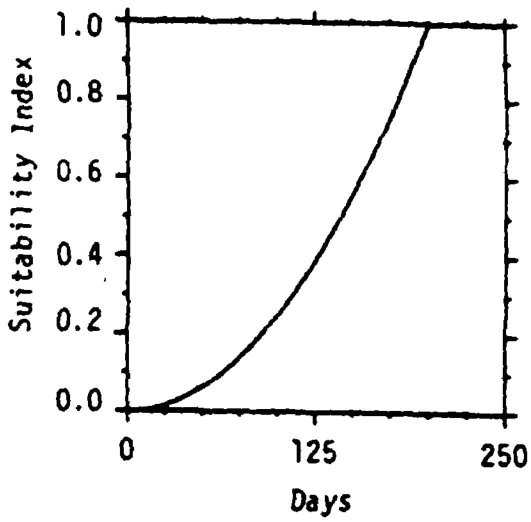
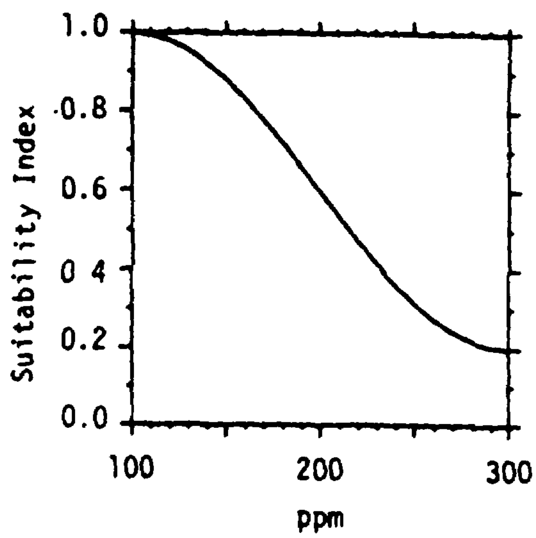
L (V_1) Percent littoral area during summer.



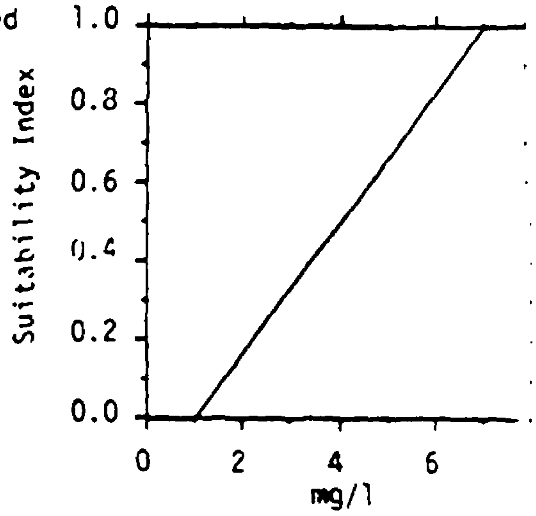
R (V_4) Food production potential in river by substrate type present during average summer flow.

A) Rubble dominant in riffle-runs with some gravel and/or boulders present; fines (silt and sand) not common; aquatic vegetation abundant (≥ 30%) in pool areas.
B) Rubble, gravel, boulders, and fines occur in nearly equal amounts in riffle-run areas; aquatic vegetation is 10-30% in pool areas.
C) Some rubble and gravel present, but fines or boulders are dominant; aquatic vegetation is scarce (< 10%) in pool areas.
D) Fines or bedrock are the dominant bottom material. Little or no aquatic vegetation or rubble present.

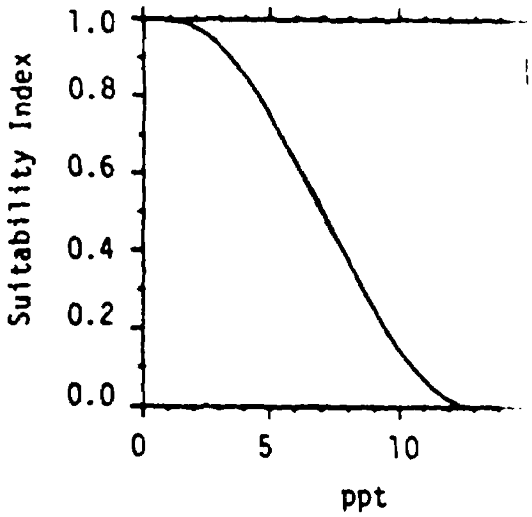


R,L	(V_5)	Average midsummer water temperature within pools, backwaters, or littoral areas (Adult).	
R,L	(V_6)	Length of agricultural growing season (frost-free days). <u>Note</u>: This variable is optional.	
R,L	(V_7)	Maximum monthly average turbidity during summer.	

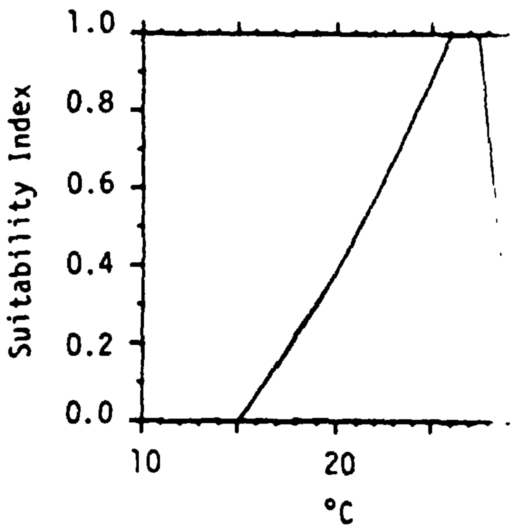
R,L (V$_8$) Average minimum dissolved oxygen levels within pools, backwaters, or littoral areas during midsummer.



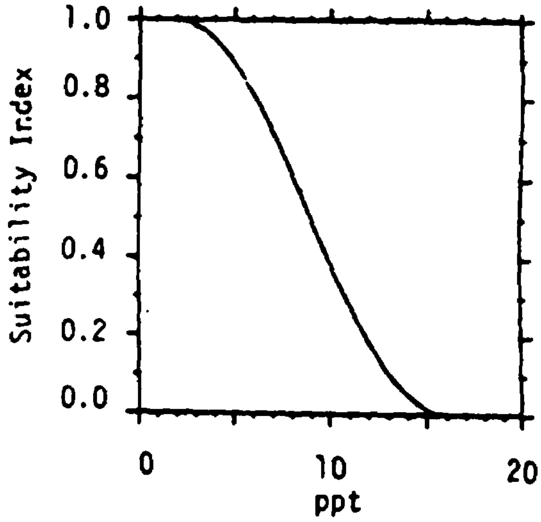
R,L (V$_9$) Maximum salinity during summer (Adult).



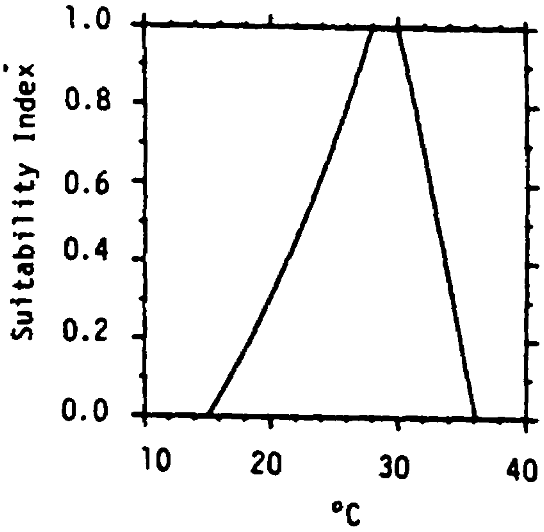
R,L (V$_{10}$) Average water temperatures within pools, backwaters, and littoral areas during spawning and embryo development (Embryo).



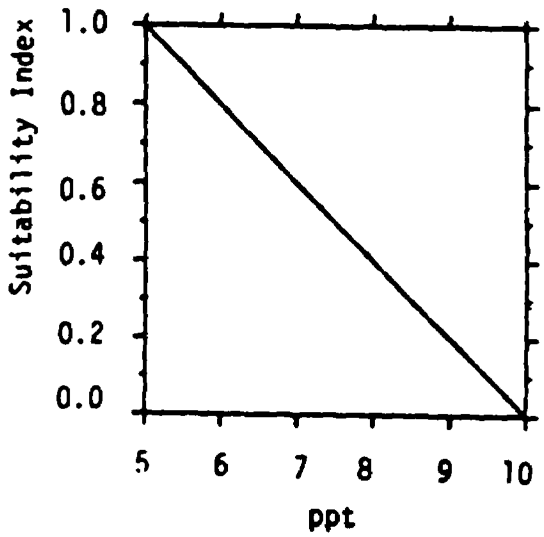
R,L (V_{11}) Maximum salinity during spawning and embryo development (Embryo).



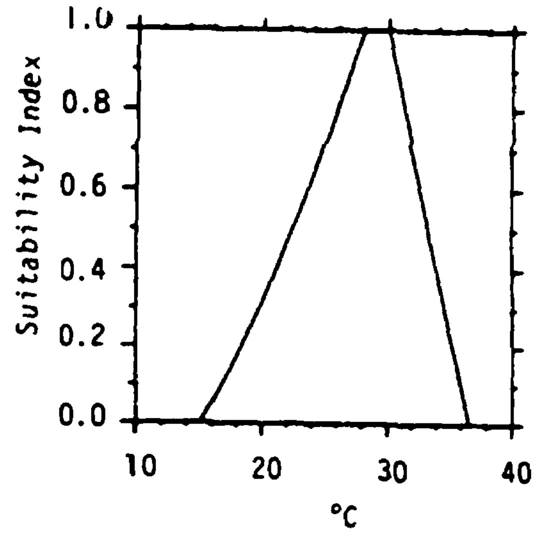
R,L (V_{12}) Average midsummer water temperature within pools, backwaters, or littoral areas (Fry).



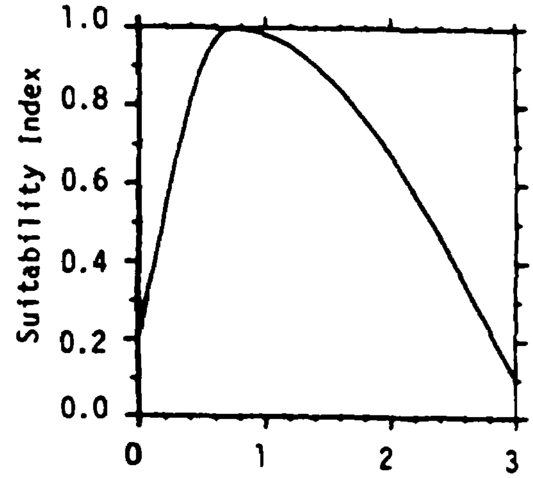
R,L (V_{13}) Maximum salinity during summer (Fry, Juvenile).



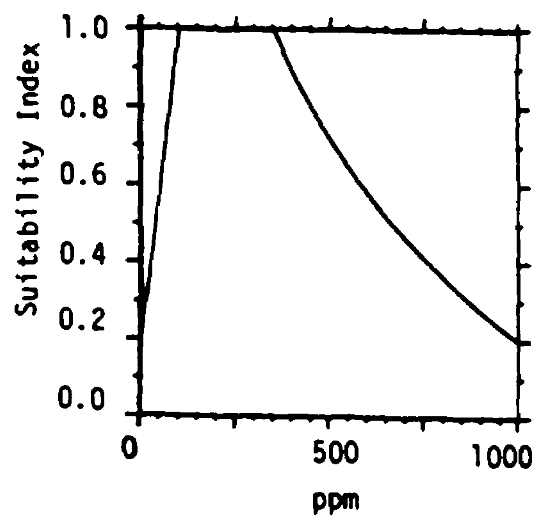
R,L (V_{14}) Average midsummer water temperature within pools, backwaters, or littoral areas (Juvenile).



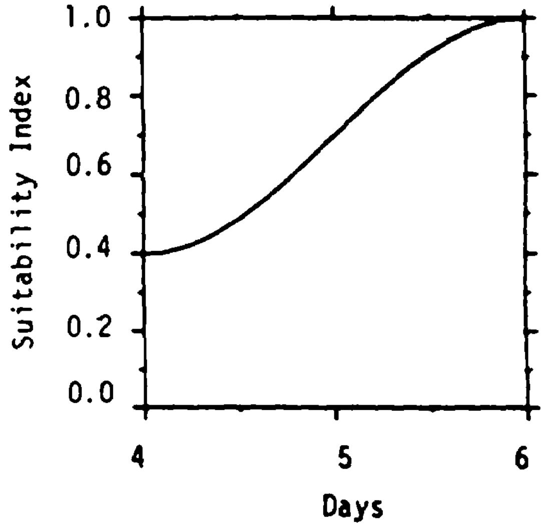
L (V_{15}) Storage ratio.



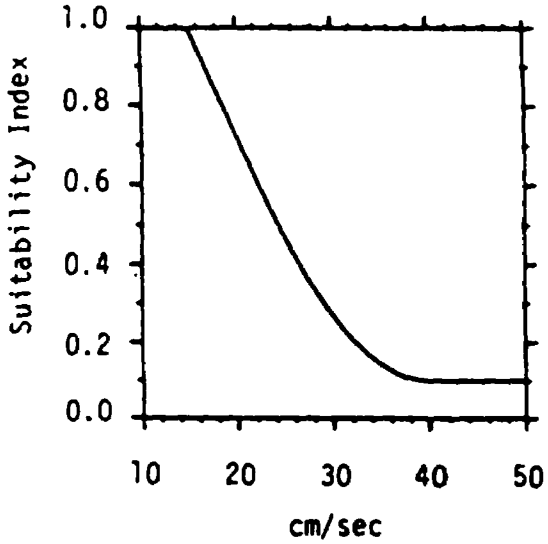
L (V_{16}) Monthly average TDS (total dissolved solids) during summer.



L (V₁₇) Maximum reservoir flushing rate while fry present (Fry).



R (V_{18}) Average current velocity in cover areas during average summer flow.



Riverine Model

These equations utilize the life requisite approach and consist of four components: food, cover, water quality, and reproduction.

Food (C_F).

$$C_F = \frac{V_2 + V_4}{2}$$

Cover (C_C).

$$C_C = (V_1 \times V_2 \times V_{10})^{1/3}$$

Water Quality (C_{WQ}).

$$C_{WQ} = \frac{\frac{2(V_5 + V_{12} + V_{14})}{3} + V_7 + 2(V_8) + V_9 + V_{13}}{7}$$

If V_5, V_{12}, V_{14} V_8, V_9, or V_{13} is ≤ 0.4, then C_{WQ} equals the lowest of the following: V_5, V_{12}, V_{14}, V_8, V_9, V_{13}, or the above equation.

Note: If temperature data are unavailable, $2(V_6)$ (length of agricultural growing season) may be substituted for the term

$$\frac{2(V_5 + V_{12} + V_{14})}{3}$$ in the above equation

Reproduction (C_R).

$$C_R = (V_1 \times V_2^2 \times V_6^2 \times V_{10}^2 \times V_{11})^{1/8}$$

If V_6, V_{10}, or V_{11} is ≤ 0.4, then C_R equals the lowest of the following: V_6, V_{10}, V_{11}, or the above equation.

HSI determination.

$$HSI = (C_F \times C_C \times C_{WQ}^2 \times C_R^2)^{1/6}$$, or

If C_{WQ} or C_R is ≤ 0.4, then the HSI equals the lowest of the following: C_{WQ}, C_R, or the above equation.

Sources of data and assumptions made in developing the suitability indices are presented in Table 1.

Sample data sets using riverine HSI model are listed in Table 2.

Lacustrine Model

This model utilizes the life requisite approach and consists of five components: food, cover, water quality, reproduction, and other.

<u>Food (C_F)</u>.

$$C_F = \frac{V_2 + V_3 + V_{16}}{3}$$

<u>Cover (C_C)</u>.

$$C_C = (V_2 \times V_3)^{1/2}$$

<u>Water Quality (C_{WQ})</u>.

C_{WQ} = same as in Riverine HSI Model

<u>Reproduction (C_R)</u>.

$$C_R = (V_2{}^2 \times V_3 \times V_8{}^2 \times V_{10}{}^2 \times V_{11})^{1/8}$$

If V_8, V_{10}, or V_{11} is ≤ 0.4, then C_R equals the lowest of the following: V_8, V_{10}, V_{11}, or the above equation.

<u>Other (C_{OT})</u>.

$$C_{OT} = \frac{V_{15} + V_{17}}{2}$$

Table 1. Data sources and assumptions for channel catfish suitability indices.

Variable and source		Assumption
V_1	Bailey and Harrison 1948	Optimum conditions for a diversity of velocities, depths, and structural features for channel catfish will be found when there are approximately equal amounts of pools and riffles.
V_2	Bailey and Harrison 1948 Marzolf 1957 Cross and Collins 1975	The strong preference of all life stages of channel catfish for cover indicates that some cover must be present for optimum conditions to occur.
V_3	Bailey and Harrison 1948 Marzolf 1957 Cross and Collins 1975	Lakes with small littoral area will provide less area for cover and food production for channel catfish and are therefore less suitable.
V_4	Bailey and Harrison 1948	The amount and type of substrate or the amount of aquatic vegetation associated with high production of aquatic insects (used as food by channel catfish and channel catfish prey species) is optimum.
V_5	Clemens and Sneed 1957 West 1966 Shrable et al. 1969 Starostka and Nelson 1974 Biesinger et al. 1979	Temperatures at the warmest time of year must reach levels that permit growth in order for habitat to be suitable. Optimum temperatures are those when maximum growth occurs.
V_6	Jenkins 1970	Growing seasons that are correlated with high standing crops are optimum.
V_7	Finnell and Jenkins 1954 Buck 1956 Marzolf 1957	High turbidity levels are associated with reduced standing crops and therefore are less suitable.
V_8	Moss and Scott 1961 Andrews et al. 1973 Carlson et al. 1974 Randolph and Clemens 1976	Lethal levels of dissolved oxygen are unsuitable. DO levels that reduce feeding are suboptimal.
V_9	Perry and Avault 1968 Perry 1973	Salinity levels where adults are most abundant are optimum. Any salinity level at which adults have been reported has some suitabilty.

Table 1. (concluded)

Variable and source		Assumption
V_{10}	Brown 1942 Clemens and Sneed 1957	Optimum temperatures are those which result in optimum growth. Temperatures that result in death or no growth are unsuitable.
V_{11}	Perry and Avault 1968 Perry 1973	Salinity levels at which spawning has been observed are suitable.
V_{12}	McCammon and LaFaunce 1961 Moss and Scott 1961 Macklin and Soule 1964 West 1966 Allen and Strawn 1968 Andrews 1972 Starostka and Nelson 1974	Optimum temperatures for fry are those when growth is best. Temperatures that result in no growth or death are unsuitable.
V_{13}	Allen and Avault 1970	Salinities that do not reduce growth of fry and juveniles are optimum. Salinities that greatly reduce growth are unsuitable.
V_{14}	Andrews et al. 1972 Andrews and Stickney 1972	Temperatures at which growth of juveniles is best are optimum. Temperatures that result in no growth or death are unsuitable.
V_{15}	Jenkins 1976	Storage ratios correlated with maximum standing crops are optimum; those correlated with lower standing crops are suboptimum.
V_{16}	Jenkins 1976	Total dissolved solids (TDS) levels correlated with high standing crops of warmwater fish are optimum; those correlated with lower standing crops are suboptimum. The data used to develop this graph are primarily from southeastern reservoirs.
V_{17}	Walburg 1971	Flushing rates correlated with reduced levels of fry abundance are suboptimal.
V_{18}	Miller 1966 Scott and Crossman 1973 Cross and Collins 1975	High velocities near cover objects will decrease the amount of usable habitat around the objects and are thus considered suboptimum.

Table 2. Sample data sets using riverine HSI model.

Variable		Data set 1		Data set 2		Data set 3	
		Data	SI	Data	SI	Data	SI
% pools	V_1	60	1.0	90	0.6	15	0.5
% cover	V_2	50	1.0	10	0.4	5	0.2
Substrate for food production	V_4	silt-gravel	0.7	silt-sand	0.5	sand	0.2
Temperature-Adult (° C)	V_5	28	1.0	32	0.4	22	0.5
Growing season	V_6	180	0.8	-	-	-	-
Turbidity (ppm)	V_7	50	1.0	210	0.5	160	0.8
Dissolved oxygen (mg/l)	V_8	4.5	0.6	4.0	0.5	4.0	0.5
Salinity-adult (ppt)	V_9	< 1	1.0	< 1	1.0	< 1	1.0
Temperature-Embryos (°C)	V_{10}	25	0.8	21.5	0.5	28.5	0.5
Salinity-Embryo (ppt)	V_{11}	< 1	1.0	< 1	1.0	< 1	1.0
Temperature-Fry (° C)	V_{12}	26.5	0.8	32	0.7	23	0.5
Salinity-Fry/ Juvenile (ppt)	V_{13}	< 1	1.0	< 1	1.0	< 1	1.0
Temperature-Juvenile (° C)	V_{14}	29	1.0	32	0.7	22	0.5
Velocity	V_{18}	15	1.0	5	1.0	30	0.3

Table 2. (concluded)

3

Variable	Data set 1 Data	Data set 1 SI	Data set 2 Data	Data set 2 SI	Data set 3 Data	Data set 3 SI
Component SI						
C_F =		0.85		0.45		0.20
C_C =		1.00		0.62		0.31
C_{WQ} =		0.87		0.40*		0.69
C_R =		0.86		0.58		0.47
HSI =		0.88		0.40*		0.43

*Note: $C_{WQ} \leq 0.4$; therefore, HSI = C_{WQ} in Data Set 2.

HSI determination.

$$HSI = (C_F \times C_C \times C_{WQ}^2 \times C_R^2 \times C_{OT})^{1/7}$$, or

If C_{WQ} or C_R is ≤ 0.4, then the HSI equals the lowest of the following: C_{WQ}, C_R, or the above equation.

Sample data sets using lacustrine HSI model are listed in Table 3.

Interpreting Model Outputs

The proper interpretation of the HSI produced by the models is one of comparison. If two water bodies have large differences in HSI's, then the one with the higher HSI should be able to support more catfish than the water body with the lower HSI, given that the model assumptions have not been violated. The actual differences in HSI that indicate a true difference in carrying capacity are unknown and likely to be high. We have aggregated a large number of variables into a single index with little or no quantitative information on how the variables interact to effect carrying capacity. The probability that we have made an error in our assumptions on variable interactions is high. However, we believe the model is a reasonable hypothesis of how the selected variables interact to determine carrying capacity.

Before using the model, any available statistical models, such as those described under model 3 in the next section, should be examined to determine if they better meet the goals of model application. Statistical models are likely to be more accurate in predicting the value of a dependent variable, such as standing crop, from habitat related variables than the HSI models described above. A statistical model is especially useful when the habitat variables in the data set used to derive the model have values similar to the proposed model application site. The HSI models described above may be most useful when habitat conditions are dissimilar to the statistical model data set or it is important to evaluate changes in variables not included in the statistical model.

The sample data sets consist of different variable values (and their corresponding SI score), which although not actual field measurements, are thought to represent realistic conditions that could occur in various channel catfish riverine or lacustrine habitats. We believe the HSI's calculated from the data reflect what carrying capacity trends would be in riverine or lacustrine habitats with the characteristics listed in the respective data sets.

Table 3. Sample data sets using lacustrine HSI model.

Variable		Data set 1		Data set 2		Data set 3	
		Data	SI	Data	SI	Data	SI
% cover	V_2	50	1.0	10	0.4	5	0.2
% littoral area	V_3	40	1.0	20	0.7	70	0.6
Temperature-Adult (° C)	V_5	26	1.0	20	0.3	33	0.2
Growing season	V_6	180	0.8	-	-	-	-
Turbidity	V_7	175	0.7	210	0.5	250	0.3
Dissolved oxygen	V_8	4.5	0.6	4.5	0.6	2.5	0.2
Salinity-Adult (ppt)	V_9	< 1	1.0	< 1	1.0	< 1	1.0
Temperature-Embryo (° C)	V_{10}	25	0.8	21.5	0.5	28	0.5
Salinity-Embryo (ppt)	V_{11}	< 1	1.0	< 1	1.0	< 1	1.0
Temperature-Fry (° C)	V_{12}	26.5	0.8	32	0.7	23	0.5
Salinity-Fry/ Juvenile (ppt)	V_{13}	< 1	1.0	< 1	1.0	< 1	1.0
Temperature-Juvenile (° C)	V_{14}	29	1.0	32	0.7	22	0.5
Storage ratio	V_{15}	1.5	0.9	.3	0.7	0.8	1.0
TDS (ppm)	V_{16}	200	1.0	300	1.0	600	0.6
Flushing rate while fry present (days)	V_{17}	15	1.0	4	0.4	11	1.0

Table 3. (concluded)

Variable	Data set 1		Data set 2		Data set 3	
	Data	SI	Data	SI	Data	SI
Component SI						
C_F =		1.00		0.70		0.47
C_C =		1.00		0.52		0.33
C_{WQ} =		0.82		0.30*		0.20*
C_R =		0.83		0.56		0.20
C_{OT} =		0.95		0.55		1.00
HSI =		0.89		0.30*		0.20*

*Note: $C_{WQ} \leq 0.4$; therefore, HSI = C_{WQ} in Data Sets 2 and 3.

ADDITIONAL HABITAT MODELS

Model 1

Optimal riverine habitat for channel catfish is characterized by the following conditions, assuming water quality is adequate: warm, stable water temperatures (summer temperatures of 25-31° C); an approximate 40-60% area of deep pools; and abundant cover in the form of logs, boulders, cavities, and debris (> 40% of pool area).

$$HSI = \frac{\text{number of above criteria present}}{3}$$

Model 2

Optimal lacustrine habitat for channel catfish is characterized by the following conditions, assuming water quality is adequate: warm, stable water temperatures (summer temperatures of 25-30° C); large surface area (> 500 ha); moderate to high fertility (TDS 100-350 ppm); clear to moderate turbidities (< 100 JTU); and abundant cover (> 40% in areas < 5 m deep).

$$HSI = \frac{\text{number of above criteria present}}{5}$$

Model 3

Use the reservoir standing crop regression equations for catfishes presented by Aggus and Morais (1979) to predict standing crop, then divide the predicted standing crop by the highest standing crop value used to develop the regression equation, in order to obtain an HSI.

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APPENDIX B-1. NATIONAL LIST OF OMNIVORE FISH SPECIES.

Common name	Latin name
Gizzard shad	Dorosoma cepedianum
Threadfin shad	Dorosoma petenense
Central mudminnow	Umbra limi
Eastern mudminnow	Umbra pygmaea
Mexican tetra	Astyanax tetra
Longfin dace	Agosia chrysogaster
Goldfish	Carassius auratus
Grass carp	Ctenopharyngodon idella
Common carp	Cyprinus carpio
Silverjaw minnow	Ericymba buccata
Alvord chub	Gila alvordensis
Utah chub	Gila atravia
Tui chub	Gila bicolor
Blue chub	Gila coerulea
Sonora chub	Gila ditaenia
Yaqui chub	Gila purpurea
Speckled chub	Hybopsis aestivalis
Blotched chub	Hybopsis insignis
California roach	Lavinia symmetricus
Virgin spinedace	Lepidomeda mollispinis
Hardhead	Mylopharodon conocephalus
Bluehead chub	Nocomis leptocephalus
Golden shiner	Notemigonus crysoleucas
White shiner	Notropis albeolus
Common shiner	Notropis cornutus
Bigmouth shiner	Notropis dorsalis
Blacknose shiner	Notropis heterolepis
Spottail shiner	Notropis hudsonius
Swallowtail shiner	Notropis procne
Sand shiner	Notropis stramineus
Skygazer shiner	Notropis uranoscopus
Mimic shiner	Notropis volucellus
Blackside dace	Phoxinus cumberlandensis
Northern redbelly dace	Phoxinus eos
Southern redbelly dace	Phoxinus erythrogaster
Bluntnose minnow	Pimephales notatus
Fathead minnow	Pimephales promelas
Blacknose dace	Rhinichthys atratulus
Speckled dace	Rhinichthys osculus
Redside shiner	Richardsonius balteatus
Creek chub	Semotilus atromaculatus
River carpsucker	Carpiodes carpio
Quillback	Carpiodes cyprinus
Highfin carpsucker	Carpiodes velifer
Utah sucker	Catostomus ardens
Longnose sucker	Catostomus catostomus
Bluehead sucker	Catostomus discobolus
Owens sucker	Catostomus fumeiventris
Flannelmouth sucker	Catostomus latipinnis
Largescale sucker	Catostomus macrocheilus
Sacramento sucker	Catostomus occidentalis

Mountain sucker	Catostomus platyrhyncus
Rio grande sucker	Catostomus plebeius
Tahoe sucker	Catostomus tahoensis
Blue sucker	Cycleptus elongatus
Smallmouth buffalo	Ictiobus bubalus
Black buffalo	Ictiobus niger
Oriental weatherfish	Misgurnus anguillicaudatus
Snail bullhead	Ictalurus brunneus
Black bullhead	Ictalurus melas
Yellow bullhead	Ictalurus natalis
Flat bullhead	Icalurus platycephalus
Channel catfish	Ictalurus punctatus
Walking catfish	Clarias batrachus
Chinese catfish	Clarias fuscus
Desert pupfish	Cyprinodon macularius
Sheepshead minnow	Cyprinodon variegatus
Plains killifish	Fundulus zebrinus
Porthole livebearer	Poeciliopsis gracilis
Gila topminnow	Poeciliopsis occidentalis
Pinfish	Lagodon rhomboides
Black acara	Cichlasoma bimaculatum
Rio grande perch	Cichlasoma cyanoguttatum
Firemouth	Cichlasoma meeki
Jewelfish	Hemichromis bimaculatus
Mozambique tilapia	Tilapia mossambica
Redbelly tilapia	Tilapia zilli
Shiner perch	Cymatogaster aggregata

APPENDIX B-2. NATIONAL LIST OF TOP CARNIVORE FISH SPECIES.

Common name	Latin name
Bull shark	Carcharhinus leucas
Alligator gar	Atractosteus spatula
Spotted gar	Lepisosteus oculatus
Longnose gar	Lepisosteus osseus
Florida gar	Lepisosteus platyrhincus
Shortnose gar	Lepisosteus platostomus
Bowfin	Amia calva
Machete	Elops affinis
Ladyfish	Elops saurus
Tarpon	Megalops atlanticus
Skipjack herring	Alosa chrysochloris
Hickory shad	Alosa mediocris
Pink salmon	Oncorhynchus gorbuscha
Chum salmon	Oncorhynchus keta
Coho salmon	Oncorhynchus kisutch
Sockeye salmon	Oncorhynchus nerka
Chinook salmon	Oncorhynchus tshawytscha
Golden trout	Salmo aguabonita
Arizona trout	Salmo apache
Cutthroat trout	Salmo clarki
Rainbow trout	Salmo gairdneri
Atlantic salmon	Salmo salar
Brown trout	Salmo trutta
Arctic char	Salvelinus alpinus
Bull trout	Salvelinus confluentus
Brook trout	Salvelinus fontinalis
Dolly varden	Salvelinus malma
Lake trout	Salvelinus namaycush
Inconnu	Stenodus leucichthys
Redfin pickerel	Esox americanus americanus
Grass pickerel	Esox americanus vermiculatus
Northern pike	Esox lucius
Muskellunge	Esox masquinongy
Chain pickerel	Esox niger
Sacramento squawfish	Ptychocheilus grandis
Colorado squawfish	Ptychocheilus lucius
Northern squawfish	Ptychocheilus oregonensis
Umpqua squawfish	Ptychocheilus umpquae
Flathead catfish	Pylodictis olivaris
Burbot	Lota lota
Fat snook	Centropomus parallelus
Tarpon snook	Centropomus pectinatus
Snook	Centropomus undecimalis
White bass	Morone chrysops
Striped bass	Morone saxatilis
Yellow bass	Morone mississippiensis
Rock bass	Ambloplites rupestris
Roanoke bass	Ambloplites cavifrons
Redeye bass	Micropterus coosae
Smallmouth bass	Micropterus dolomieui
Suwanee bass	Micropterus notius

Spotted bass	Micropterus punctulatus
Largemouth bass	Micropterus salmoides
Guadalupe bass	Micropterus treculi
White crappie	Pomoxis annularis
Black crappie	Pomoxis nigromaculatus
Yellow perch	Perca flavescens
Sauger	Stizostedion canadense
Walleye	Stizostedion vitreum
Gray snapper	Lutjanus griseus
Freshwater drum	Aplodinotus grunniens
Spotted seatrout	Cynoscion nebulosus
Red drum	Sciaenops ocellatus
Goldeye	Hiodon alosoides
White catfish	Ictalurus catus
Blue catfish	Ictalurus furcatus
Tucunare	Cichla ocellaris
Snakehead	Channa striata

APPENDIX C. NATIONAL LIST OF INTOLERANT FISH SPECIES.

Common name	Latin name
Cisco	Coregonus artedii
Arctic cisco	Coregonus autumnalis
Lake whitefish	Coregonus clupeaformis
Bloater	Coregonus hoyi
Kiyi	Coregonus kiyi
Bering cisco	Coregonus laurettae
Broad whitefish	Coregonus nasus
Humpback whitefish	Coregonus pidschian
Shortnose cisco	Coregonus reighardi
Least cisco	Coregonus sardinella
Shortjaw cisco	Coregonus zenithicus
Pink salmon	Oncorhynchus gorbuscha
Chum salmon	Oncorhynchus keta
Coho salmon	Oncorhynchus kisutch
Sockeye salmon	Oncorhynchus nerka
Chinook salmon	Oncorhnchus tshawytscha
Pygmy whitefish	Prosopium coulteri
Round whitefish	Prosopium cylindraceum
Mountain whitefish	Prosopium williamsoni
Golden trout	Salmo aguabonita
Arizona trout	Salmo apache
Cutthroat trout	Salmo clarki
Rainbow trout	Salmo gairdneri
Atlantic salmon	Salmo salar
Brown trout	Salmo trutta
Arctic char	Salvelinus alpinus
Bull trout	Salvelinus confluentus
Brook trout	Salvelinus fontinalis
Dolly varden	Salvelinus malma
Lake trout	Salvelinus namaycush
Inconnu	Stenodus leucichthys
Arctic grayling	Thymallus arcticus
Largescale stoneroller	Campostoma oligolepis
Redside dace	Clinostomus elongatus
Cutlips minnow	Exoglossum maxillingua
Bigeye chub	Hybobsis amblops
River chub	Nocomis micropogon
Pallid shiner	Notropis amnis
Pugnose shiner	Notropis anogenus
Rosefin shiner	Notropis ardens
Bigeye shiner	Notropis boops
Pugnose minnow	Noropis emiliae
Whitetail shiner	Notropis galacturus
Blackchin shiner	Notropis heterodon
Blacknose shiner	Notropis heterolepis
Spottail shiner	Noropis hudsonius
Sailfin shiner	Notropis hypselopterus
Tennessee shiner	Notropis leuciodus
Yellowfin shiner	Notropis lutipinnis
Ozark minnow	Notropis nubilus
Ozark shiner	Notropis ozarcanus

Silver shiner	Notropis photogenis
Duskystripe shiner	Notropis pilsbryi
Rosyface shiner	Notropis rubellus
Safron shiner	Notropis rubricroceus
Flagfin shiner	Notropis signipinnis
Telescope shiner	Notropis telescopus
Topeka shiner	Notropis topeka
Mimic shiner	Notropis volucellus
Steelcolor shiner	Notropis whipplei
Coosa shiner	Notropis xaenocephalus
Bleeding shiner	Notropis zonatus
Bandfin shiner	Notropis zonistius
Blackside dace	Phoxinus cumberlandensis
Northern redbelly dace	Phoxinus eos
Southern redbelly dace	Phoxinus erythrogaster
Blacknose dace	Rhinichthys atratulus
Pearl dace	Semotilus margarita
Alabama hog sucker	Hypentelium etowanum
Northern hog sucker	Hypentelium nigricans
Roanoke hog sucker	Hypentelium roanokense
Spotted sucker	Minytrema melanops
Silver redhorse	Moxostoma anisurum
River redhorse	Moxostoma carinatum
Black jumprock	Moxostoma cervinum
Gray redhorse	Moxostoma congestum
Black redhorse	Moxoatoma duquesnei
Rustyside sucker	Moxostoma hamiltoni
Greater jumprock	Moxostoma lachneri
Blacktail redhorse	Moxostoma poecilurum
Torrent sucker	Moxostoma rhothoecum
Striped jumprock	Moxostoma rupiscartes
Greater redhorse	Moxostoma valenciennesi
Ozark madtom	Noturus albater
Elegant madtom	Noturus elegans
Mountain madtom	Noturus eleutherus
Slender madtom	Noturus exilis
Stonecat	Noturus flavus
Black madtom	Noturus funebris
Least madtom	Noturus hildebrandi
Margined madtom	Noturus insignis
Speckled madtom	Noturus leptacanthus
Brindled madtom	Noturus miurus
Frecklebelly madtom	Noturus munitus
Brown madtom	Noturus phaeus
Roanoke bass	Ambloplites cavifrons
Ozark rockbass	Ambloplites constellatus
Rock bass	Ambloplites rupestris
Longear sunfish	Lepomis megalotis
Darters	Ammocrypta sp.
Darters	Etheostoma sp.
Darters	Percina sp.
Sculpins	Cottus sp.
O'opu alamoo (goby)	Lentipes concolor
O'opu nopili (goby)	Sicydium stimpsoni
O'opu nakea (goby)	Awaous stamineus

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Technical Support Manual: Waterbody Surveys and Assessments for Conducting Use Attainability Analyses

Volume II: Estuarine Systems

FOREWORD

The Technical Support Manual: Water Body Surveys and Assessments for Conducting Use Attainability Analyses in Estuarine Systems contains guidance prepared by EPA to assist States in implementing the revised Water Quality Standards Regulation (48 FR 51400, November 8, 1983). This document addresses the unique characteristics of estuarine systems and supplements the Technical Support Manual: Water Body Surveys and Assessments for Conducting Use Attainability Analyses (EPA, November, 1983). The central purpose of these documents is to provide guidance to assist States in answering three central questions:

(1) What are the aquatic protection uses currently being achieved in the water body?

(2) What are the potential uses that can be attained based on the physical, chemical and biological characteristics of the waterbody? and

(3) What are the causes of any impairment of the uses?

Consideration of the suitability of a water body for attaining a given use is an integral part of the water quality standards review and revision process. EPA will continue to provide guidance and technical assistance to the States in order to improve the scientific and technical bases of water quality standards decisions. States are encouraged to consult with EPA at the beginning of any standards revision project to agree on appropriate methods before the analyses are initiated, and to consult frequently as they are conducted.

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

Elliot Lomnitz
Criteria and Standards Division (WH-585)
401 M Street S.W.
Washington, D.C. 20460

Steven Schatzow, Director
Office of Water Regulations and Standards

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CHAPTER I

INTRODUCTION

EPA's Office of Water Regulations and Standards has prepared guidance to accompany changes to the Water Quality Standards Regulation (48 FR 51400). Programmatic guidance has been compiled and published in the Water Quality Standards Handbook (EPA, December 1983). This document discusses the water quality review and revision process; general programmatic guidance on mixing zones, flow, and economic considerations; use attainability analyses; and site specific criteria.

One of the major pieces of guidance in the Handbook is "Water Body Surveys and Assessments for Conducting Use Attainability Analyses." This guidance lays out the general framework for designing and conducting a use attainability analysis, whose objective is to answer the questions:

1. What are the aquatic life uses currently being achieved in the water body?

2. What are the potential uses that can be attained, based on the physical, chemical and biological characteristics of the water body?

3. What are the causes of impairment of the uses?

Technical guidance on conducting water body surveys and assessments was provided in the Technical Support Manual: Water Body Surveys and Assessments for Conducting Use Attainability Analyses (EPA, November 1983) in response to requests by several States for additional information. The Technical Support Manual essentially provides methods and tools for freshwater evaluations, but does not cover estuarine water bodies. The chapters presented in this volume address those considerations which are unique to the estuary. Those factors which are common to the freshwater and the estuarine system -- chemical evaluations in particular, are not discussed in this volume. Thus it is important that those who will be involved in the water body survey should also consult the 1983 Technical Support Manual. The methods and procedures offered in these guidance documents are optional and the States may apply them selectively, or they may use their own techniques or methods for conducting use attainability analyses.

The technical material presented in this volume deals with the major physical, chemical and biological attributes of the estuary: tides and currents, stratification, substrate characteristics; the importance of salinity, dissolved oxygen and nutrient enrichment; species diversity, plant and animal populations, and physiological adaptations which permit freshwater or marine organisms to survive in the estuary.

Given that estuaries are very complex receiving waters which are highly variable in description and are not absolutes in definition, size, shape, aquatic life or other attributes, those who will be performing use attainability analyses on estuarine systems should consider this volume as a frame of reference from which to initiate study design and execution, but not as an absolute guide.

CHAPTER II

PHYSICAL AND CHEMICAL CHARACTERISTICS

INTRODUCTION

The term estuary is generally used to denote the lower reaches of a river where tide and river flows interact. The generally accepted definition for an estuary was provided by Pritchard in 1952: "An estuary is a semi-enclosed coastal body of water having a free connection with the open sea and containing a measureable quantity of seawater." This description has remained remarkably consistent with time and has undergone only minor revisions (Emery and Stevenson, 1957; Cameron and Pritchard, 1963). To this day, such qualitative definitions are the most typical basis for determining what does and what does not constitute an estuary.

Estuaries are perhaps the most important social, economic, and ecologic regions in the United States. For example, according to the Department of Commerce (DeFalco, 1967), 43 of the 110 Standard Metropolitan Statistical Areas are on estuaries. Furthermore, recent studies indicate that many estuaries, including Delaware Bay and Chesapeake Bay, are on the decline. Thus, the need has arisen to better understand their ecological functions to define what constitutes a "healthy" system, to define actual and potential uses, to determine whether designated uses are impaired, and to determine how these uses can be preserved or maintained. This is the basis for the Use Attainability Analysis.

As part of such a program, there is a need to define impact assessment procedures that are simple, in light of the wide variability among estuaries, yet adequately represent the major features of each system studied. Estuaries are three-dimensional waterbodies which exhibit variations in physical and chemical processes in all three directions (longitudinal, vertical, and lateral) and also over time. However, following a careful consideration of the major physical and chemical processes and the time scales involved in use assessment, one can often define a simplified version of the prototype system for study.

In this chapter, a discussion is presented of important estuarine features and of major physical processes. A description of chemical evaluations is also presented, although the discussion herein is very limited since an extensive presentation was included in the earlier U.S. EPA Technical Support Manual (U.S. EPA November 1983). From this background, guidance for use attainability evaluations is given which considers the various assumptions that may be made to simplify the complexity of the analysis, while retaining an adequate description of the system. Finally, a framework for selecting appropriate desk-top and computer models for use attainability evaluations is outlined.

PHYSICAL PROCESSES

Introduction

Estuarine flows are the result of a complex interaction of:

- tides,
- wind shear,
- freshwater inflow (momentum and buoyancy),
- topographic frictional resistance,
- Coriolis effect,
- vertical mixing, and
- horizontal mixing.

In performing a use attainability study, one must simplify the complex prototype system by determining which of these effects or combination of effects is most important at the time scale of the evaluation. To do this, it is necessary to understand each of these processes and their impacts on the evaluation. A complete description of all of the above is beyond the scope of this report. Rather, illustrated are some of the features of each process, particularly in terms of magnitude and time scale.

Tides

Tides are highly variable throughout the United States, both in amplitude and phase. Figure II-1 (NOAA 1983) shows some typical tide curves along the Atlantic, Gulf of Mexico, and Pacific Coasts. Tidal amplitude can vary from 1 foot or less along the Gulf of Mexico (e.g., Pensacola, Florida) to over 30 feet in parts of Alaska (e.g., Anchorage) and the Maritime Provinces of Canada (e.g., the Bay of Fundy). Tidal phasing is a combination of many factors with differing periods. However, in the United States, most tides are predominantly based on 12.5-hour (semidiurnal), 25-hour (diurnal) and 4-day (semi-lunar) combinations. In some areas, such as Boston (Figure II-1), the tide is predominantly semidiurnal with 2 high tides and 2 low tides each day. In others, such as along the Gulf of Mexico, the tides are more typically mixed.

Tidal power is directly related to amplitude. This potential energy source can promote increased mixing through increased velocities and interactions with topographic features.

Wind

In many exposed bays or estuaries, particularly those in which tidal forcing is smaller, wind shear can have a tremendous impact on circulation patterns at time scales of a few hours to several days. An example is Tampa Bay on the West Coast of Florida, where tidal ranges are approximately 3 feet, and the terrain is generally quite flat. Wind can be produced from localized thunderstorms of a few hours duration, or from frontal movements with durations on the order of days. Unlike tides, wind is unpredictable in a real time sense. The usual approach to studying wind driven circulations is to develop a wind rose (Figure II-2) from local meteorological data, and base the study of impacts on statistically significant magnitudes and directions, or on winds that might produce the most severe impact.

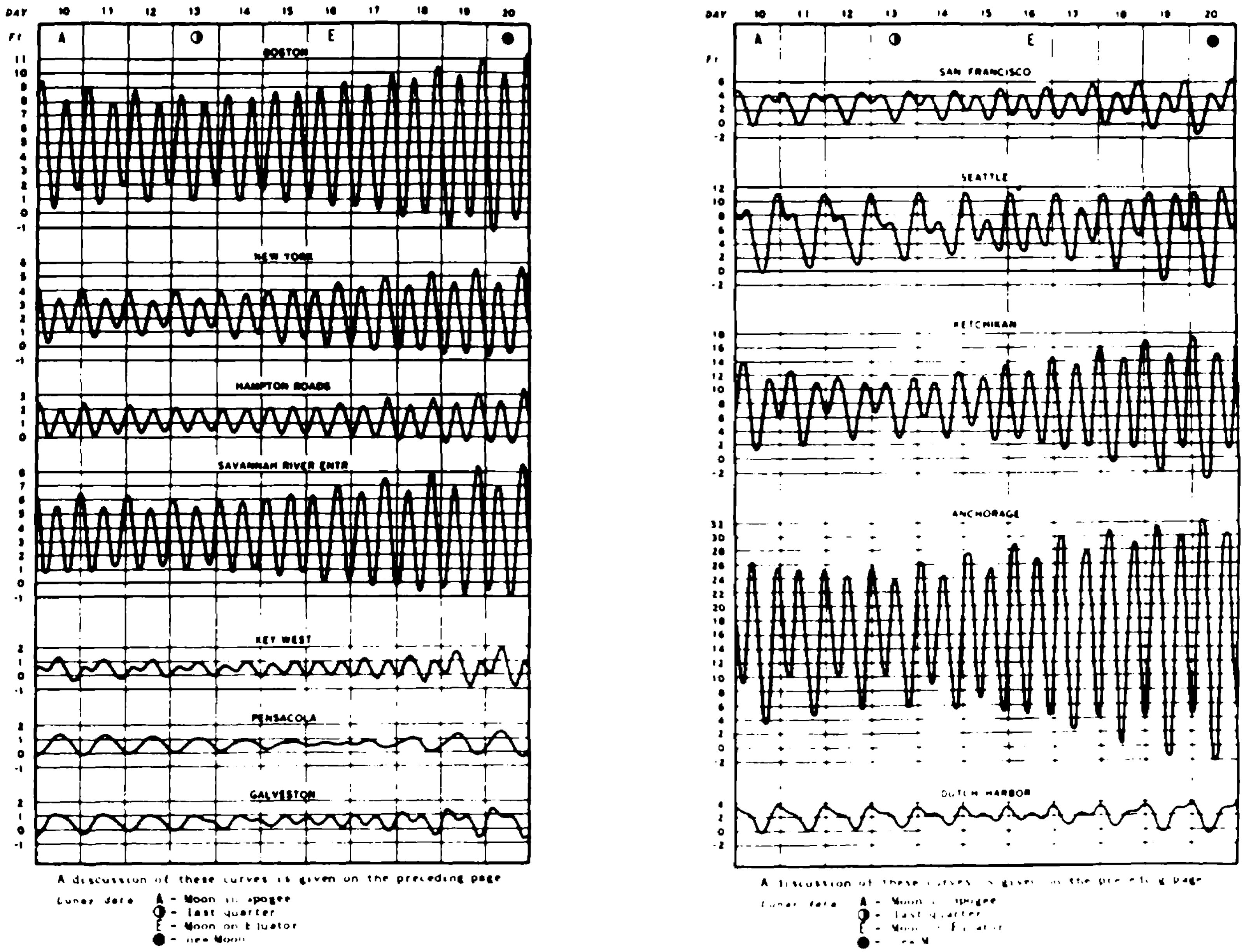


Figure II-1. Typical Tide Curves for United States Ports.

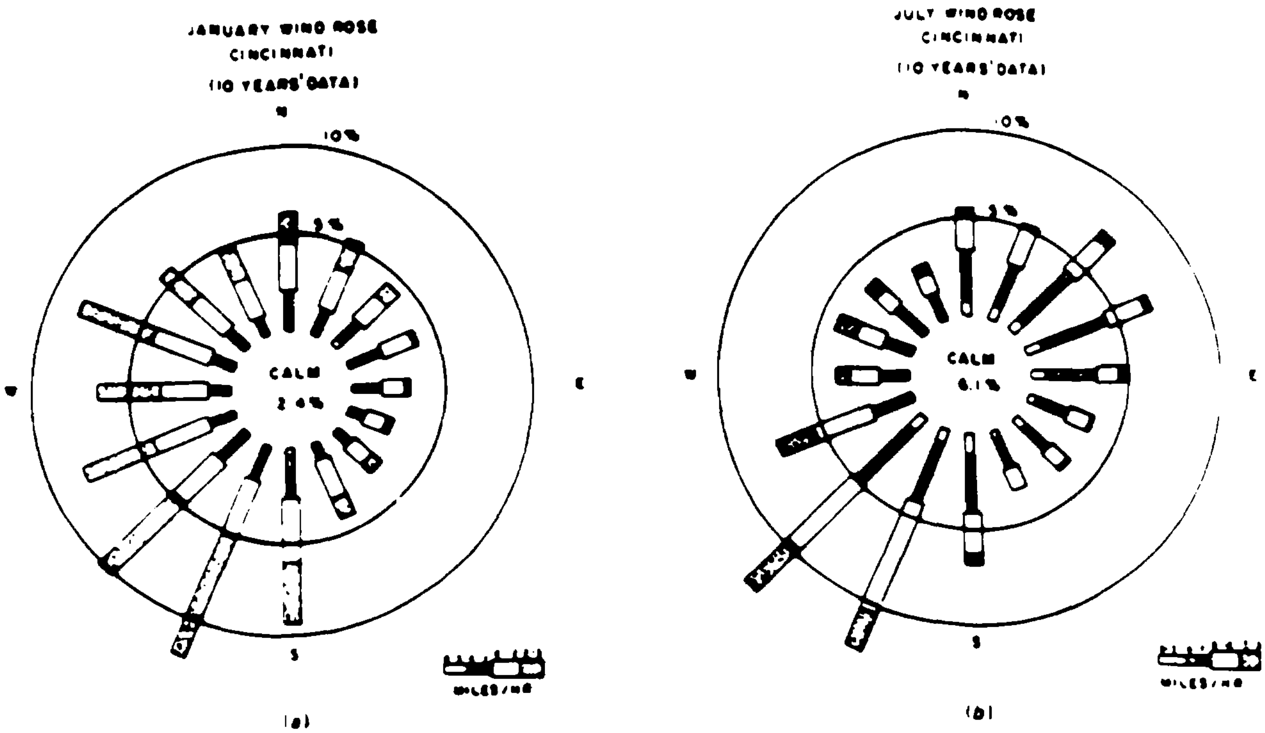


Figure II-2. Typical Wind Rose. (H.C. Perkins, 1974)

Freshwater Inflows

Freshwater inflows from a major riverine source can be highly variable from day to day and season to season. At the shorter time scale, the river may be responding to a localized thunderstorm, or the passage of a front. In many areas, however, the frequency of these events tends to group into a season (denoted the wet season) which is distinct from the remainder of the year (the dry season). The average monthly streamflow distributions in Figure II-3 illustrate that in Virginia the wet season is typically from December to May and comes mainly from portal systems. In Florida, however, the trend is reversed, with the wet season coinciding with the summer months when localized thunderstorms predominate.

It is important to consider the effect of freshwater flows on estuarine circulation, because streamflow is the only major mechanism which produces a net cross sectional flow over long averaging times. A common approach is to represent the estuary as a system drive by net freshwater flows in the downstream directory with other effects averaged out and lumped into a dispersion-type parameter. When using this assumption to evaluate the estuary system, one must weigh the consequences very carefully.

Freshwater is less dense and tends to "float" over seawater. In some cases, freshwater may produce a residual 2-layer flow pattern (such as in the James Estuary (Virginia) or Potomac Rivers) or even a 3-layer flow pattern (as in Baltimore Harbor). The danger is to treat such a distinctly 2-layer system as a cross-sectionally averaged, river driven system, and then try to explain why pollutants are observed upstream of a discharge point when no mechanism exists to produce this effect using a one-dimensional approach.

Friction

The estuary's topographic boundaries (bed and sides) produce frictional resistance to local currents. In some estuaries with highly variable geometries, this can produce a number of net nontidal (or tidally-averaged) effects such as residual eddies near headlands or tidal rectification. Pollutants trapped in residual eddies, perhaps from a wastewater treatment plant outfall, may have very large residence times that are not predictable from cross-sectionally averaged flows before such pollutants are flushed from the system.

Coriolis Effect

In wide estuaries, the Coriolis effect can cause freshwater to adhere to the right-hand bank (facing the open sea) so that the surface slopes upward to the right of the flow. The interface has an opposite slope to maintain geostrophic balance. For specific configurations and corresponding flow regimes, the boundary between outflow and inflow may actually cut the surface (Figure II-4a). This is the case in the lower reaches of the St. Lawrence estuary, for example, where the well-defined Gaspe current holds against the southern shore and counter flow is observed along the northern side. This effect is augmented by tidal circulation which forces ocean waters entering the estuary with the flood tide to adhere to the left side of the estuary (facing the open sea), and the ebb flow to the right side. Thus, as is often apparent from the surface salinity pattern in an estuary, the outflow is stronger on the right-hand side (Figure II-4b). The exact location and configuration of the saltwater/freshwater interface depends on the relative magnitude of the forces at play. Quantitative estimates of various mixing modes in estuaries are discussed below.

Vertical Mixing

All mixing processes are caused by local differences in velocities and by the fact that liquids are viscous (i.e., possess internal friction). In the vertical direction, the most common mixing occurs between riverine fresh waters and the underlying saline ocean waters.

If there were no friction, freshwater would flow seaward as a shallow layer on top of the seawater. The layer would become shallower and the velocity would decrease as the estuary widened toward its mouth. Friction between the two types of water requires a balancing pressure gradient down-estuary, explaining the salt wedge formation which deepens toward the mouth of the estuary, as seen in Figure II-5. Friction also causes mixing along the interface. A particularly well-defined salt wedge is observed in the estuary of the Mississippi River.

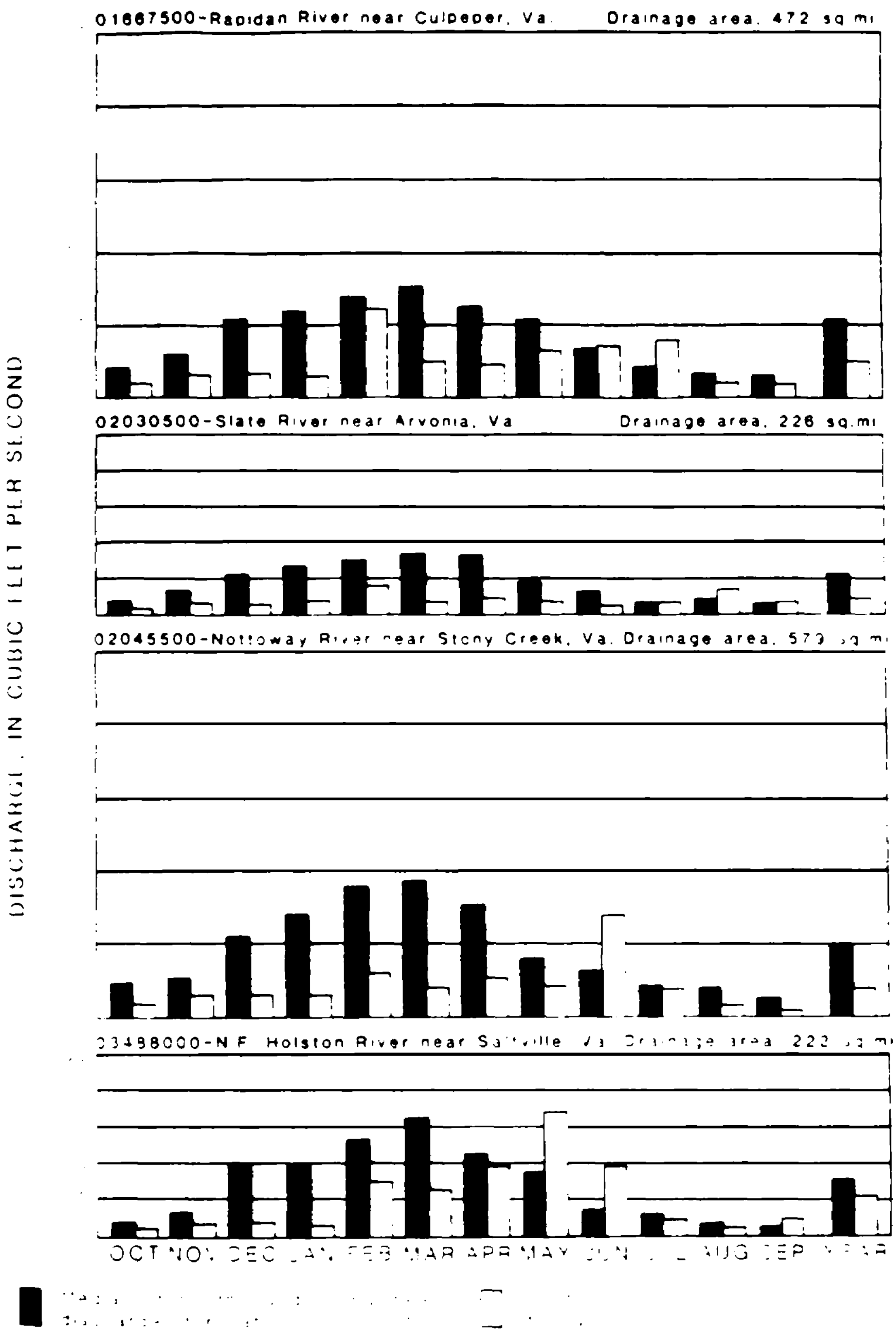
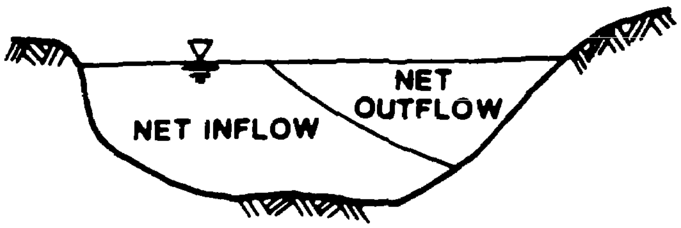
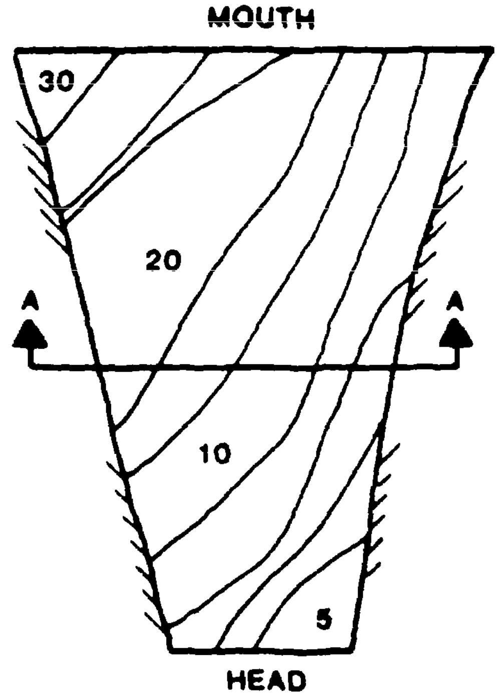


Figure II-3. Monthly Average Streamflows for location in Virginia. (from U. S. Geological Survey 1982)



a. Cross-section A-A looking Down-estuary.



b. Surface Salinity Distribution (ppt).

Figure II-4. Net Inflow and Outflow in a Tidal Estuary, Northern Hemisphere.

If significant mixing does not occur along the freshwater/saltwater interface, the layers of differing density tend to remain distinct and the system is said to be highly stratified in the vertical direction. If the vertical mixing is relatively high, the mixing process can almost completely break down the density difference, and the system is called well-mixed or homogeneous.

In sections of the estuary where there is a significant difference between surface and bottom salinity levels over some specified depth (e.g., differences of about 5 ppt or greater over about a 10 foot depth), the water column is regarded as highly stratified. An important impact of vertical stratification on use attainability is that the vertical density differences significantly reduce the exchange of dissolved oxygen and other constituents between surface and bottom waters. Consequently, persistent stratification can result in a depression of dissolved oxygen (DO) in the high salinity bottom waters that are cut off from the low salinity surface waters. This is because bottom waters depend upon vertical mixing with surface waters, which can take advantage of reaeration at the air-water interface, to replenish DO that is consumed as a result of organic materials within the water column and bottom sediments. In sections of the estuary exhibiting significant vertical stratification, vertical mixing of DO contributed by reaeration is limited to the low salinity surface waters.

As a result, persistent stratified conditions can cause the DO concentration in bottom water to fall to levels that cause stress on or mortality to the resident communities of benthic organisms.

Another potential impact of vertical stratification is that anaerobic conditions in bottom waters can result in increased release of nutrients such as phosphorus and ammonia-nitrogen from bottom sediments. During later periods or in sections of the estuary exhibiting reduced levels of stratification, these increased bottom sediment contributions of nutrients can eventually be transported to the surface water layer. These increased

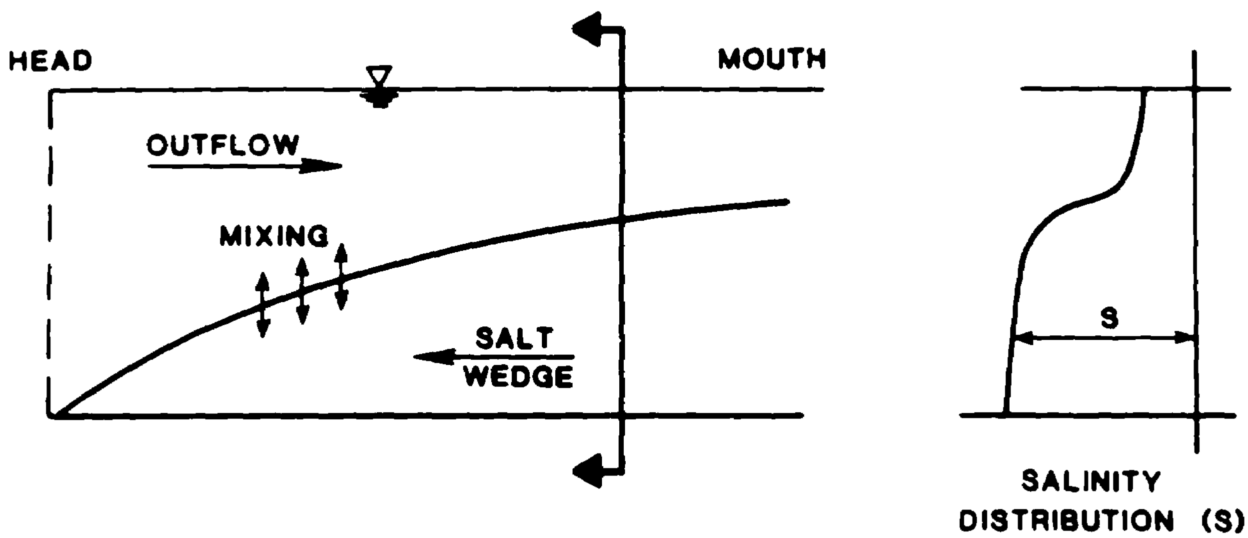


Figure II-5. Layered Flow in a Salt-wedge Estuary (Longitudinal Profile).

nutrient loadings on surface waters can result in higher phytoplankton concentrations that can exert diurnal DO stresses and reduced light penetration for rooted aquatic plants. In summary, the persistence and areal extent of vertical stratification is an important determinant of use attainability within an estuary.

Horizontal Mixing

Mixing also occurs in the horizontal plane, although it is often neglected in favor of vertical processes. As with vertical mixing, horizontal mixing is caused by localized velocity variations and internal friction, or viscosity. The velocity variations are usually produced by the interactions of topographic and bed or side frictional effects, resulting in eddies of varying sizes. Thus, horizontal constituent distributions tend to be broken down by differential advection, which when viewed as an average advection (laterally, or cross-sectionally) is called dispersion.

ESTUARINE CLASSIFICATION

Introduction

It is often useful to consider some broad classifications of estuaries, particularly in terms of features and processes which enable us to analyze them in terms of simplified approaches. The most commonly used groupings are based on geomorphology, stratification, circulation patterns, and time scales.

Geomorphological Classification

Over the years, a systematic structure of geomorphological classification has evolved. Dyer (1973) and Fischer et al. (1979) identify four groups:

- Drowned river valleys (coastal plain estuaries),
- Fjords
- Bar-built estuaries, and
- Other estuaries that do not fit the first three classifications.

Typical examples of North American estuaries are presented in Table II-1.

Coastal plain estuaries are generally shallow with gently sloping bottoms, with depths increasing uniformly towards the mouth. Such estuaries have usually been cut by erosion and are drowned river valleys, often displaying a dendritic pattern fed by several streams. A well-known example is Chesapeake Bay. Coastal plain estuaries are usually moderately stratified (particularly in the old river valley section) and can be highly influenced by wind over short time scales.

Bar built estuaries are bodies of water enclosed by the deposition of a sand bar off the coast through which a channel provides exchange with the open sea, usually servicing rivers with relatively small discharges. These

TABLE II-1. TOPOGRAPHIC ESTUARINE CLASSIFICATION

Type	Dominant Long-Term Process	Degree of Stratification	Examples
Coastal Plain	River Flow	Moderate	Chesapeake Bay, MD/VA James River, VA Potomac River, MD/VA Delaware Estuary, DE/NJ New York Bight, NY
Bar Built	Wind	Low or None	Little Sarasota Bay, FL Apalachicola Bay, FL Galveston Bay, TX Roanoke River, VA Albemarle Sound, NC Pamlico Sound, NC
Fjords	Tide	High	Alberni Inlet, B.C. Silver Bay, AL
Other Estuaries	Various	Various	San Francisco Bay, CA Columbia River, WA/OR

are usually unstable estuaries, subject to gradual seasonal and catastrophic variations in configuration. Many estuaries in the Gulf Coast and Lower Atlantic Regions fall into this category. They are generally a few meters deep, vertically well mixed and highly influenced by wind.

Fjords are characterized by relatively deep water and steep sides, and are generally long and narrow. They are usually formed by glaciation, and are more typical in Scandinavia and Alaska than the contiguous United States. There are examples along the Northwest Pacific Ocean, such as Alberni Inlet in British Columbia. The freshwater streams that feed a fjord generally pass through rocky terrain. Little sediment is carried to the estuary by the streams, and thus the bottom is likely to be a clean rocky surface. The deep water of a fjord is distinctly cooler and more saline than the surface layer, and the fjord tends to be highly stratified.

The remaining estuaries not covered by the above classification are usually produced by tectonic activity, faulting, landslides, or volcanic eruptions. An example is San Francisco Bay which was formed by movement of the San Andreas Fault System (Dyer, 1973).

Stratification

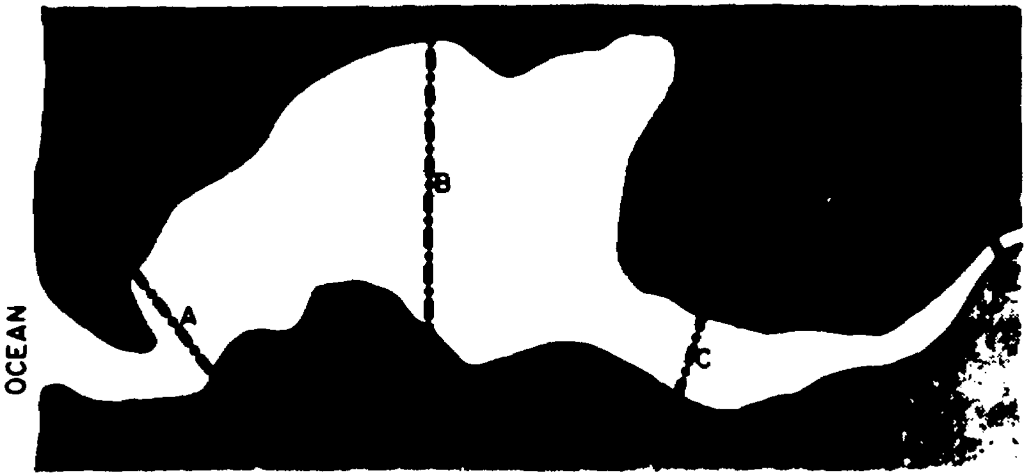
A second classification of estuaries is by the degree of observed stratification, and was developed originally by Pritchard (1955) and Cameron and Pritchard (1963). They considered three groupings (Figure II-6):

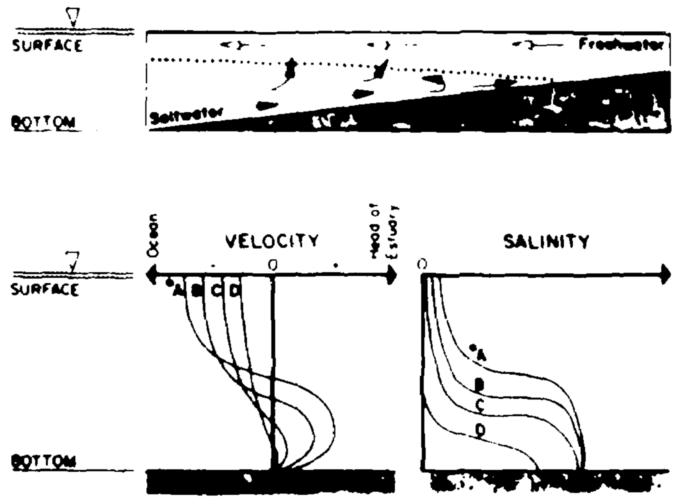
- The highly stratified (salt wedge) type
- Partially mixed estuary
- Vertically homogeneous estuary

Such a classification is intended for the general case of the estuary influenced by tides and freshwater inflows. Shorter term events, such as strong winds, tend to break down highly stratified systems by inducing greater vertical mixing. Examples of different types of stratification are presented in Table II-2.

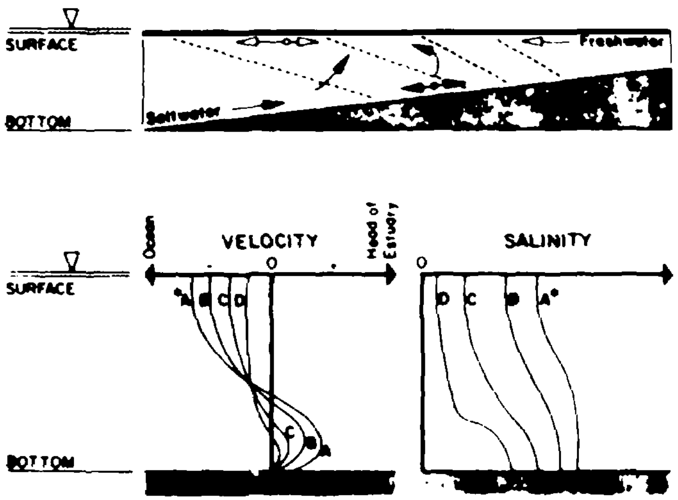
In the stratified estuary (Figure II-6a), large freshwater inflows ride over saltier ocean waters, with little mixing between layers. Averaged over a tidal cycle, the system usually exhibits net seaward movement in the freshwater layer, and net landward movement in the salt layer, as salt water is entrained into the upper layer. The Mississippi River Delta is an example of this type of estuary.

As the interfacial forces become great enough to partially break down the density differences, the system becomes partially stratified, or partially well-mixed (Figure II-6b). Tidal flows are now usually much greater than river flows, and flow reversals in the lower layer may still be observed, although they are generally not as large as for the highly stratified system. Chesapeake Bay and the James River estuary are examples of this type.

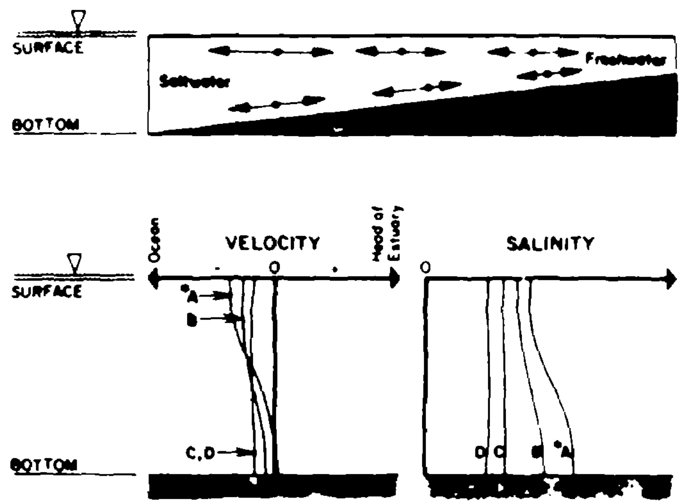




(a) Stratified



(b) Partially mixed



(c) Well-mixed

Figure II-6. Classification of Estuarine Stratification.

TABLE II-2. STRATIFICATION CLASSIFICATION

Type	River Discharge	Examples
Highly Stratified	Large	Mississippi River, LA Mobile River, AL
Partially Mixed	Medium	Chesapeake Bay, MD/VA James Estuary, VA Potomac River, MD/VA
Vertically Homogeneous	Small	Delaware Bay, DE/NJ Raritan River, NJ Biscayne Bay, FL Tampa Bay,FL San Francisco Bay, CA San Diego Bay, CA

In a well mixed system (Figure II-6c), the river inflow is usually very small, and the tidal flow is sufficient to completely break down the stratification and thoroughly mix the system vertically. Such systems are generally shallow so that the tidal amplitude to depth ratio is large and mixing can easily penetrate throughout the water column. The Delaware and Raritan River estuaries are examples of well-mixed systems.

Circulation Patterns

Circulation in an estuary (i.e., the velocity patterns as they change over time) is primarily affected by the freshwater outflow, the tidal inflow, and the effect of wind. In turn, the difference in density between outflow and inflow sets up secondary currents that ultimately affect the salinity distribution across the estuary. The salinity distribution is important in that it affects the distribution of fauna and flora within the estuary. It is also important because it is indicative of the mixing properties of the estuary as they may affect the dispersion of pollutants, flushing properties, and additional factors such as friction forces and the size and geometry of the estuary contribute to the circulation patterns.

The complex geometry of estuaries, in combination with the presence of wind, the effect of the earth's rotation (Coriolis effect), and other effects, often results in residual currents (i.e., of longer period than the tidal cycle) that strongly influence the mixing processes in estuaries. For example, uniform wind over the surface of an estuary produces a net wind drag force which may cause the center of mass of the water in the estuary to be displaced toward the deeper side since there is more water there. Hence a torque is induced causing the water mass to rotate.

In the absence of wind, the pure interaction of tides and estuary geometry may also cause residual currents. For example, flood flows through narrow inlets set up so-called tidal jets, which are long and narrow as compared to the ebb flows which draw from a larger area of the estuary, thus forcing a residual circulation from the central part of the estuary to the sides (Stommel and Farmer, 1952). The energy available in the tide is in part extracted to drive regular circulation patterns whose net result is similar to what would happen if pumps and pipes were installed to move water about in circuits. This is why this type of circulation is referred to as "tidal pumping" to differentiate from wind and other circulation (Fisher, et al., 1979).

Tidal "trapping" is a mechanism -- present in long estuaries with side embayments and small branching channels -- that strongly enhances longitudinal dispersion. It is explained as follows. The propagation of the tide in an estuary -- which represents a balance between the water mass inertia, the hydraulic pressure force due to the slope of the water surface, and the retarding bottom friction force -- results in main channel tidal elevations and velocities that are not in phase. For example, high water occurs before high slack tide and low water before low slack tide because the momentum of flow in the main channel causes the current to continue to flow against an opposing pressure gradient. In contrast, side channels which have less momentum can reverse the current direction faster, thus "trapping" portions of the main channel water which are then available for further longitudinal dispersion during the next flood tide.

Time Scales

The consideration of the time scales of the physical processes being evaluated is very important for any water quality study. Short-term conditions are much more influenced by a variety of short-term events which perhaps have to be analyzed to evaluate a "worst case" scenario. Longer term (seasonal) conditions are influenced predominantly by events which are averaged over the duration of that time scale.

The key to any study is to identify the time scale of the impact being evaluated and then analyze the forcing functions over the same time scale. As an example, circulation and mass transport in the upper part of Chesapeake Bay can be wind driven over a period of days, but is river driven over a period of one month or more. Table II-3 lists the major types of forcing functions on most estuarine systems and gives some idea of their time scales.

INFLUENCE OF PHYSICAL CHARACTERISTICS ON USE ATTAINABILITY

"Segmentation" of an estuary can provide a useful framework for evaluating the influence of estuarine physical characteristics such as circulation, mixing, salinity, and geomorphology on use attainability. Segmentation is the compartmentalizing of an estuary into subunits with homogeneous physical characteristics. In the absence of water pollution, physical characteristics of different regions of the estuary tend to govern the suitability for major water uses. Therefore, one major objective of segmentation is to subdivide the estuary into segments with relatively homogeneous physical characteristics so that differences in the biological communities among similar segments may be related to man-made alterations. Once the segment network is established, each segment can be subjected to a use attainability analysis. In addition, the segmentation process offers a useful management structure for monitoring conformance with water quality goals in future years.

The segmentation process is an evaluation tool which recognizes that an estuary is an interrelated ecosystem composed of chemically, physically, and biologically diverse areas. It assumes that an ecosystem as diverse as an estuary cannot be effectively managed as only one unit, since different uses and associated water quality goals will be appropriate and feasible for different regions of the estuary. The segmentation approach to use attainability assessment and water quality management has been successfully applied to several major receiving water systems, most notably Chesapeake Bay, the Great Lakes, and San Francisco Bay.

A potential source of concern about the construction and utility of the segmentation scheme for use attainability evaluations is that the estuary is a fluid system with only a few obvious boundaries, such as the sea surface and the sediment-water interface. Boundaries fixed in space are to be imposed on an estuarine system where all components are in communication with each other following a pattern that is highly variable in time. Fixed boundaries may seem unnatural to scientists, managers, and users, who are

TABLE II-3. TIME SCALES OF MAJOR PROCESSES

Forcing Function	Time Scale
TIDE	
One cycle	0.5-1 day
Neap/Spring	14 days
WIND	
Thunderstorm	1-4 hours
Frontal Passage	1-3 days
RIVER FLOW	
Thunderstorm	0.5-1 day
Frontal Passage	3-7 days
Wet/Dry Seasons	4-6 months

more likely to view the estuary as a continuum than as a system composed of separable parts. The best approach to dealing with such concerns is a segmentation scheme that stresses the dynamic nature of the estuary. The scheme should emphasize that the segment boundaries are operationally defined constructs to assist in understanding a changeable, intercommunicating system of channels, embayments, and tributaries.

In order to account for the dynamic nature of the estuary, it is recommended that estuarine circulation patterns be a prominent factor in delineating the segment network. Circulation patterns control the transport of and residence times for heat, salinity, phytoplankton, nutrients, sediment, and other pollutants throughout the estuary. Salinity should be another important factor in delineating the segment network. The variations in salinity concentrations from head of tide to the mouth typically produce a separation of biological communities based on salinity tolerances or preferences.

A segmentation scheme based upon physical processes such as circulation and salinity should track very well with the major chemical and biological processes. However, after developing a network based upon physical characteristics, segment boundaries can be refined with available chemical and biological data to maximize the homogeneity of each segment.

To illustrate the segmentation approach to evaluating relationships between physical characteristics and use attainability, the segmentation scheme applied to Chesapeake Bay is described below. While most of the estuaries subjected to use attainability evaluations will be considerably smaller and less diverse than Chesapeake Bay, the principles illustrated in the following example can serve as useful guidance for most estuary evaluations regardless of the spatial scale. Figure II-7 shows the main stem and tributary segments defined for Chesapeake Bay by the U.S. Environmental Protection Agency's Chesapeake Bay Program (U.S. EPA Chesapeake Bay Program 1982). As may be seen, the segment network consists of eight main stem segments designated by the prefix "CB" and approximately forty segments covering major embayments and tributaries. The methodology for delineating the main stem segments will be described first, followed by a discussion of the major embayments and tributaries.

Starting at the uppermost segment and working down the main stem, the boundary between CB-1 and CB-2 separates the mouth of the Susquehanna River from the upper Bay and lies in the region of maximum penetration of saltwater at the head of the Bay. South of this region most freshwater plankton would not be expected to grow and flourish, although some may be continually brought into the area by the Susquehanna River.

The boundary between CB-2 and CB-3 is the southern limit of the turbidity maximum, a region where suspended sediment causes light limitation of phytoplankton production most of the year. This boundary also coincides with the long-term summer average for the 5 parts per thousand (ppt) salinity contour which is an important physiological parameter for oysters.

The boundary between CB-3 and CB-4 is located at the Chesapeake Bay Bridge. It marks the northern limit of the 10 ppt salinity contour and of deep water anaerobic conditions in Chesapeake Bay stratification. In segment

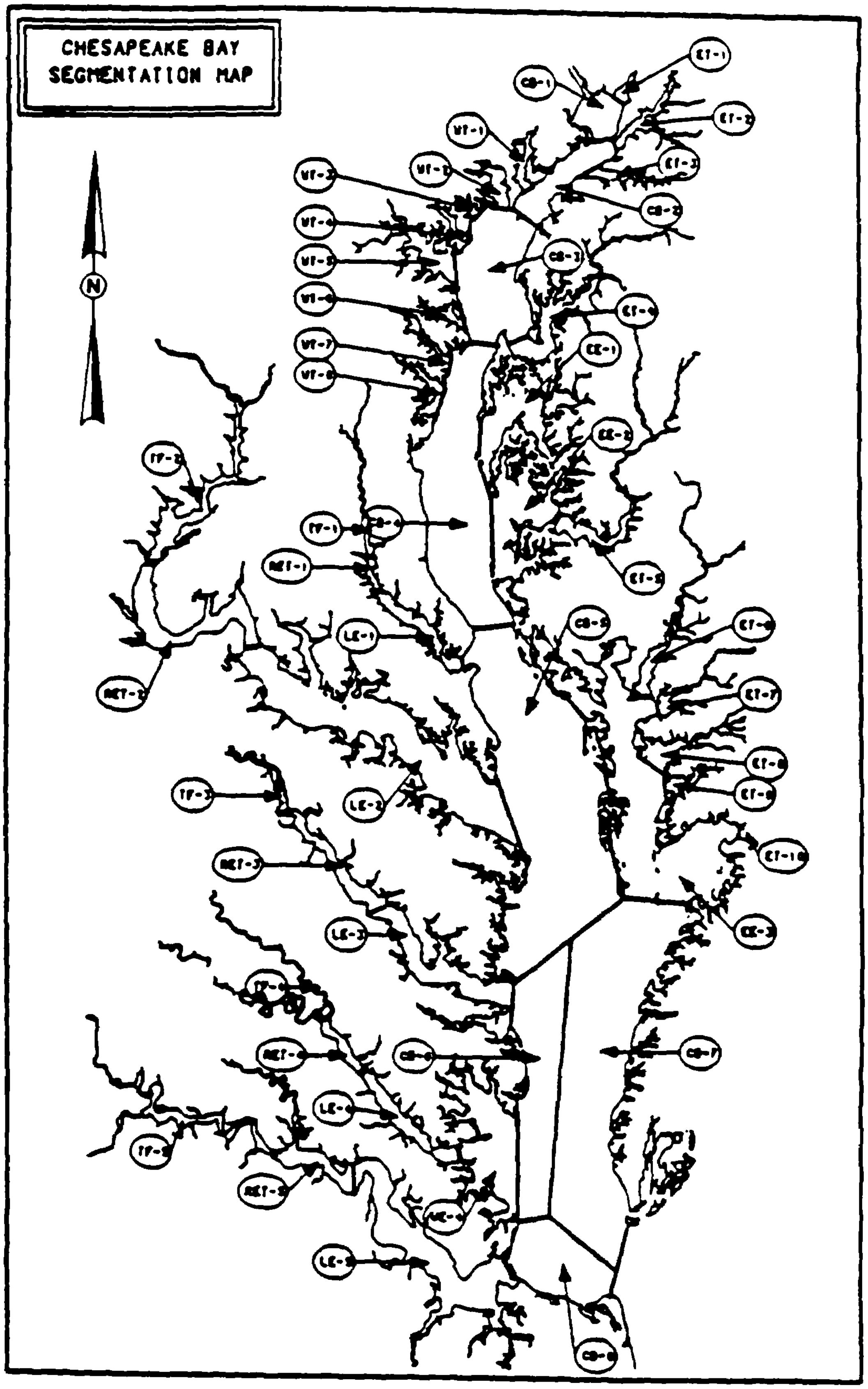


Figure II-7. Chesapeake Bay Program segments used in data analysis.(from U.S.EPA Chesapeake Bay Program 1982)

CB-4, water deeper than about 30 ft usually experiences oxygen depletion in summer which may result in oxygenless conditions and hydrogen sulfide production. When anaerobic conditions occur, these deep waters are toxic to fish, crabs, shellfish, and other benthic animals. Due to the increased release of nutrients from bottom sediments under oxygenless conditions, the anaerobic layer is also rich in phosphorus and ammonia-N which may reach surface waters by diffusion, mixing, and vertical advection either later in the year or in less stratified sections of the Bay. In spring, the region near the bridge is the site where phytoplankton and fish larvae that travel in the deep layer from the Bay mouth are brought to the surface by a combination of physical processes.

The boundary between CB-4 and CB-5 was established at a narrows. Below this point, the Patuxent and Potomac Rivers intersect the main stem of the Bay. It is characterized by average summer salinities of 12 to 13 ppt and is located at the approximate midpoint of the area subject to bottom water anaerobic conditions during the summer.

The boundary between CB-5 and CB-6/7 approximates the 18 ppt salinity contour and the southern limit of significant vertical stratification and anaerobic conditions in the bottom waters. Most of the deeper areas of the Bay are found in segment CB-5. As mentioned earlier, the bottom waters of segments CB-4 and CB-5 experience considerable nutrient enrichment during the summer when phosphorus and ammonia-N are released from bottom sediments. This region also exhibits high nitrate-N concentrations in the fall when the ammonia-N accumulated in summer is oxidized. The southern boundary of CB-5 also approximates the region where the elevated nitrate-N concentrations from the relatively high streamflows during the spring season becomes a critical factor in phytoplankton growth.

The boundary between CB-6 and CB-7 horizontally divides the lower Bay into two regions with different circulation patterns. North of this boundary, the Bay's density stratification results in two distinct vertical layers, with bottom waters moving in a net upstream flow and the surface layer flows moving downstream. Between this boundary and the Bay mouth the density distribution tends toward a cross-stream (i.e., horizontal) gradient rather than a vertical gradient. Net advective flows throughout a vertically well-mixed water column tend to flow northward in segment CB-7 and southward in CB-6 and CB-8. This pronounced horizontal gradient also exists across the Bay mouth. Thus, plankton and fish larvae are brought into the Bay with the higher salinity ocean waters along the eastern side of the lower Bay until they become entrained into the lower layer at segment CB-5 and are transported up the Bay to grow and mature.

Eastern shore embayments such as Eastern Bay (EE-1), the subestuary of the Choptank River (EE-2) and the Pocomoke and Tangier Sounds (EE-3) have salinities similar to adjacent Bay waters, and they are shallow enough to permit light penetration necessary for the growth of submerged aquatic vegetation (SAVs). These areas provide shelter for many benthic invertebrates and small fish which make an important contribution to the Bay's rich environment.

Boundaries have been delineated at the mouths of the Bay's major tributaries. These boundaries define the sources of freshwater, sediment, nutrients, and other constituents delivered to the main stem of the Bay. Along these boundaries, frontal zones between the tributary and main stem waters tend to concentrate detrital matter and nutrients, with circulation patterns governing the transport of many organisms to this food source.

The major tributaries are further subdivided into three segment classifications: tidal fresh (TF), river estuarine transition zone (RET), and lower subestuary (LE). The tidal fresh segments are biologically important as spawning areas for anadromous and semianadromous fish such as the alewife, herrings, shad, striped bass, white perch and yellow perch. There are also freshwater species which are resident in these areas such as catfish, minnows and carps. Algal blooms tend to be most prolific within the tidal fresh zone. The extent of these blooms is dependent upon nutrient supply, a range of factors such as retention time, and light availability. Most of the algal species that can flourish within tidal fresh segments are inhibited as they encounter the more saline waters associated with the transition zone.

The highest concentration of suspended solids is found at the interface of fresh and saline waters and it approximates the terminus of density dependent estuarine circulation. The area where this phenomenon occurs is typically referred to as the "turbidity maximum" zone. The significance of this area lies in its value as a sediment trap entraining not only material introduced upstream but, additionally, material transported in bottom waters from downstream. This mechanism also tends to concentrate any material associated with the entrained sediment. For example, Kepone accumulations within the James River estuary are highest in the turbidity maximum zone.

The final segment type found within the major tributaries is identified as the lower subestuary segment. This area extends from the turbidity maximum to the point where the tributary intersects the main stem of the Bay. Highly productive oyster bars are found in these segments. There is a heavy concentration of oyster bars in the lower subestuaries because of the favorable depth, salinities, and substrate. In general, the oyster bars are located in depths of less than 35 feet in salinities greater than 7-8 ppt and on substrates which are firm. Seasonal depressions of dissolved oxygen in bottom waters prevent the establishment of oyster bars in most waters over 35 feet deep.

CHEMICAL PARAMETERS

This section provides a brief discussion of chemical indicators of aquatic use attainment for estuaries. Three clarifications are necessary before beginning this discussion. First, while it is useful to refer to these parameters as "chemical" characteristics to distinguish them from the physical and biological parameters in a use attainability evaluation, these characteristics are traditionally referred to as water quality criteria and are referred to as such in other sections of this report. Second, chlorophyll-a is introduced in this section rather than in Chapter III because it is the primary impact indicator for chemicals such as nitrogen and phosphorus. Third, because an extensive discussion of chemical water quality indicators is presented in the earlier U.S. EPA Technical Support Manual (U.S. EPA November 1983), the discussion herein is very limited. Manual users who are interested in a more extensive discussion are referred to the previous volume.

The most critical water quality indicators for aquatic use attainment in an estuary are dissolved oxygen, nutrients and chlorophyll-a, and toxicants. Dissolved oxygen (DO) is an important water quality indicator for all fisheries uses. The DO concentration in bottom waters is the most critical indicator of survival and/or density and diversity for most shellfish and an important indicator for finfish. DO concentrations at mid-depth and surface locations are also important indicators for finfish. In evaluating use attainability, assessments of DO impacts should consider the relative contributions of three different sources of oxygen demand: (a) photosynthesis/respiration demand from phytoplankton; (b) water column demand; and (c) benthic oxygen demand. If use impairment is occurring, assessments of the significance of each oxygen sink can be used to evaluate the feasibility of achieving sufficient pollution control to attain the designated use.

Chlorophyll-a is the most popular indicator of algal concentrations and nutrient overenrichment which in turn can be related to diurnal DO depressions due to algal respiration. Typically, the control of phosphorus levels can limit algal growth in the upper end of the estuary, while the control of nitrogen levels can limit algal growth near the mouth of the estuary; however, these relationships are dependent upon factors such as N:P ratios and light penetration potential which can vary from one estuary to the next, thereby producing different limiting conditions within a given estuary. Excessive phytoplankton concentrations, as indicated by chlorophyll-a levels, can cause adverse DO impacts such as: (a) wide diurnal variations in surface DO's due to daytime photosynthetic oxygen production and nighttime oxygen depletion by respiration, and (b) depletion of bottom DO's through the decomposition of dead algae. Thus, excessive chlorophyll-a levels can deplete the oxygen resources required for bottom water fisheries, exert stress on the oxygen resources of surface water fisheries, and upset the balance of the detrital foodweb in the seagrass community through the production of excessive organic matter.

Excessive chlorophyll-a levels also result in shading which reduces light penetration for submerged aquatic vegetation. Consequently, the prevention of nutrient overenrichment is probably the most important water quality requirement for a healthy SAV community.

Blooms of certain phytoplankton can also be toxic to fish. For example, blooms of the toxic "red tide" organism during the early 1970's resulted in extensive fish kills in several Florida estuaries.

The nutrients of concern in the estuary are nitrogen and phosphorus. Their sources typically are discharges from sewage treatment plants and industries, and runoff from urban and agricultural areas. Increased nutrient levels lead to phytoplankton blooms and a subsequent reduction in DO levels, as discussed above. In addition, algal blooms decrease the depth to which light is able to penetrate, thereby affecting SAV populations in the estuary.

Sewage treatment plants are typically the major source of nutrients to estuaries in urbanized areas. Agricultural land uses and urban land uses represent significant nonpoint sources of nutrients. Often wastewater treatment plants are the major source of phosphorus loadings while nonpoint sources tend to be major contributors of nitrogen. In estuaries located near highly urbanized areas, municipal discharges probably will dominate the point source nutrient contributions. Thus, it is important to base control strategies on an understanding of the sources of each type of nutrient, both in the estuary and in its feeder streams.

In the Chesapeake Bay, an assessment of total nitrogen, total phosphorus, and N:P ratios indicates that regions where resource quality is currently moderate to good have lower concentrations of ambient nutrients, and N:P ratios between 10:1 and 20:1, indicating phosphorus-limited algal growth. Regions characterized by little or no SAV's (i.e., phytoplankton-dominated systems) or massive algal blooms had high nutrient concentrations and significant variations in the N:P ratios. Moving a system from one class to another could involve either a reduction of the limiting nutrient (N or P) or a reduction of the non-limiting nutrient to a level such that it becomes limiting. For example, removal of P from a system characterized by massive algal blooms could force it to become a more desirable phytoplankton-dominated system with a higher N:P ratio.

Clearly the levels of both nitrogen and phosphorus are important determinants of the uses that can be attained in an estuary. Because point sources of nutrients are typically much more amenable to control than nonpoint sources, and because nutrient (phosphorus) removal for municipal wastewater discharges is typically less expensive than nitrogen removal operations, the control of phosphorus discharges is often the method of choice for the prevention or reversal of use impairment in the upper estuary (i.e., tidal fresh zone). However, the nutrient control programs for the upper estuary can have an adverse effect on phytoplankton growth in the lower estuary (i.e., near the mouth) where nitrogen is typically the critical nutrient for eutrophication control. This is because the reduction of phytoplankton concentrations in the upper estuary will reduce the uptake and settling of the non-limiting nutrient which is typically nitrogen, thereby resulting in increased transport of nitrogen through the upper estuary to the lower estuary where it is the limiting nutrient for algal growth. The result is that reductions in algal blooms within the upper estuary due to the control of one nutrient (phosphorus) can result in increased phytoplankton concentrations in the lower estuary due to higher levels of the uncontrolled nutrient (nitrogen). Thus, tradeoffs between nutrient controls for the upper and lower estuary should be considered in evaluating measures for preventing or reversing use impairment. The Potomac Estuary is a good example of a system where tradeoffs between nutrient controls for the upper and lower estuary are being evaluated.

The impacts of toxicants such as pesticides, herbicides, heavy metals and chlorinated effluents are beyond the scope of this volume. However, the presence of certain toxicants in excessive concentrations within bottom sediments or the water column may prevent the attainment of water uses (particularly fisheries propagation/harvesting and seagrass habitat uses) in estuary segments which satisfy water quality criteria for DO, chlorophyll-a/nutrient enrichment, and fecal coliforms. Therefore, potential interferences from toxic substances need also to be considered in a use attainability study.

TECHNIQUES FOR USE ATTAINABILITY EVALUATIONS

Introduction

Use attainability evaluations generally follow the conceptual outline:

- Determine the present use of the estuary,
- Determine whether the present use corresponds to the designated use,
- If the present use does not correspond to the designated use, determine why, and
- Determine the optimal use for the system.

In assessing use levels for aquatic life protection, the first two items are evaluated in terms of biological measurements and indices. However, if the present use does not correspond to the designated use, one turns to physical and chemical factors to explain the lack of attainment, and the highest level the system can achieve.

The physical and chemical evaluations may proceed on several levels depending on the level of detail required, amount of knowledge available about the system (and similar systems), and budget for the use- attainability study. As a first step, the estuary is classified in terms of physical processes (e.g., stratification, flushing time) so that it can be compared with reference estuaries that exhibit similar physical characteristics. Once a similar estuary is identified, it can be compared with the estuary of interest in terms of water quality differences and differences in biological communities which can be related to man-made alteration (i.e., pollution discharges). It is important to consider a number of simplifying assumptions that can be made to reduce the conceptual complexity of the prototype system for easier classification and more detailed analyses.

The second step is to perform desk-top or simple computer model calculations to improve the understanding of spatial and temporal water quality conditions in the present system. These calculations include continuous point source and simple box model type calculations, among others.

The third step is to perform more detailed analyses to investigate system impact from known anthropogenic sources through the use of more sophisticated computer models. These tools can be used to evaluate the system response to removing individual point and nonpoint source discharges, so as to assist with assessments of the cause(s) of any use impairment.

Desktop Evaluations of System Characteristics

This section discusses desktop analyses for evaluating relationships between physical/chemical characteristics and use attainability. Desktop evaluations that can provide guidance for the selection of appropriate mathematical models for use attainability studies are also discussed.

Such evaluations can be used to characterize the complexity of an estuary, important physical characteristics such as the level of vertical stratification and flushing times, and violations of water quality criteria. Depending upon the complexity of the estuary, these evaluations can quantify the temporal and spatial dimensions of important physical/chemical characteristics and relationships to use attainability needs as summarized below:

1. Vertical Stratification

   a. Temporal Scale: During which seasons does it occur? What is the approximate duration of stratification in each season?

   b. Spatial Scale: How much area is subject to significant stratification in each season?

2. Flushing Times

   a. Temporal Scale: What are the flushing times for each major estuary segment and the estuary as a whole?

   b. Spatial Scale: Which segments exhibit relatively high flushing times? Relatively low flushing times?

3. Violations of Water Quality Criteria (based upon statistical analysis of measured data)

   a. Temporal Scale: Which seasons exhibit violations? How frequently and for what durations do violations occur in each season? Are the violations caused by short-term or long-term phenomena? Short-term phenomena include: DO sags due to combined sewer overflows or short-term nonpoint source loadings, and diurnal DO variations due to significant chlorophyll-a levels. Long-term phenomena include: seasonal eutrophication impacts due to nutrient loadings, seasonal DO sag due to point source discharges, and seasonal occurrence of anaerobic conditions in bottom waters due to persistent vertical stratification.

   b. Spatial Scale: What is the spatial extent of the violations (considering longitudinal, horizontal, and vertical directions)?

4. Relationship of Physical/Chemical Characteristics to Use Attainability Needs

a. Temporal Scale: Are use designations more stringent during certain seasons (e.g., spawning season)? Are acceptable physical/chemical characteristics required 100 percent of the time in each season in order to ensure use attainability?

b. Spatial Scale: Are there segments in the estuary which cannot support designated uses due to physical limitations? Are acceptable physical/chemical characteristics required in 100 percent of the estuary segment or estuary in order to ensure attainability of the use?

Simplifying Assumptions. Zison et al. (1977) and Mills et al. (1982) list a number of simplifying assumptions that can be made to reduce the complexity of estuary evaluations. However, care must be taken to ensure that such assumptions are applicable to the estuary under study and that they do not reduce the problem to one which is physically or chemically unreasonable. The following assumptions may be considered (Zison et al., 1977; Mills et al., 1982):

a. The present salinity distribution can be used as a direct measure of the distribution of all conservative continuous flow pollutants entering the estuary, and can be used as the basis for calculating dispersion coefficients for a defined freshwater discharge condition,

b. The vertical water column is assumed to be well mixed from top to bottom,

c. Flow and transport through the estuary is essentially one-dimensional,

d. The Coriolis effect may be neglected, which means that the estuary is assumed to be laterally homogeneous,

e. Only steady-state conditions will be considered, by using calculations averaged over one or more tidal cycles to estimate a freshwater driven flow within the estuary,

f. Regular geometry may be assumed, at least over the length of each segment, which means that topographically induced circulations are neglected,

g. Only one river inflow can be used in the evaluation,

h. No variations in tidal amplitude are permitted, and

i. All water leaving the estuary on each tidal cycle is replaced by a given percentage of "fresh" seawater.

The above list of assumptions are directed towards the specific objective of reducing the estuary to a one-dimensional, quasi-steady-state system amenable to desktop calculations. In reality these assumptions need to be carefully weighed so that important processes are not omitted from the analysis.

One approach is to start with a completely three-dimensional system, determine which assumptions can reasonably be made, and see what the answer means in terms of a simplified analysis. Procedures for making such determinations are discussed in the next section, but several examples are presented here for illustration.

The fact is that many narrow estuarine systems in which lateral homogeneity can be assumed, also exhibit 2 or more layers of residual flow, making the assumption of a one-dimensional system invalid. Conversely, given a vertically well-mixed system like Biscayne Bay, one cannot assume lateral homogeneity because the system is usually very wide wind mixing is too significant to permit such a simple analysis.

Degree of Stratification.

Freshwater is lighter than saltwater. Therefore, the river may be thought of as a source of buoyancy, of amount:

$$\text{Buoyancy} = \Delta\rho g Q_f \tag{1}$$

where
- $\Delta\rho$ = the difference in density between sea and river water, M/L^3
- g = acceleration of gravity, L/T^2
- Q_f = freshwater river flow, L^3/T
- M = units of mass
- L = units of length
- T = units of time

The tide on the other hand is a source of kinetic energy, equal to:

$$\text{kinetic energy} = \rho W U_t^3 \tag{2}$$

where
- ρ = the seawater density,
- W = the estuary width
- U_t = the square root of the averaged squared velocities.

The ratio of the above two quantities, called the "Estuarine Richardson Number" (Fischer 1972), is an estuary characterization parameter which is indicative of the vertical mixing potential of the estuary:

$$R = \frac{g\, Q_f}{W U_t^3} \tag{3}$$

If R is very large (above 0.8), the estuary is typically considered to be strongly stratified and the flow to be typically dominated by density currents. If R is very small, the estuary is typically considered to be well-mixed and the density effects to be negligible.

Another desktop approach to characterizing the degree of stratification in the estuary is to use a stratification-circulation diagram (Hansen and Rattray 1966). The diagram (shown in Figure II-8) requires the calculation of two parameters:

$$\text{Stratification Parameter} = \frac{\Delta S}{S_o} \tag{4}$$

$$\text{and Circulation Parameter} = \frac{U_s}{U_f}$$

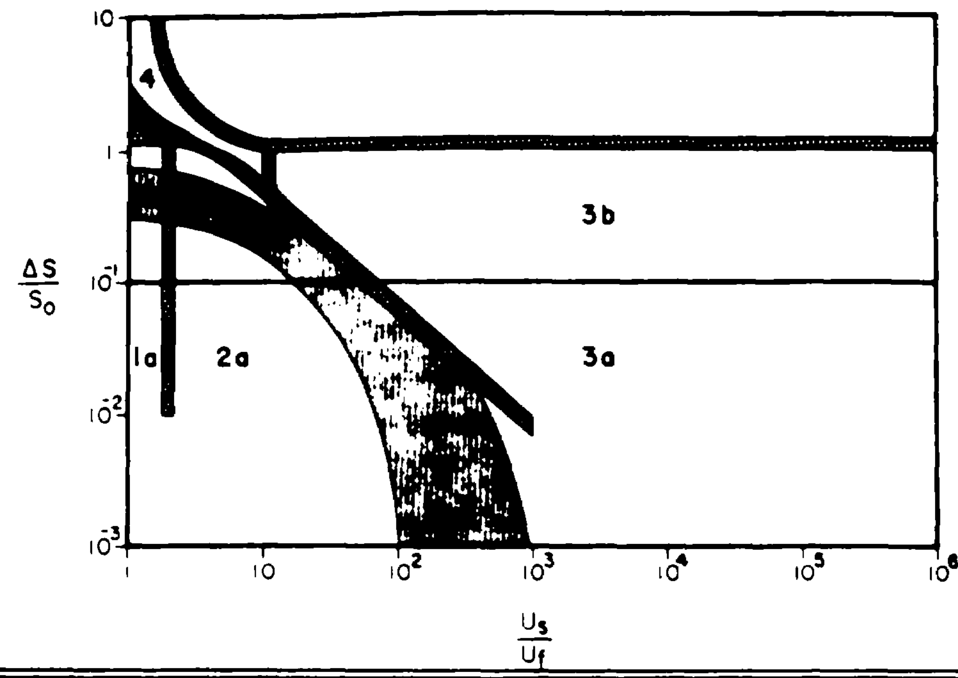
where

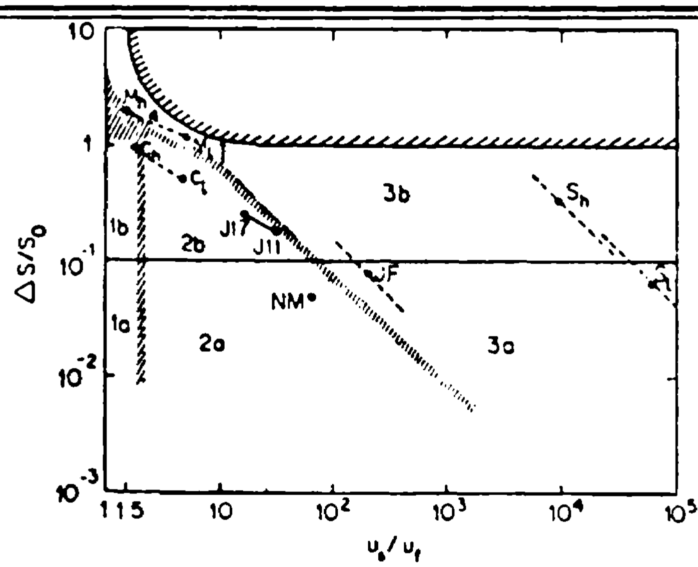
ΔS = time averaged difference between salinity levels at the surface and bottom of the estuary,
S_o = cross-sectional mean salinity,
U_s = net non-tidal surface velocity, and
U_f = mean freshwater velocity through the section.

To apply the stratification-circulation diagram in Figure II-8, which is based on measurements from a number of estuaries with known degrees of stratification, calculate the parameters of Equation (4) and plot the resulting point on the diagram. Type 1a represents slight stratification as in a laterally homogeneous, well-mixed estuary. In Type 1b, there is strong stratification. Type 2 is partially well-mixed and shows flow reversals with depth. In Type 3a the transfer is primarily advective, and in Type 3b the lower layer is so deep, as in a fjord, that circulation does not extend to the bottom. Finally, Type 4 represents the salt-wedge type with intense stratification (Dyer 1973).

The purpose of the analysis is to examine the degree of vertical resolution needed for the analysis. If the estuary is well-mixed, the vertical dimension may be neglected, and all constituents in the water column assumed to be dispersed evenly throughout. If the estuary is highly stratified, at least a 2-layer analysis must follow. For the case of a partially-mixed system, a judgment call must be made. The James River may be considered as an example which is partially stratified but was treated as a 2-layer system for a recent toxics study (O'Connor, et al., 1983).

A final desktop method for characterizing the degree of stratification is the calculation of the estuary number proposed by Thatcher and Harleman (1972):





(Station code: M, Mississippi River mouth; C, Columbia River estuary; J, James River estuary; NM, Narrows of the Mersey estuary; JF, Strait of Juan de Fuca; S, Silver Bay. Subscripts h and l refer to high and low river discharge; numbers indicate distance (in miles) from mouth of the James River estuary.

Figure II-8. Stratification Circulation Diagram and Examples.

$$E_d = \frac{P_t F_d^2}{Q_f T} \tag{5}$$

where

E_d = estuary number,
P_t = tidal prism volume (volume between low and high tides),
Q_f = freshwater inflow,
T = tidal period, and
F_d = densimetric Froude number =

$$\frac{U_1}{\left(g\left(\frac{\Delta\rho}{\rho_1}\right)h_1\right)^{\frac{1}{2}}}$$

where

U_1 = layer velocity,
g = acceleration due to gravity,
$\Delta\rho$ = density difference across interface,
ρ_1 = density in layer, and
h_1 = layer thickness.

Again, by comparing the calculated value with the values from known systems, one can infer the degree of stratification present. The reader should consult Thatcher and Harleman (1972) for further details.

Horizontal variations in density may still exist in a vertically well-mixed estuary, resulting in circulation that is density driven in the horizontal direction. It is helpful to understand density-driven circulation in an estuary (baroclinic circulation) in order to assess its effect in relation to turbulent diffusion on the landward transport of salinity. While numerous studies have been performed over the years (e.g., Hansen and Rattray 1965, 1966; Rigter, 1973), no unifying theory has emerged clearly delineating longitudinal, transverse and vertical dispersion mechanisms. This means that we still have to rely to a large extent on actual in-situ data.

Decisions about whether it is reasonable to neglect processes such as Coriolis effects and wind is often judgmental. However, Cheng (1977) did offer the following criterion for neglecting the Coriolis effect. The criterion is based on the Rossby number:

$$R_o = \frac{\Omega \bar{u}}{L} \tag{6}$$

where R_o = Rossby number,
$\bar{u}^o$ = characteristic wind velocity = 1/2 peak surface velocity,
Ω = earth's rotation rate, and
L = length of estuary,

Cheng suggested that for $R_o < 0.1$, the Coriolis effect is small. Wind is so highly variable and unpredictable that it is almost always neglected. In general, it has little effect on steady-state conditions, except in large open estuaries.

Finally, the use of simplified geometries, such as uniform depth and width is highly judgmental. One may choose to neglect side embayments, minor tributaries, narrows and inlets as a symplifying approach to achieve uniform geometry. However, it is always important to consider the consequences of this assumption.

Flushing Time. The time that is required to remove pollutant mass from a particular point in an estuary (usually some upstream location) is called the flushing time. Long flushing times are often indicative of poor water quality conditions due to long residence times for pollutants. Flushing time, particularly in a segmented estuary, can also be used in an initial screening of alternate locations for facilities which discharge constituents detrimental to estuarine health if they persist in the water column for lengthy periods.

Factors influencing flushing times are tidal ranges, freshwater inflows, and wind. All of these forcing functions vary over time, and may be somewhat unpredictable (e.g., wind). Thus, flushing time calculations are usually based on average conditions of tidal range and freshwater inflows, with wind effects neglected.

The Fraction of Fresh Water Method for flushing time calculation is based upon observations of estuarine salinities:

$$F = \frac{S_o - S_e}{S_e} \tag{7}$$

where F = flushing time in tidal cycles,
S_o = salinity of ocean water, and
S_e = mean estuarine salinity.

The tidal prism method for flushing time calculation considers the system as one unit with tidal exchange being the dominant process:

$$F = \frac{V_L + P}{P} \tag{8}$$

where F = flushing time in tidal cycles,
V_L = low tide volume of the estuary, and
P = tidal prism volume (volume between low and high tides).

The Tidal Prism technique was further modified by Ketchum (1951) to segment the estuary into lengths defined by the maximum excursion of a particle of water during a tidal cycle. This technique can now include a freshwater inflow:

$$F = \sum_{i=1}^{n} \frac{V_{Li} + P_i}{P_i} \tag{9}$$

where F = flushing time in tidal cycles,
i = segment number,
n = number of segments
V_{Li} = low tide volume in segment i, and
P_i = tidal prism volume in segment.

Riverine inflow is accounted for by setting the upstream length equal to the river velocity multiplied by the tidal period, and setting:

$$P_o = Q_f T \tag{10}$$

where P_o = tidal prism volume in upstream segment,
Q_f = freshwater flow, and
T = tidal period.

Finally, the replacement time technique is based upon estuarine geometry and longitudinal dispersion:

$$t_R = 0.4\ L^2/E_L \tag{11}$$

where t_R = replacement time,
L = length of estuary, and
E_L = longitudinal dispersion coefficient.

This technique requires knowledge of a longitudinal dispersion coefficient, E_L, which may not be known from direct estuarine measurements. A coefficient based upon measured data from a similar estuary may be assumed (see Table II-4 for typical values in a number of U.S. estuaries) or it may be estimated from empirical relationships, such as the one reported by Harleman (1964):

$$E_L = 77 \; n \; u \; R^{5/6} \tag{12}$$

or Harleman (1971):

$$E_L = 100 \; n \; u_{max} \; R^{5/6} \tag{13}$$

where E_L = longitudinal dispersion coefficient (ft^2/sec),
n = Manning's roughness coefficient (0.028-0.035, typically),
u = velocity (ft/sec),
u_{max} = maximum tidal velocity, and
R = hydraulic radius = A/P

where A = cross sectional area,
P = wetted perimeter.

Desktop Calculations of Pollutant Concentrations

Classification and characterization are means of identifying estuarine types and their major processes as a basis for comparison with reference estuaries. There are some desktop methods for calculating ambient water quality for defined pollutant loading conditions which can provide further insight into system response for use attainability evaluations.

These techniques usually assume uniform geometry, a well-mixed system, and net freshwater driven flows. There are essentially two types of desktop calculations for ambient water quality evaluations -- mixed tank analyses and simple analytic solutions to the governing equations.

Under the first approach, the pollutant discharge is continuously mixed with an inflowing river, or else at a point along the estuary. Solutions at steady-state are well-known (Mills et al., 1982). For a river borne pollutant inflow, the steady-state concentration for a conservative pollutant may be calculated as follows:

$$C_{pi} = \frac{T_i \; Q_f}{V_i} \tag{14}$$

TABLE II-4

OBSERVED LONGITUDINAL DISPERSION COEFFICIENTS

Estuary	River Flow	Dispersion Coefficents	
	(cfs)	(m^2/sec)	(ft^2/sec)
Delaware River (DE/NJ)	2500	150	1600
Hudson River (NY)	5000	600	6500
East River (NY)	0	300	3250
Cooper River (SC)	10000	900	9700
Savannah River (GA, SC)	7000	300-600	3250-6500
Lower Raritan River (NJ)	150	150	1600
South River (NJ)	23	150	1600
Houston Ship Channel (TX)	900	800	8700
Cape Fear River (NC)	1000	60-300	650-3250
Potomac River (MD/VA)	550	30-300	325-3250
Compton Creek (NJ)	10	30	325
Wappinger and Fishkill Creek (NY)	2	15-30	160-325
San Francisco Bay (CA):			
Southern Arm	-	18-180	200-2000
Northern Arm	-	46-1800	500-20000

SOURCE: From Mills et al. (1982).

where C_{pi} = pollutant concentration in segment i,
T_i = flushing time for segment i,
Q_f = freshwater flow, and
V_i = water volume at segment i.

For a direct discharge along the estuary, the concentration of a conservative pollutant at any section downstream is given by (Dyer 1973):

$$C_x = \left(\frac{C_p\ Q_p\ f_x}{Q_f + Q_p}\right)\left(\frac{S_s - S_x}{S_s - S_o}\right) \tag{15}$$

and at a section upstream:

$$C_x = \left(\frac{C_p\ Q_p\ f_o}{Q_f + Q_p}\right)\left(\frac{S_x}{S_o}\right) \tag{16}$$

where C = concentration,
C_p = inflow concentration,
Q_p = inflow rate,
f_o = fraction of freshwater in segment,
Q_f = river flow,
S = salinity,
subscript x - denotes distance downstream,
subscript o - denotes point of injection, and
subscript s - denotes ocean salinity.

A refinement to the above desktop methods involve calculations for nonconservative pollutants. The usual approach is to rely upon a first order decay relationship:

$$C_t = C_o\ e^{-k_T t} \tag{17}$$

where C_t = concentration at time t,
C_o = initial concentration, and
k_T = decay or reaction rate at temperature T.

The decay rate, k, is often expressed as a function of water temperature, based upon the departure from a standard temperature (usually 20°C):

$$k_T = k_{20}\, \Theta^{T-20} \tag{18}$$

where k_{20} = decay or reaction rate at 20°C, and
= constant (1.03-1.04).

The final pollutant concentration is then calculated by applying a first-order decay to the dilution concentration given from Equations (14)-(16), based on an estimate of travel time to the cross-section of interest.

The second approach is to greatly simplify the governing mass transport equation, and derive a closed-form solution which can be evaluated using a hand-held calculator, for continuous, discrete discharges of either conservative or non-conservative pollutants (Mills et al., 1982). From the basic simplified equation for a continuous discharge of a nonconservative pollutant:

$$u\frac{dc}{dx} = E_L \frac{d^2c}{dx^2} - kc \tag{19}$$

the following solution can be readily derived:

$$c_x = c_o \exp \left[\frac{ux}{2E_L}\left(1 \pm \left(1 + \frac{4kE_L}{u^2}\right)^{\frac{1}{2}}\right)\right] \tag{20}$$

where c_x = concentration at distance x (x is positive downstream, and negative upstream)
c_o = initial concentration,
u = mean velocity,
E_L = longitudinal dispersion coefficient, and
k = decay rate.

in the upstream and downstream directions, respectively. Again, dispersion coefficients, if not directly known, can be estimated from similar estuaries, or from empirical formulas, such as those given in Equations (12) and (13).

For multiple pollutant discharges, the resulting concentration curves for each source may be superimposed to give a final composite profile along the estuary (Figure II-9).

Finally, Equation (20) can be used to estimate the length of salinity intrusion by using salt as the constituent and assuming cross-sectional homogeneity and an ocean salinity of 35 ppt (Stommel 1953):

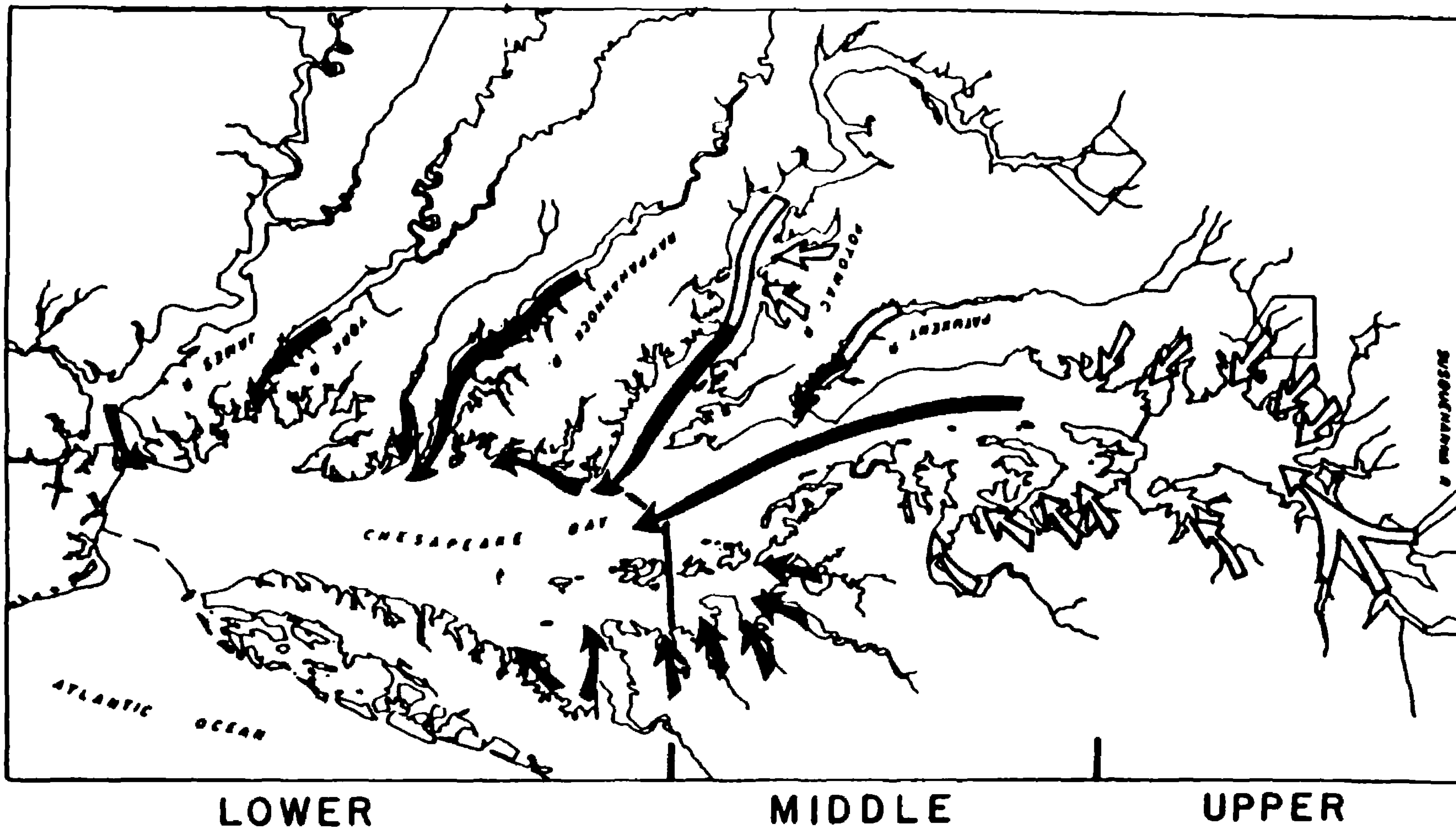


Figure II-9 Pattern of Recent Changes in the Distribution of Submerged Aquatic Vegetation (SAV) in the Chesapeake Bay: 1950-1980. Arrows Indicate Former to Present Limits. Solid Arrows Indicate Areas Where Eelgrass (Zostera Marina) Dominated. Open Arrows Indicate Other SAV Species.
(from U.S. EPA Chesapeake Bay Program, 1982)

$$x = \frac{3.5554\ A\ E_L}{Q_f} \tag{21}$$

where x = length of intrusion from ocean to 1 ppt isohaline,
A = cross-sectional area of estuary,
E_L = longitudinal dispersion coefficient, and
Q_f = freshwater inflow rate.

Such a desktop evaluation of salinity intrusion can be used to relate changes in freshwater inflow to use attainability within the upper estuary.

Other Desktop Evaluations for Use Attainability Assessments

The most common desktop evaluations of use attainability within estuaries are statistical analyses of water quality monitoring data to determine the frequency of violation of criteria for the designated aquatic use. Statistical evaluations of contraventions of water quality criteria should consider the confidence intervals for the number of violations that are attributable to random variations (rather than actual water quality deterioration). For example, consider an estuary monitoring station with 12 dissolved oxygen (DO) observations per year (i.e., a single slackwater sample each month) with a standard of 5 mg/l DO. If statistical analyses of the DO observations indicate that the upper and lower confidence limits for the frequency of random violations of the 5 mg/l DO standard cover a range of 1 to 4 violations per year, a regulatory agency should be cautious in deciding whether actual use impairment has occurred unless more than 4 violations are observed annually.

In addition to the State water quality standard values, both quantitative and qualitative measures should be considered for relationships between water quality criteria and use attainment. Quantitative measures include parametric statistical tests (i.e., assume normal frequency distribution) such as correlation analyses and simple and multiple regression analyses, as well as nonparametric statistical tests (i.e., distribution-free) such as the Spearman and Kendall correlation analysis. These quantitative tests might involve relating water quality indicators (e.g., DO, chlorophyll-a) to use attainability indicators such as juvenile index data (numbers per haul) for different finfish or commercial landings data (tons) for selected fisheries. Qualitative measures include graphical displays of historical trends in water quality and use attainment. For example, a map showing the areas which have experienced a decline in bottom DO conditions during the past 25 years could be overlaid on a map showing areas which experienced a decline in oyster beds over the same period. Another example, which proved to be very persuasive in the recent development of the U.S. EPA Chesapeake Bay management program (U.S. EPA Chesapeake Bay Program, 1982), is described in Figures II-9 through II-12. Figures II-9 and II-10 illustrate the decline in submerged aquatic vegetation (SAV) in Chesapeake Bay during the past three decades. Figures II-11 and II-12 illustrate changes in nutrient enrichment within Chesapeake Bay over the same period. The water quality index plotted in Figure II-12 is based on changes in the concentrations of both nitrogen and phosphorus. As may be seen, the areas of

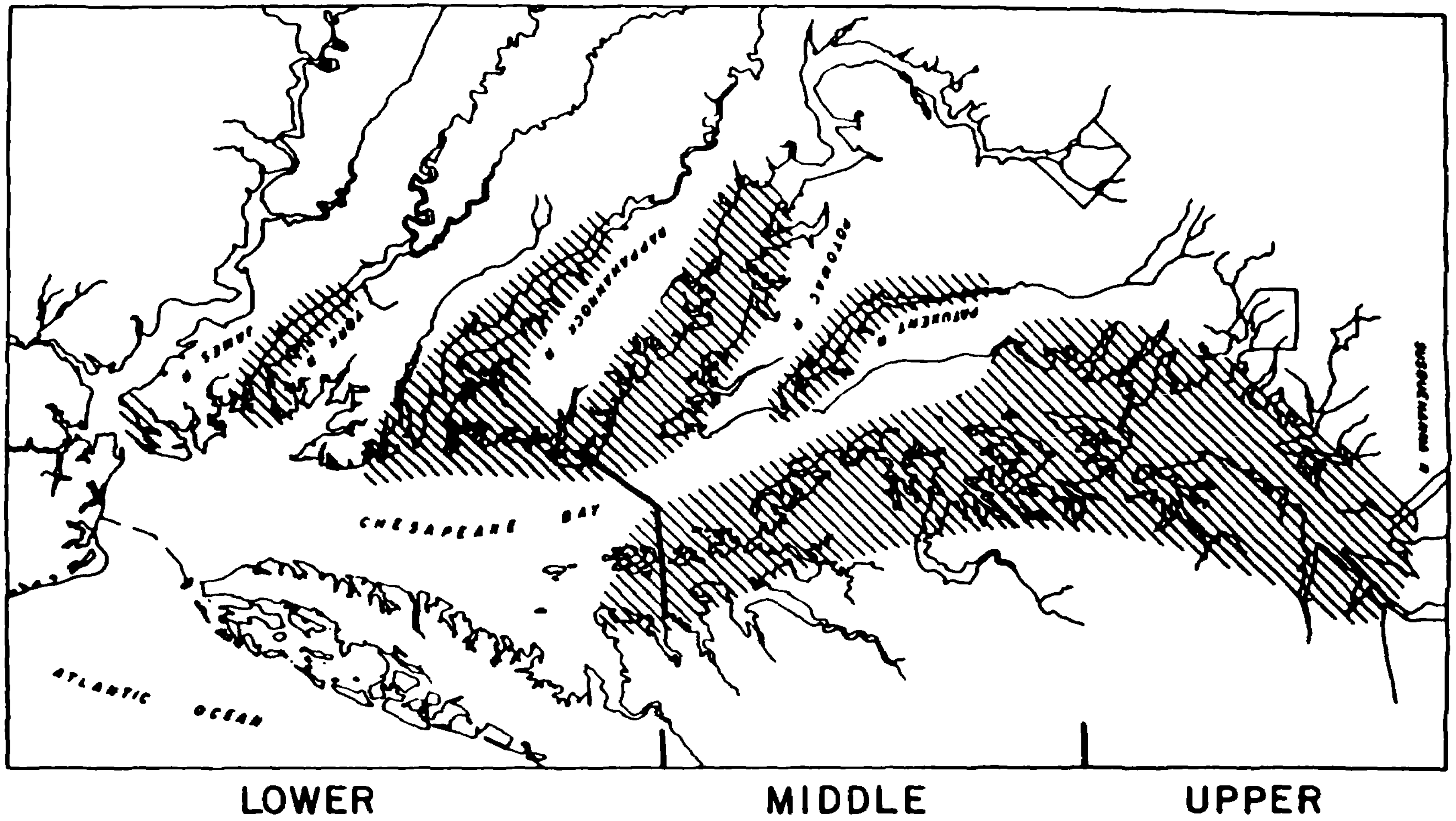


Figure II-10 Sections of Chesapeake Bay Where Submerged Aquatic Vegetation (SAV) has Experienced the Greatest Decline: 1950-1980 (from U.S. EPA Chesapeake Bay Program, 1982)

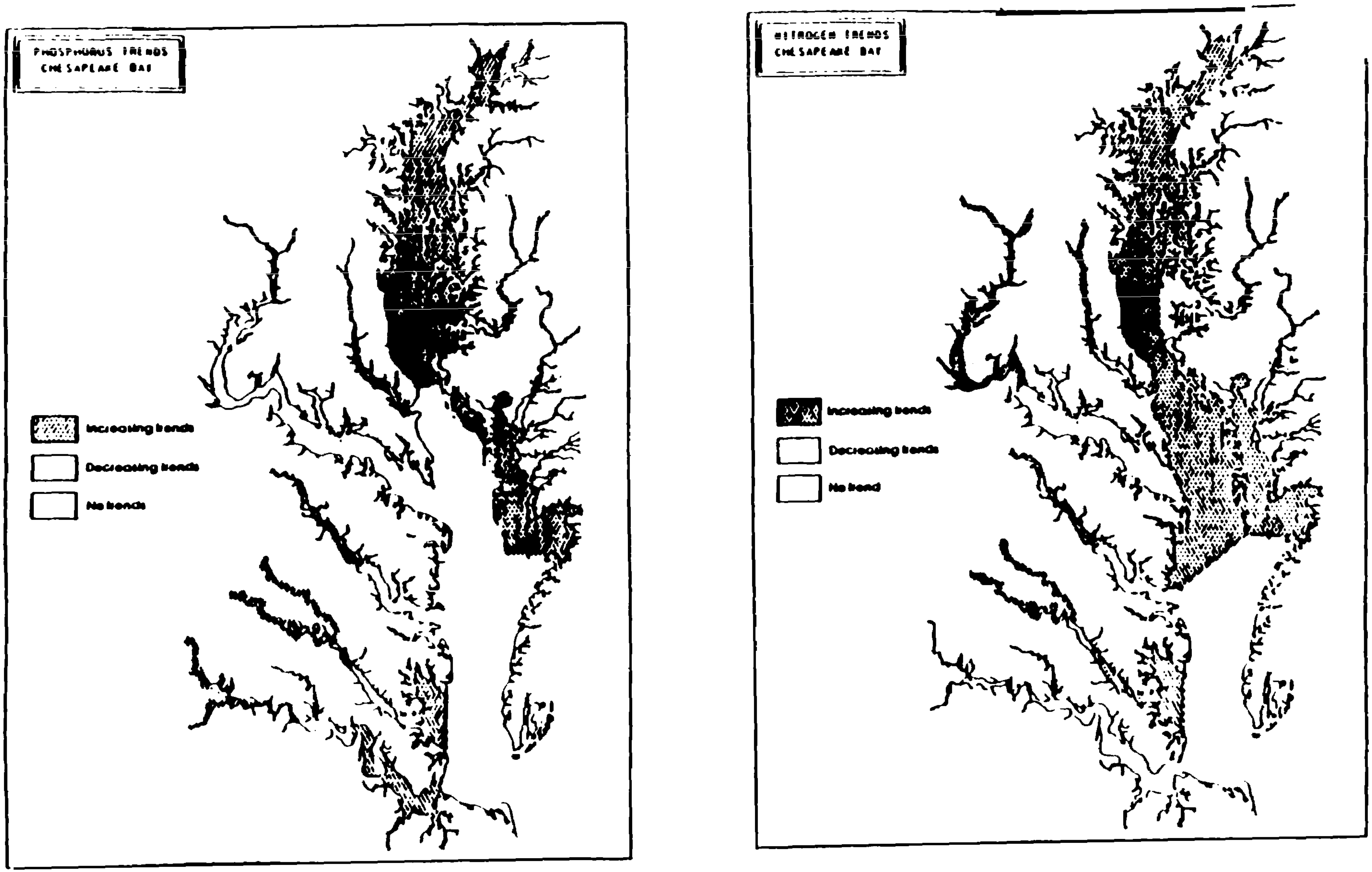


Figure II-21. Phosphorus and Nitrogen Trends in Chesapeake Bay: 1950-1980
(from: EPA Chesapeake Bay Program, 1982)

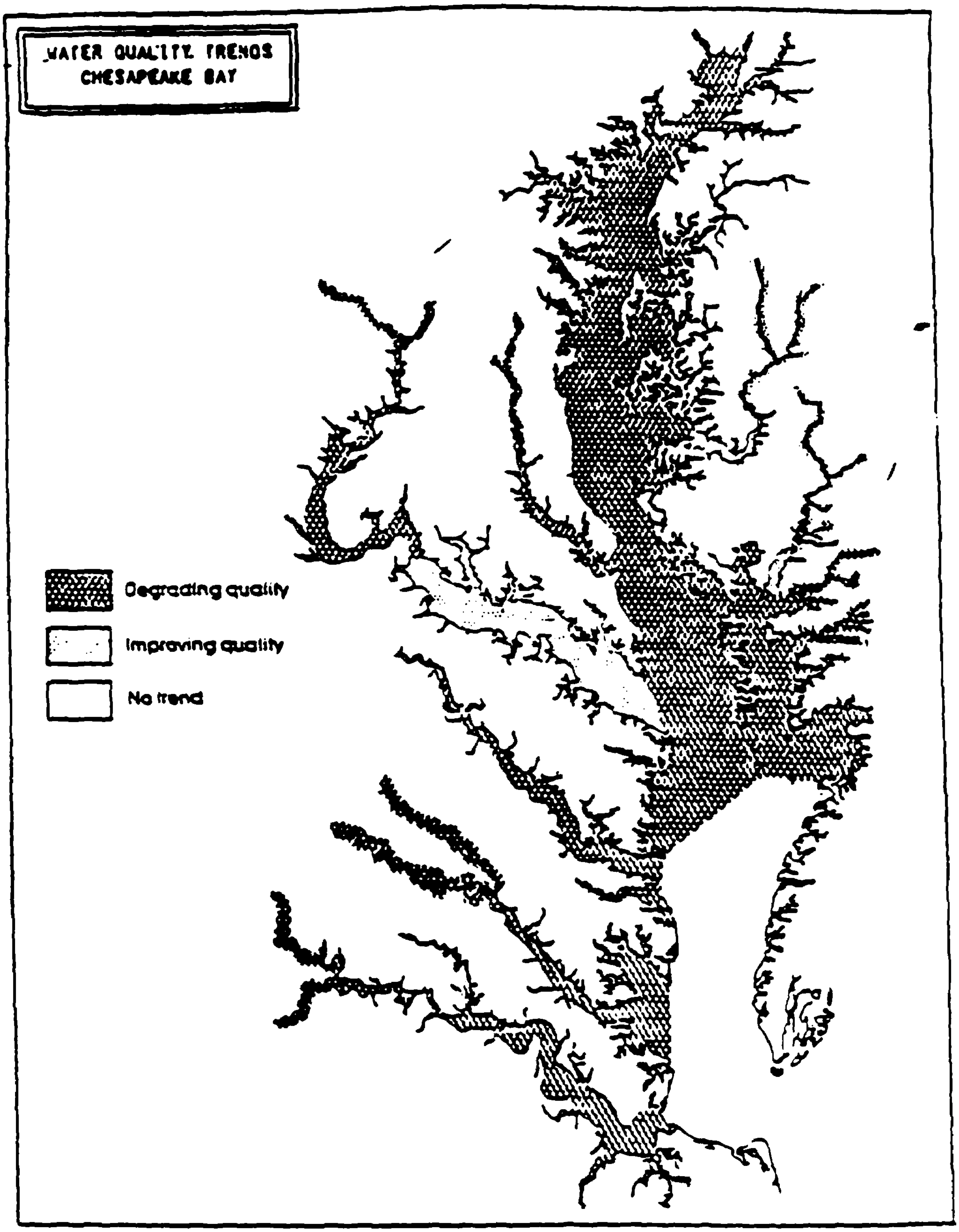


Figure II-12. Water Quality Trends in Chesapeake Bay. If either N or P trends (from Figure II-11) are increasing, then the overall water quality is said to be degrading.

"degrading quality" in Figure II-12 typically correspond to areas where submerged aquatic vegetation has experienced the greatest decline. Based on these types of qualitative comparisons and quantitative evaluations, the U.S. EPA Chesapeake Bay Program has secured considerable State, Federal, and Regional support for more aggressive water quality management efforts to protect Chesapeake Bay. Key to making decisions is the presentation of quantitative data as well as qualitative information.

In developing quantitative and qualitative measures for relationships between water quality and use attainability, care should be taken to distinguish the impacts of pollution discharges from the impacts of non-water quality factors such as physical alterations of the system. For example, in some estuaries, dredging/spoil disposal activities associated with the construction and maintenance of ship navigation channels and harbors may have contributed to use impairment over the years. Among the potential impacts of channel dredging is the reduction in the coverage of SAV's. Therefore, in order to minimize interferences from dredging/spoil disposal, analyses of water quality and use impairment for certain fisheries (e.g., shellfish) and SAV habitats should be based upon periods which do not include major dredging/spoil disposal operations. Another example of physical alterations which should be accounted for in any trend analyses is poor tidal flushing resulting from the construction of bridges and causeways. Potential contributions of extreme meteorologic conditions (e.g., hurricanes, air temperature) to use impairment should also be considered.

If it is determined that some estuary segments exhibit use attainment although violations of water quality criteria occur, the development of site-specific water quality criteria should be considered. Development of site-specific criteria is a method for taking unique local conditions into account. In the case of the water quality indicators (i.e., non-toxicants) being considered in this guidance manual, a potential application of site-specific criteria could be the establishment of temporal dimensions for water quality criteria to restrict use attainment requirements to certain seasons (i.e., in the event that year-round conformance with the water quality criteria is not required to protect the viability of the designated water use).

Computer Modeling Techniques for Use Attainability Evaluations

For many estuaries, field data on circulation, salinity, and chemical parameters may be inadequate for desktop evaluations of use attainability. In these cases, computer-based mathematical models can be used to expand the data base and define causal relationships for use attainability assessments. Specifically, there are three major areas in which computer models of estuaries can contribute to use attainability evaluations:

1. Applications of hydrodynamic and mass transport models can expand physical parameter data bases (i.e., circulation, salinity) in order to identify aquatic use segments and to determine whether physical characteristics are adequate for use attainment.

2. Applications of water quality models can expand chemical parameter (i.e., water quality) data bases in order to determine whether ambient water quality conditions are adequate for use attainment.

3. In cases where use impairment is noted despite acceptable physical characteristics, applications of water quality models can identify the causes of use impairment and alternative control measures that promise use attainment.

The major problem facing the engineer or scientist performing the evaluation is to select the most appropriate numerical model for a given study. Such a selection process must be based on a consideration of system geometry, physical and chemical processes of importance, and the temporal and spatial scales at which the evaluation is being conducted.

Previously discussed were some of the simplifications that can be made to reduce the conceptual complexity of an estuary from its inherently three-dimensional nature. Unfortunately, few quantitative measures exist to define precisely how such determinations should be made. Most criteria for selecting the most appropriate mathematical modeling approach are based on "intuitive judgment" or "experience" with few comparative indices, such as stratification diagrams and numbers, to make the selection less arbitrary.

One particular problem that needs to be addressed is the selection of steady-state versus dynamic approaches to estuarine modeling. Again, intuition leads one to accept that steady-state approaches are fine for rivers or river-flow dominated systems, such as the upper 50-miles of the Potomac River estuary near Washington, D.C. However, for areas further downstream in the estuary where the river flow is less dominant particularly in the dry season, one would intuitively consider using a dynamic approach. The question then is how to formulate a criterion for choosing between steady-state and dynamic modeling approaches. The governing parameters in the selection criteria might be expected to be some combination of freshwater inflow, tidal prism volume, density variations, and tidal period, perhaps in the form of the estuary number, E_D, given by Equation (7) or some other "number." A comparative study of various approaches at differing estuary numbers, E_D, might lead to an empirical formulation of a useful criterion for model selection, similar to the stratification diagram.

Once the appropriate simplifying assumptions have been made, the type of model needed can be determined. There are several model classifications that could be utilized for selection purposes. A four level scheme was used by Ambrose et al. (1981) to classify and compare a number of estuarine receiving water models. The recommended model classification scheme is as follows:

Level 1 - desktop methodologies,
Level 2 - steady-state or tidally averaged models
Level 3 - one-dimensional or quasi-two-dimensional real time models, and
Level 4 - two-dimensional or three-dimensional real time models.

Within each of the four levels, a number of numerical models are listed (Ambrose et al. 1981) and their utility for problem solving is discussed. In actuality, however, there are many more categories, which are subdivisions of the levels suggested by Ambrose et al. (1981). These are summarized in Table II-5 and discussed below, except Level 1 which was previously discussed.

Within Level 2, there are two subdivisions: one-dimensional steady-state models, and two-layer steady-state models. One-dimensional steady-state models assume that the hydraulics are driven entirely by a constant river inflow to the estuary or by net non-tidal (tidally averaged) flow. Conditions are assumed to be uniform over the cross-section, and the effects of Coriolis, wind, tidal, and stratification are neglected. Examples in this category are QUAL II (Roesner et al., 1981) and the WASP models (DiToro et al. 1981).

Two-layer (hydraulic) steady-state models are a simple, but fairly significant extension beyond the one-layer models, in that the advective transport can be resolved to allow for layered residual flow as in the James River. O'Connor et al. (1983) developed such a model to study the fate of Kepone in the James River, in which the net river flow could be specified in the top layer, and the net upstream density-driven flow specified in the lower hydraulic layer. In addition, this model has two sediment layers, one fluid and one fixed, with exchanges between all layers.

In Level 3, models can be subdivided into two categories: one-dimensional real time, and quasi-two-dimensional real time. The category of one-dimensional real-time models has an advantage over steady-state models in that the velocity field simulation can be completely dynamic, allowing tides, wind, friction, variable freshwater inflows, and longitudinal density variations to be included. Again, the estuary is assumed to be cross-sectionally homogeneous.

Quasi-two-dimensional real-time models are an improvement on the one-dimensional real-time representation in that they allow branching systems to be simulated. In addition, the link-node models (such as DEM and RECEIV) can be configured to approximate a two-dimensional horizontal geometry, thus allowing lateral variations to be included in the system evaluation. A very popular model in both these Level 3 categories is the Dynamic Estuary Model (DEM) which represents the geometry with a branching link-node network (Genet et al., 1974). This model is probably the most versatile of its kind and has been applied to numerous estuarine systems, bays, and harbors throughout the world. It contains a hydrodynamic program, DYNHYD, or DYNTRAN (Walton et al., 1983) in its density driven form, and a compatible water quality program, DYNQUAL, which can simulate up to 25 water quality constituents, including four trophic levels.

There are a variety of categories that might be considered in Level 4. Many two-dimensional, vertically-integrated, finite-difference hydrodynamic programs exist. There are, however, relatively few that contain a water quality program that simulates constituents other than salinity and/or temperature (Blumberg, 1975; Hamilton, 1975; Elliot, 1976). These are real time models, assuming only vertical homogeneity (Coriolis effects are now

TABLE II-5. CATEGORIES OF RECEIVING WATER MODELS

LEVEL	CATEGORY	INCLUDES	NEGLECTS	EXAMPLE MODELS
1	Desktop	Uniform flows	Wind, Coriolis, friction, tide Lateral and vertical variations	See text
2	1-D, steady-state	River flows Longitudinal variability	Wind, Coriolis, friction, tide Lateral and vertical variations	QUAL II WASP
2	2-layer, steady-state	River flows Residual upstream flows Longitudinal and vertical variability	Wind, Coriolis, friction, tides Lateral variations	O'Connor et al. (1983)
3	1-D real time	Tides, wind, river flows, friction Longitudinal variability	Coriolis Lateral and vertical effects	DEM RECEIV
3	Quasi 2-D real time	Tides, wind, river flows, friction Longitudinal and lateral variability	Coriolis, lateral momentum transfer Vertical variations	DEM RECEIV
4	2-D, finite-difference vertical integrated	Tides, wind, river flows, friction Coriolis Longitudinal and lateral variability	Vertical variations	Ross and Jerkins (1983)
4	2-D, finite-element vertically integrated	Tides, wind, river flows, friction Coriolis Longitudinal and lateral variability	Vertical variations	CAFE1/DISPER1 CBCM Chen (1978)
4	2-D, finite-difference laterally integrated	Tides, wind, river flow, friction Coriolis Longitudinal and vertical variability	Coriolis Lateral variations	CBCM
4	3-D	All physical processes	--	CBCM Leendertse et al. (1973)

included). An example of a water quality model in this category is the hydrodynamic and water quality model developed by Ross and Jerkins (1983) which has been extensively applied to Tampa Bay.

Similar to the above category are the two-dimensional, vertically-integrated, finite-element models. The physical process and simplifications are identical. The difference is that the geometry is represented as a series of elements (usually triangles) which can better represent complex coastlines. Examples of models in this category are the CAFE1/DISPER1 hydrodynamic models (Wang and Connor 1975; Leimkuhler 1974), the Chesapeake Bay Circulation Model, CBCM (Walton et al., 1983), and a water quality model developed by Chen (1978). The first two models can simulate only mass transport of a non-conservative constituent, whereas Chen's model is capable of representing most major water quality processes. CBCM has the additional advantages of a three-dimensional form and the capability to link 1-2 or 2-3-dimensional models to treat tributaries from a main bay or subgrid scale cuts in a main bay which cannot be resolved adequately at the horizontal spatial scale.

There are a number of two-dimensional, laterally-averaged models (longitudinal and vertical transport simulations) that treat mass transport of salt and temperature, but very few that include nonconservative constituents or water quality routines. While models in this category assume lateral homogeneity and neglect Coriolis effects, they can represent vertical stratification although for numerical reasons, care should be taken in defining vertical layers to represent the saltwater/freshwater interface of high stratified systems. The tributary submodels of CBCM (Walton et al., 1983) are included in this category.

Last is the category of three-dimensional, finite-difference and finite-element models. These models allow all physical processes to be included, although many were developed for systems of constant salinity (lakes or oceans) which cannot simulate stratification processes. Models in this category include CBCM (Walton et al. 1983) and the models of Leendertse et al. (1973) which simulate hydrodynamics and the transport of salt, temperature, and other conservative constituents.

Sample Applications of Estuary Models

Delineation of Aquatic Use Segments. Figure II-7 illustrates the use of measured data on physical parameters to delineate homogeneous aquatic use segments in Chesapeake Bay. For many estuaries, the measured data on circulation and salinity will not have sufficient spatial and temporal coverage to permit a comprehensive analysis of use attainability zones. In cases where the measured data base is inadequate, computer models can be used to expand the physical parameter data bases for segmentation of the estuary.

Figure II-13 illustrates the use of model projections for Tampa Bay, located on the Gulf Coast of central Florida, to delineate relatively homogeneous segments for use attainability evaluations (Camp Dresser & McKee, Inc. 1983). Tampa Bay is considerably smaller and shallower than Chesapeake Bay, with a surface area (approx. 350 sq. mi.) that is less than 10 percent of the Maryland/Virginia estuary's (approx. 5,000 sq. mi.

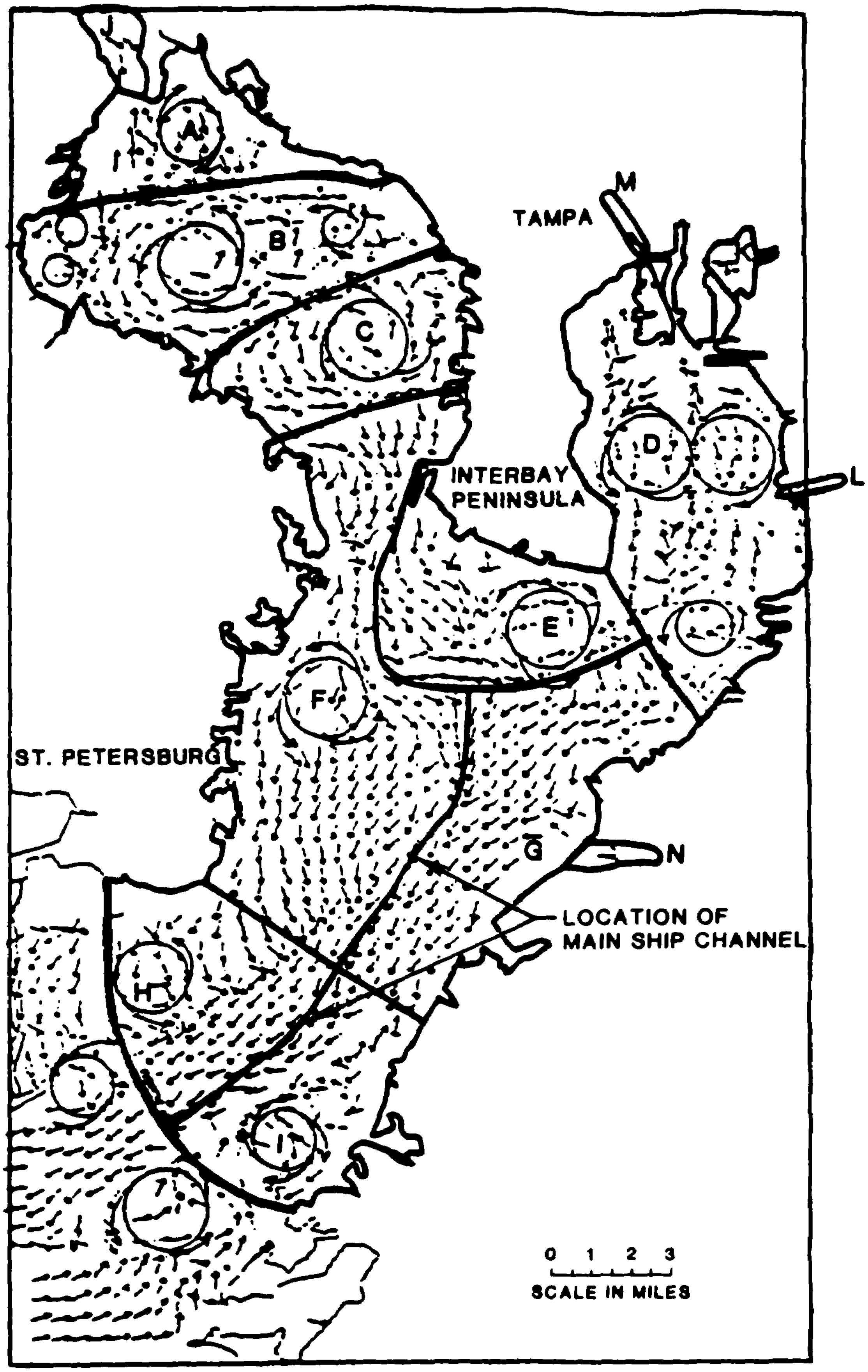


Figure II-13. Map of Tampa Bay Showing Sample Estuary Segments (A through N) and Net Current Velocities for a Single Tidal Cycle (from Camp Dresser and McKee 1983) including tributaries). The Tampa Bay estuary exhibits extremely diverse and abundant marine life which has been attributed to the geographic position of the estuary between temperate and subtropical waters. As a result of Tampa Bay's location, winter water temperatures rarely fall to levels which could kill tropical organisms and summer water temperatures are moderate enough to be tolerated by many of the temperate species. Another contributing factor to the diversity and abundance of Tampa Bay marine life is that salinity is typically in the range 25-35 ppt over most of the estuary, without the wide fluctuations and significant vertical stratification that characterize many other estuaries. As a result of the stability of the salinity regime, many ocean species can coexist with typical estuarine species.

Tampa Bay's salinity regime is also much different from Chesapeake Bay's. Whereas extensive areas in Chesapeake Bay exhibit vertical stratification, Tampa Bay is very well-mixed vertically due in large part to its relatively shallow mean depth (i.e., relationship of storage volume to surface area). Unlike Chesapeake Bay where circulation and mass transport must be evaluated in the vertical as well as horizontal and longitudinal directions, only the horizontal and longitudinal directions need to be considered for Tampa Bay evaluations. Therefore, the sample analysis of Tampa Bay is a good example of a segmentation approach to an estuary where the use is not significantly influenced by vertical stratification. It is also a good example of how an estuary circulation model can be used to segment an estuary for use attainability analyses.

The estuary segment boundaries shown in Figure II-13 have been delineated on a map of Tampa Bay showing circulation model projections of net current velocities (i.e., magnitude and direction) for a single tidal cycle. The model projections are based upon a two-dimensional circulation model (horizontal and longitudinal directions) which had previously been calibrated to measured current velocity and tidal elevation data for Tampa Bay (Ross and Jerkins, 1978). The use of the model expanded the available circulation data base from a limited number of gaging stations to comprehensive coverage of the entire Bay. One of the most important factors in subdividing the Tampa Bay estuary system into relatively homogeneous subunits is the ship navigation channel extending from the mouth of the Bay to the vicinity of Interbay Peninsula with branches extending into Hillsborough Bay (segment D) and into the lower end of Old Tampa Bay (segment C). As may be seen from the convergence of velocity vectors in the vicinity of the navigation channel, there tends to be relatively little mixing between waters on either side of the Main Bay channel. Therefore in Figure II-13, the navigation channel and the adjoining dredge spoil areas serve as the approximate boundary between segments H and I and between segments F and G. Each of these segments appears to be relatively isolated from its counterpart on the opposite side of the navigation trench before mixing occurs in the vicinity of the navigation channel, thereby justifying the designation of each as a separate segment. Water movement is also somewhat isolated on approximately either side of the navigation channel branches extending into Hillsborough Bay and the lower end of Old Tampa Bay. However, since net current velocities tend to converge a short distance south of the two ship channel branches, the boundaries between segments E and F and E and G in Figure II-13 depart somewhat from the navigation trench.

Another circulation factor considered in the delineation of estuary segments is the impact of causeways and bridges on tidal flushing. Based upon the circulation patterns shown in Figure II-13, it seems appropriate to assign separate segment designations (A, B, and C) to the areas above the three bridge crossings in Old Tampa Bay: Courtney Campbell Causeway (boundary between segments A and B), Howard Franklin Bridge (boundary between segments B and C) and Gandy Bridge (boundary between segments C and F). Likewise, McKay Bay (segment K), which is separated from Hillsborough Bay by the 22nd Street Causeway, also merits a separate segment designation.

A final circulation factor in the open bay is the location of net rotary currents (indicated by circles in Figure II-13) which are called "gyres." The gyres result from water moving back and forth with the tides, while following a net circular path. Gyres can have a significant effect on flushing times, since waters caught in the gyres typically exhibit much higher residence times than waters which are not affected by these areas of net rotary currents. The use of the main ship channel and causeway/bridge crossings as segment boundaries in Figure II-13 has generally isolated the major gyres or groups of gyres. Further subdivision of the Hillsborough Bay segment (D) to isolate the waters on the eastern and western sides of the ship channel (which bisects segment D) does not appear to be warranted because of the two gyres in the middle section of the Bay and the gyre in lower Bay. In other words, the gyres in Hillsborough Bay are indicative of an irregular circulation pattern that seems to mix waters on both sides of the ship channel. Likewise, the gyres within segment B are indicative of a circular mixing pattern throughout the segment which suggests that further subdivision into eastern and western sections is not justified.

The segment network in Figure II-13 also maintains relatively homogeneous salinity levels within each segment. The greatest longitudinal variations in salinity occur in segments F and G which exhibit 3-5 ppt increases in average annual values between the upper and lower ends of the segment. If these longitudinal variations in salinity will result in significant differences in the biological community, further subdivision of segments F and G should be considered.

Figure II-13 also shows five separate segments for significant embayments: Safety Harbor (J), McKay Bay (K), Alafia River (L), Hillsborough River (M), and Little Manatee River (N). The latter three represent the tidal sections of the indicated river. In addition to these five embayments there may be other inlets which should be separated from Tampa Bay segments for separate use attainability studies.

In summary, the network shown in Figure II-13 illustrates how hydrodynamic and salinity data produced by an estuary model can be used to segment the Tampa Bay system. In addition to the type of hydrodynamic data shown in Figure II-13, the estuary model can be used for "particle tracer" studies that can further address issues such as mixing of waters on either side of the ship channel and the impacts of gyres.

<u>Evaluation of Use Attainment Based Upon Ambient Water Quality Data.</u> It is often the case that the measured ambient water quality data base is inadequate from temporal and/or spatial standpoints for a definitive assessment of use attainment.

An example of temporal limitations is an ambient water quality data base that suffers from a small sample size (e.g., 6-12 slackwater observations at each station per year), thereby resulting in extremely wide confidence intervals for the number of violations of standards and criteria that are attributable to random variations (rather than actual water quality deterioration).

Another example of temporal limitations is an observed water quality data base that is restricted to a single daytime observation on each sampling day. This type of data base may not provide any insights into diurnal variations in DO which can result in use impairment, since nighttime DO's can be significantly lower than daytime values due to diurnal variations in algal production/respiration.

An example of spatial limitations in the measured water quality data base is inadequate coverage of longitudinal and/or horizontal variations in water quality. Adequate longitudinal coverage is required in all estuaries to assess the significance and spatial extent of maximum and minimum concentrations in the estuary. Adequate horizontal coverage is required in relatively wide estuaries where horizontal transport processes are significant.

Another example of spatial limitations would be the collection of surface water samples only within an estuary which exhibits extensive areas of vertical stratification. The lack of bottom water samples may prevent an adequate assessment of use attainment, since potential depressions of bottom water DO levels cannot be evaluated.

In cases where the measured water quality data base is inadequate from either temporal or spatial standpoints, an estuary model should be used to expand the data base for use attainability evaluations. The model must first be calibrated with the available measured data base to demonstrate that its representation of the prototype produces water quality statistics that are not significantly different from the measured statistics. The reliability of the estuary model projections depends upon the amount and type of measured data available for model calibration. If the measured data base provides reasonably good coverage of spatial and temporal (e.g., both short-term and long-term) variations in water quality, projections by a model calibrated to this data base should be quite reliable in a statistical sense. If the measured data base used for calibration is quite limited, estuary model projections will be less reliable; however, the application of an appropriate model to an estuary with limited measured data can still provide significant insights for use attainability evaluations and considerable guidance for future estuary monitoring programs.

To illustrate the use of an estuary model for use attainment evaluations, a sample application of a one-dimensional (1-D) model to Naples Bay, Florida is described below (Camp Dresser & McKee, Inc. 1983). Naples Bay (see Figure II-14) is a rather small estuary (less than 1.5 sq. mi. surface

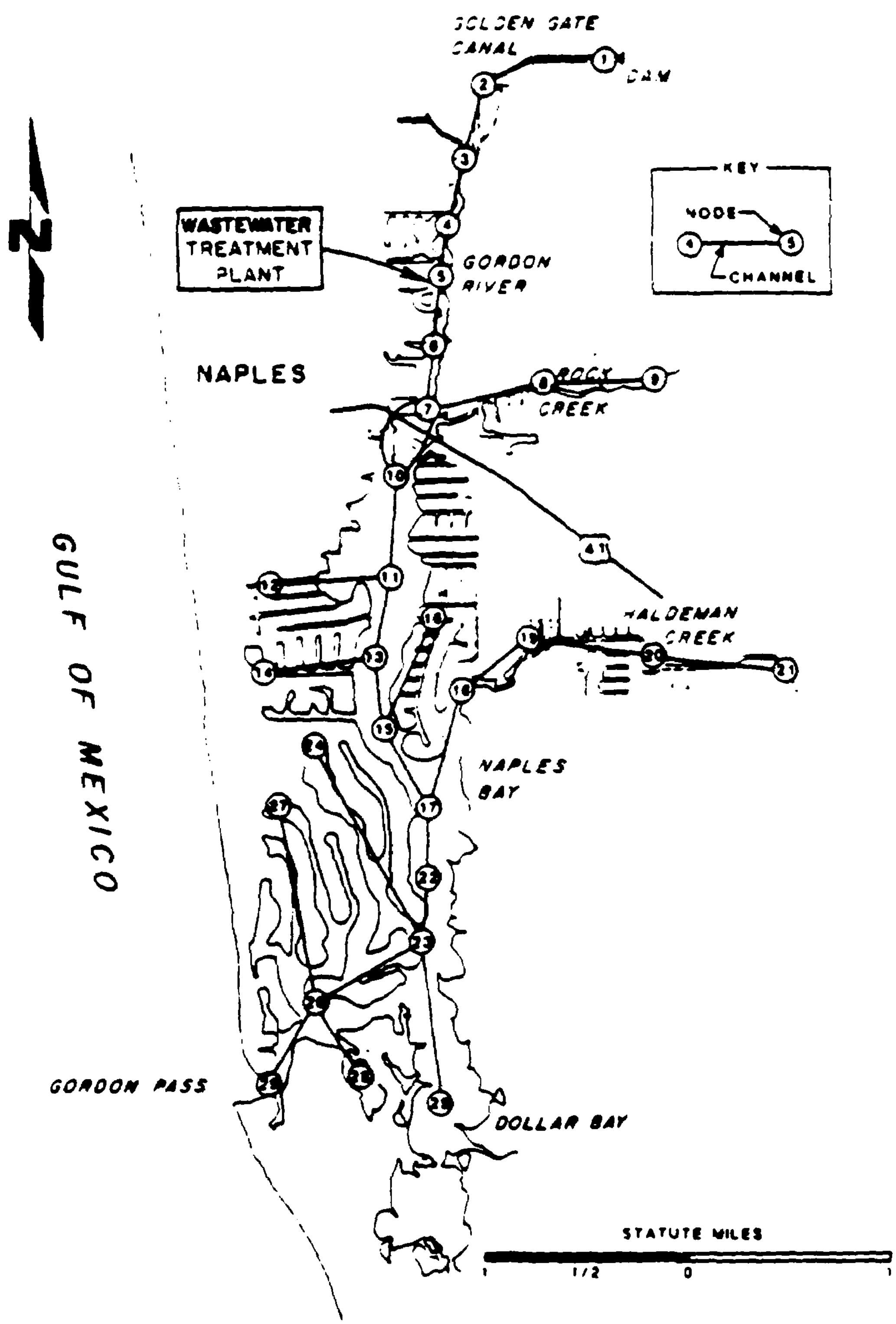


Figure II-14. Node and Channel Network for the Naples Bay DEM model.

area) located on the Gulf Coast of southeastern Florida. The City of Naples' municipal wastewater treatment plant (secondary treatment) which discharges to the Gordon River portion of the Naples Bay estuary, is the only major point source of pollution. This sample application illustrates the impacts of an 8.0 million gallons per day (mgd) discharge from the Naples wastewater treatment plant. Nonpoint pollution loadings are contributed by rainfall runoff and groundwater recharge from a 155 sq. mi. drainage area, the majority of which discharges to the estuary at the uppermost point in the system (node no. 1 in Figure II-14). The Gulf of Mexico boundary condition (introduced at node no. 29 in Figure II-14) also contributes nutrients and other constituents to the lower Bay. Since the Naples Bay system is a relatively narrow and shallow estuary, it was assumed that a 1-D model which only represents longitudinal transport would be adequate for this water quality evaluation (i.e., horizontal and vertical gradients are neglected). A schematic of the 1-D representation of the Naples Bay system with the Dynamic Estuary Model (DEM) is shown in Figure II-14.

As indicated in the earlier section on modeling techniques, the DEM model (Genet et al., 1974) applied to Naples Bay is one of the most widely used estuary models in the U.S. DEM provides a representation of intertidal hydrodynamics and mass transport with computation intervals which are typically less than one hour. The model simulates 1-D flow, mass transport, and water quality processes in a network of channels connected by junctions called "nodes." As shown in Figure II-14, the DEM model network applied to Naples Bay consists of 29 nodes and 28 channels. This network includes all the appropriate conveyance and storage features of the prototype system, including bifurcation around an island (between nodes 7 and 10), and the canal system adjacent to the main water body. Streamflows, wastewater discharges, and associated pollutant loadings are added to the system at the nodes. Based upon a set of motion equations solved for the channels and a set of continuity equations solved for the nodes, the hydrodynamic portion of the model calculates flows and velocities in the channels and water surface elevations at the nodes. An accurate representation of hydrodynamic processes within the system is developed to adequately model mass transport and water quality processes.

The output from the hydrodynamic model becomes input to the water quality model which calculates mass transport between nodes and calculates changes in concentration due to physical, chemical and biological processes. Water quality processes represented by this portion of the model include: mass transport based upon advection and dispersion, BOD decay, nitrification, algal productivity, benthic sources of pollutants, dissolved oxygen sources and sinks, and fecal coliform die-off.

Following calibration and verification of the Naples Bay model with measured hydrodynamic and water quality data, the model was used to assess estuary-wide water quality. Figure II-15 shows the model projections of wet season chlorophyll-a (i.e., phytoplankton concentrations) for secondary treatment operations which were in effect at the Naples wastewater treatment plant. As indicated in an earlier section, chlorophyll-a is an important indicator of estuary health for use attainability evaluations.

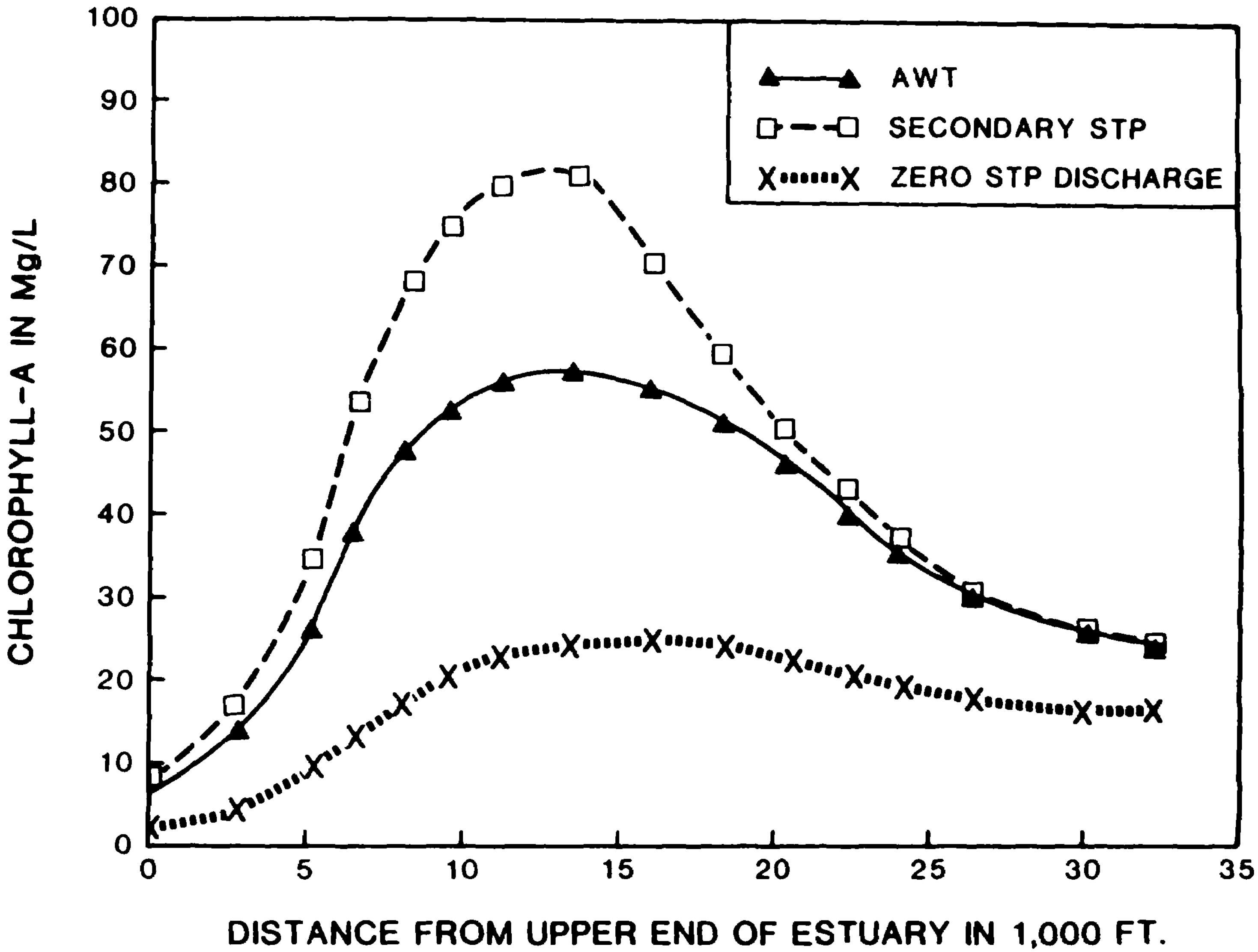


Figure II-15 Comparison of Simulated Average Daily Chlorophyll-a in Main Stem of Naples Bay Projected for Different Wastewater Discharge Scenarios: "Worst Case" Wet Season Conditions

The chlorophyll-a simulations shown in Figure II-15 represent "worst case" water quality conditions at the start of the wet season (i.e., 4-month period of significant rainfall and high streamflow). As may be seen from the plot of "Secondary STP" conditions along the main stem of the Bay, the combination of point and nonpoint source loadings of nitrogen and phosphorus under wet season conditions results in chlorophyll-a levels exceeding 50 ug/l for almost 3.0 miles and maximum values on the order of 80 ug/l for about 1.0 mile. The volume-weighted mean chlorophyll-a (i.e., weighted by the storage volume in each estuary segment) for the upper two miles (i.e., Gordon River) of the estuary is about 60 ug/l, while the volume-weighted mean for the entire estuary is about 45 ug/l. These maximum and mean concentrations can be compared with state or regional water quality criteria for local use attainability evaluations. Additional model projections can be developed for other wet season and dry season conditions to evaluate the frequency of use impairment expressed in terms of ambient water quality. Since chlorophyll-a impacts are primarily of interest in terms of associated impacts on DO, the estuary model can also be used to evaluate diurnal DO impacts for use attainability assessments. Once chlorophyll-a and DO relationships have been evaluated, the estuary model can be used to evaluate nitrogen and phosphorus goals that maintain chlorophyll-a at levels ensuring use attainment.

Evaluations of Use Impairment Causes and Alternative Controls. Estuary models are probably most useful for management evaluations following a determination of use impairment in certain sections of the estuary. Models can be used to define the causes of impairment and to define the effect of alternate controls on attaining the use. Such analyses require the development of causal relationships between pollution loadings, physical modifications and the resulting changes in uses. It is very difficult to develop such causal relationships from statistical analyses of measured data. For example, regression equations can merely indicate that pollution loadings and impairment of the uses appear to be correlated based upon the measured data base. Such regression equations should not be interpreted as definitive indications of cause-effect relationships. Evaluations of cause-effect relationships require the use of a deterministic estuary model.

Evaluations of use impairment causes will typically focus on comparisons of point and nonpoint source pollution impacts. The estuary model is well-equipped to perform such evaluations because both point and nonpoint source loadings can be "shut off" (i.e., deleted from the system) for evaluations of relative contributions to use impairment. Applications of the Naples Bay model will be used to illustrate how evaluations of cause-effect relationships can be performed. After analyses of the impacts of existing secondary treatment operations at the 8.0 mgd wastewater treatment plant, the Naples Bay model was rerun with no wastewater discharges. For this model run, the only sources of nutrients and other constituents were nonpoint source flows from the Bay's 155 sq. mi. drainage area and ocean boundary conditions at the mouth of the Bay. The resulting chlorophyll-a projection for "worst case" wet season conditions are shown in Figure II-15 as the "Zero STP Discharge" plot. As may be seen, the maximum chlorophyll-a concentration is about 25 ug/l, with concentrations on the order of 15-25 ug/l for about 5.0 miles. The chlorophyll-a concentrations for the "Zero STP Discharge" condition are typically only 25-50 percent of the existing

"Secondary STP" levels for about 5.0 miles. Also, the location of the maximum chlorophyll-a concentration is shifted about 1.0 mile further downstream for the "Zero STP Discharge" condition. The mean volume-weighted chlorophyll-a for the entire Bay is approximately 20 ug/l which is less than half of the "Secondary STP" mean. These evaluations suggest that secondary effluent discharges from the wastewater treatment plant are the major cause of relatively high chlorophyll-a levels under wet season conditions. Approximately 50-55 ug/l or about 70 percent of the peak chlorophyll-a concentration (80 ug/l) and about 25 ug/l or 55 percent of systemwide volume-weighted mean concentration can be attributed to the wastewater treatment plant.

Chlorophyll-a is a specific index of phytoplankton biomass. Thus, assuming that the chlorophyll-a levels associated with the "Secondary STP" condition indicate use impairment, the estuary model provides a mechanism for evaluating the use attainability benefits of alternate controls. The Naples Bay model was rerun with the 8.0 mgd discharge upgraded to advanced wastewater treatment (AWT) levels. The simulated AWT upgrading involved reducing total phosphorus effluent levels from 7.0 mg/l to 0.5 mg/l as P, the achievement of almost total nitrification in comparison with less than 50 percent nitrification for secondary treatment conditions, and reducing 5-day biochemical oxygen demand (BOD) from 20 mg/l to 5 mg/l. Nonpoint source loadings and ocean boundary conditions were set at the same levels as the "Secondary STP" model runs. As shown in Figure II-15, the projected chlorophyll-a concentrations for the "AWT" conditions are 20-30 percent lower than the "Secondary STP" levels for approximately a two mile section that includes the maximum concentrations for both scenarios. The AWT scenario's maximum concentrations of chlorophyll-a are on the order of 50-60 ug/l for about 2.5 miles, while the volume-weighted mean concentration for the entire Bay system is about 40 ug/l. Even under AWT conditions, the maximum chlorophyll-a levels for AWT conditions are still about 35 ug/l greater than the maximum values for "Zero STP Discharge" conditions.

The maximum and mean concentrations for AWT conditions can be compared with water quality criteria to determine if this control measure can achieve use attainment. If the projected chlorophyll-a reductions are not sufficient to prevent use impairment, the model can be rerun to assess the use attainability benefits of nonpoint source controls in addition to AWT implementation.

ESTUARY SUBSTRATE COMPOSITION

The bottom of most estuaries is a mix of sand, silt and mud that has been transported and deposited by ocean currents or by freshwater sources. Rocky areas may also be seen, particularly in the fjord-type estuary. None of these substrate types are particularly hospitable to aquatic plants and animals, which accounts in part for the paucity of species seen in an estuary.

Much of the estuarine substrate is in flux. The steady addition of new bottom material, transported by currents, may smother existing communities and hinder the establishment of new plants and animals. Currents may cause a constant shifting of bottom sediment, further hindering the colonization of species. Severe storms or flooding may also disrupt the bottom.

The sediment load introduced at the head of the estuary will be determined by the types of terrain through which the river passes, and upon land use practices which may encourage runoff and erosion. It is important to take land use practices into consideration when examining the attainable uses of the estuary. The heavier particles carried by a river will settle out first when water velocity decreases at the head of the estuary. Smaller particles do not readily settle and may be carried a considerable distance into the estuary before they settle to the bottom. The fines may never settle and will contribute to the overall turbidity which is characteristic of estuaries.

It is often difficult for plants to colonize estuaries because they may be hindered by a lack of suitable anchorage points, and by the turbidity of the water which restricts light penetration (McLusky, 1971). Attached plant communities (macrophytes) develop in sheltered areas where silt and mud accumulate. Plants which become established in these areas help to slow prevailing currents, leading to further deposition of silt (Mann). The growth of plants often keeps pace with rising sediment levels so that over a long period of time substantial deposits of sediment and plant material may be seen.

Attached plant communities, also known as submerged aquatic vegetation (SAV), serve very important roles as habitat and as food source for much of the biota of the estuary. Major estuary studies, including an intensive years-long study of the Chesapeake Bay, have shown that the health of SAV communities serves as an important indicator of estuary health. Although excess siltation may have some adverse effects on SAV, as discussed above, this problem is minor compared to the effects of nutrient and toxics loadings to the estuary. When SAV communities are adversely affected by nutrients and/or toxics, the aquatic life uses of the estuary also will be affected. The ecological role of SAV in the estuary will be discussed further in Chapter III, and its importance to the study of attainable uses in Chapter IV.

Sediment/substrate properties are important because such properties: (1) determine the extent to which toxic compounds in sediments are available to the biota; and (2) determine what types of plants and animals may become established. The presence of a suitable substrate may not be sufficient, however, since nutrient, DO, and/or toxics problems may cause the demise and prevent the reestablishment of desirable plants and animals. Therefore, characterization of the substrate is important to a use attainability study in order to understand what types of aquatic life should be expected in a given area.

ADJACENT WETLANDS

Tidal and freshwater wetlands adjacent to the estuary can serve as a buffer to protect the estuary from external phenomena. This function may be particularly important during wet weather periods when relatively high streamflows discharge high loads of sediment and pollutants to the estuary.

The volume of sediment carried by streamflow during wet weather periods is substantially greater than the amount transported into the estuary by rivers and streams during dry weather periods. Such shock loads could quickly smother plant and animal communities and jeopardize their survival. Wetlands can serve an important function by protecting the estuary from such shock loads. Because of the sinuous pattern of streams that flow through the wetlands, and the high density of plants, water velocities will be reduced enough to allow settlement of a substantial proportion of the sediment load before it reaches the estuary. This simultaneously protects the estuary and contributes to the maintenance of the wetlands.

The sediment load discharged by streamflow may be accompanied by nutrients and other pollutants. Excessive loadings of nutrients such as nitrogen and phosphorus may promote eutrophication and the growth of algal mats in the estuary, which is undesirable from both aquatic use and aesthetic standpoints. On the other hand, these nutrients are beneficial to the maintenance of plant life in the wetland.

Another important function of a wetland is to reduce peak streamflow discharges into the estuary during wet weather periods. To the extent that this peak flow attenuation prevents abrupt changes in salinity, the flora and fauna of the estuary are protected. It has been common practice to straighten existing channels and cut new channels in wetlands to speed drainage and enable the use of wetlands for agriculture or other development. Such channelization may diminish the protective functions of the wetland and have an adverse impact on the health of the estuary.

While the wetland may help to withhold nutrients in the form of nitrogen and phosphorus from the estuary, it serves as a major source of nutrients in the form of detritus. A substantial portion of dead plant material in the wetland is transported to the estuary as detritus. Detritus is a basic fuel of the estuary, serving as the main source of nutrient for filter feeders and many fish at the bottom of the food chain. The estuary is highly productive, more so than the freshwater or marine environment, because of this source of nutrients.

Since the alteration or destruction of wetlands may hold important implications for the health of the estuary, it is important during the course of a water body survey to examine historical trends in the wetland acreage, locations, and characteristics for clues which explain changes in the estuary and its uses. The extent to which wetlands have been irreversibly altered may establish bounds on the uses that might be expected. Conversely, restoration of wetlands may provide some means of restoring uses provided that other conditions such as toxic or nutrient loadings are not a problem, or some other irreversible change has not been made to the estuary.

HYDROLOGY AND HYDRAULICS

There are two important sources of freshwater to the estuary-streamflow and direct precipitation. In general, streamflow represents the greatest contribution to the estuary and direct precipitation the smallest.

The location of the salinity gradient in the river controlled estuary is to a large extent an artifact of streamflow. The location of salinity isoconcentration lines may change considerably, depending upon whether streamflow is high or low. This in turn may affect the biology of the estuary, resulting in population shifts as biological species adjust to changes in salinity.

Most species are able to survive within a range of salinity levels, and therefore most aquatic uses may not be adversely affected by minor shifts in the salinity gradient. Most of the biota can also sustain temporary extreme changes in salinity, either by flight or through some other mechanism. For example, molluscs may be able to withstand temporary excursions beyond their preferred salinity range by simply closing themselves off from their environment. This is important to their survival since the adult is unable to relocate in response to salinity changes. However, molluscs cannot survive this way indefinitely.

Generally speaking, the response of a stream or estuary to rainfall events depends upon the intensity of rainfall, the drainage area affected by the rainfall and the size of the estuary. Movement of the salt front is dependent upon tidal influences and freshwater flow to the estuary. Variations in salinity generally follow seasonal patterns such that the salt front will occur further down-estuary during a rainy season than during a dry season. The salinity profile may also vary from day to day reflecting the effect of individual rainfall events, but may also undergo major changes due to extreme meteorological events.

The location of the salt front in a small estuary may be easily displaced but rapidly restored in response to a rainstorm, whereas the effect of the same size storm on salinity distribution within a larger estuary may be minor. For a large system, the contribution of a given storm may be only a fraction of the overall freshwater flow and thus will have no appreciable effect. For a small system the contribution of a given storm may be very large compared to overall flow, and the system will respond accordingly.

A rapid increase in flow may have several deleterious effects on a small estuary: (1) the salinity gradient changes drastically, placing severe stress on non-motile species and forcing the migration of motile forms, (2) a sediment and pollutant load which is too large to be captured by surrounding wetlands may be transported into the estuary, and (3) the bottom may be scoured in areas of high flow velocity, destroying floral and faunal communities and existing habitat, and eliminating the conditions that would be required for replacement communities to become established.

Major shifts in salinity due to extreme changes in freshwater flow are not uncommon. An excellent example is the impact of Hurricane Agnes on the Chesapeake Bay in 1972. The enormous and prolonged increase in freshwater flow to the Bay shifted the salinity gradient many miles seaward and had a devasting effect on the shellfish population. The flow was so great that salinity levels did not return to normal for several months, a period far longer than non-motile species would be able to survive such radical reductions in salinity. In addition, the enormous quantities of sediment delivered to the Bay by Hurricane Agnes exerted considerable stress on the Bay environment.

Anthropogenic activity may also have a significant effect on salinity in an estuary. When feeder streams are used as sources of public water supply and the withdrawals are not returned, freshwater flow to the estuary will be reduced, and the salt wedge found further up the estuary. If the water is returned, usually in the form of wastewater effluent, the salinity gradient of the estuary may not be affected although other problems might occur which are attributable to nutrients and other pollutants in the wastewater.

Even when there is no appreciable change in annual freshwater flow or quality due to water supply uses, the salinity profile may still be affected by the way in which dams along the river are operated. Flood control dams may result in controlled discharges to the estuary rather than relatively short but massive discharge during high flow periods. A dam which is operated so as to impound water for adequate public water supply during low-flow periods may severely alter the pattern of freshwater flow to the estuary. Although annual input to the estuary may remain unchanged, seasonal changes may have a significant impact on the estuary and its biota.

The discussion of hydrology, meteorology and the effect of hydraulic structures in this section provides only an overview of their possible effects on the health of an estuary. Hydrologic impacts will depend upon the unique physical characteristics of the estuary and its feeder streams, including structural activity that may have changed flow characteristics to the estuary. Extreme rainfall events are particularly important because they may result in physical damage to wetlands and to the estuarine substrate, and may subject the biota to abnormally low salinities as the salt wedge is driven seaward. Extreme periods of drought may also have an adverse impact on the estuary. The operation of hydraulic structures -- dams and diversions -- can significantly alter the characteristics and the uses of an estuary. Clearly, these characteristics must be taken into account in determining the attainable uses of the water body.

CHAPTER III

CHARACTERISTICS OF PLANT AND ANIMAL COMMUNITIES

INTRODUCTION

Salinity, light penetration and substrate composition are the most critical factors to the distribution and survival of plant and animal communities in an estuary. This Chapter begins with an overview of the physical phenomena and biological adaptations which influence the colonization of the estuary. Following this, specific information is presented on Estuarine Plankton (phytoplankton and zooplankton), Estuarine Benthos (infaunal forms, crustaceans and molluscs), Submerged Aquatic Vegetation, and Estuarine Fish. There is also a short discussion of measures of biological health and diversity. This last subject is presented in much greater detail in the Technical Support Manual (U.S. EPA, November 1983).

The information in this Chapter (and its associated Appendices) has been compiled to provide an overview of the types of habitat, ranges of salinity, and life cycle and other requirements of plants and animals one might expect to find in an estuary, as well as analyses that might be performed to characterize the biota of the system.

With this information having been presented as a base, discussion in Chapter IV will be directed towards how the biological, chemical and physical data descriptive of the estuary may be synthesized into an assessment of the present and potential uses of the estuary.

COLONIZATION AND PHYSIOLOGICAL ADAPTATIONS

The estuarine environment is characterized by variations in circulation, salinity, temperature and dissolved oxygen supply. Due to differences in density, the water is generally fresher near the surface and more saline toward the bottom. Colonizing plants and animals must be able to withstand the fluctuating conditions in estuaries. Rooted plants need a stable substrate to colonize an area. Once established, the roots of aquatic vegetation help to stabilize the sediment surface, and the stems interfere with and reduce local currents so that more material may be deposited. Thus, small hummocks become larger beds as the plants extend their range.

The depth to which attached plants may become established is limited by turbidity, since they require light for photosynthesis. Estuaries are typically turbid because of large quantities of detritus and silt contributed by surrounding marshes and rivers. Algal growths may also hinder the penetration of light. If too much light is withheld from the lower depths, animals cannot rely heavily on visual cues for habitat selection, feeding, or in finding a mate.

Estuarine animals are recruited from three major sources: the sea, freshwater environments, and the land. Animals of the marine component have been most successful in colonizing estuarine systems, although the extent to which they penetrate the environment varies (Green 1968). Estuarine animals that belong to groups prevalent in freshwater habitats are presumed to have originated there. Such species comprise the freshwater component. The invasion of estuaries from the land has been accomplished mainly by arthropods.

When animals encounter stressful conditions in an estuary, they have two alternatives: they can migrate to an area where more suitable conditions exist, or if sedentary or sessile they can respond by sealing themselves inside a shell, or by retreating into a burrow.

Most stenohaline marine animals can survive in salinities as low as 10-12 ppt by allowing the internal environment (blood, cells, etc.) to become osmotically similar to the surrounding water (McLusky 1981). Such "conformers" often change their body volume. In contrast oligohaline animals actively regulate their internal salt concentration. They do so by active transport of sodium and potassium ions (Na^+, K^+). Osmoregulation relies on several possible physiological adaptations. Reduced surface permeability helps minimize osmotic flow of water and salts. In addition, the animal's excretory organs serve to conserve ions or water needed for osmoregulation.

Upper and lower tolerance limits define a range between which environmental factors are suitable for life (zone of compatibility). The adaptations of these tolerance limits are referred to as resistance adaptations. In estuaries, the major environmental factors to which organisms must adjust are periodic submersion and desiccation as well as fluctuating salinity, temperature, and dissolved oxygen.

Vernberg (1983) notes several generalizations concerning the responses of estuarine organisms to salinity: (1) those organisms living in estuaries subjected to wide salinity fluctuations can withstand a wider range of salinities than species that occur in high salinity estuaries; (2) intertidal zone animals tend to tolerate wider ranges of salinities than do subtidal and open-ocean organisms; (3) low intertidal species are less tolerant of low salinities than are high intertidal ones; and (4) more sessile animals are likely to be more tolerant of fluctuating salinities than those organisms which are highly mobile and capable of migrating during times of salinity stress. These generalizations reflect the correlation of an organism's habitat to its tolerance. Some estuarine animals are able to survive in adverse salinities, provided that the stress is fluctuating, not constant. For example, initial mortalities of the oyster drill (Urosalpinx cinerea) were very high when exposed to constant low salinity values. However, little or no mortalities occurred during ten days of exposure to low fluctuating salinities. Tolerance limits may also differ between larval and adult stages, as in the case of fiddler crabs (Uca pugilator). Adults are able to survive extended periods of 5 ppt salinity, while larvae cannot tolerate salinities below 20 ppt (Vernberg 1983). The salinity in which they were spawned may also influence larval responses.

Temperature also has an effect on salinity tolerances of organisms. Generally, cold-water species can tolerate low salinities best at low temperatures and tropical species can withstand low salinities best at high temperatures. The previous thermal history of an organism influences its resistance to temperature extremes. Acclimation to higher salinities can also broaden an organism's zone of compatibility for temperature.

The transport of oxygenated surface water to the bottom is greatly inhibited when an estuary is stratified. In addition, the solubility of oxygen in water is suppressed by salinity, so that estuarine DO levels at a given temperature may not be as high as would be seen in freshwater. As a consequence, many estuaries exhibit consistently low DO levels in the lower part of the water column, and may become anoxic at the bottom. This condition may be exacerbated by benthic DO demand. Many estuarine organisms must be tolerant of low DO. Those that are able will leave to seek areas of sufficient dissolved oxygen, while others (such as bivalves) will respond by regulating metabolic activity to levels that can be supported by the ambient DO concentration.

Intertidal organisms experience alternating periods of desiccation and submersion. These animals, mainly molluscs, are able to resist desiccation because of morphological characteristics that aid in controlling water losses. Others burrow into the moist substrate to avoid prolonged exposure to the air. Small animals with high ratios of surface area to volume are less resistant to water loss than are larger organisms.

MEASURES OF BIOLOGICAL HEALTH AND DIVERSITY

Estuaries are characterized by high productivity but low species diversity. Several authors have noted decreased species diversity in estuaries when compared to freshwater or marine systems (Green 1968, McLusky 1971, McLusky 1981, Haedrich 1983). Two major hypotheses explain the paucity of estuarine species. The first explanation is that of physiological stress caused by variable conditions in estuaries (McLusky 1981). Plants and animals must be able to withstand considerable changes in salinity, DO and temperature. In addition, because of tidal variation, they may be subjected to periods of dessication. Variable salinities are especially challenging to an organism's ability to osmoregulate. Because conditions in estuaries are not stable, fewer species inhabit estuaries than inhabit fresh or marine waters.

The second hypothesis explains decreased species diversity by the relative youth of present-day estuaries (McLusky 1971, McLusky 1981, Haedrich 1983). The estuaries that we see today probably did not exist several thousand years ago. Since this is a short period relative to the same scale over which speciation has taken place, few species have been able to adapt to and colonize the estuarine system. An investigation by Allen and Horn (1975) of several small estuarine systems in the United States revealed that a small number of species (<5) comprised more than 75 percent of the total number of individuals. Similarly, Haedrich (1983) noted that the number of fish families characteristic of estuaries comprises only six percent of the total number of families described.

Investigations of diversity in estuarine systems have employed the same diversity indices that are commonly used in freshwater systems (see U.S. EPA, 1983b, Chapter IV-2). The Shannon-Wiener index is often employed in conjunction with the two components that influence its value, a species richness index and a measure of evenness (McErlean 1973, Allen and Horn 1975, Hoff and Ibara 1977).

Because seasonal changes are so marked in estuaries, the selected diversity index should be sensitive to changes in species composition. Thus, quantitative similarity coefficients and cluster analyses may be used to determine the extent of similarity between samples. Such measures are discussed in Chapter IV-2 of the Technical Support Manual: Waterbody Surveys and Assessments for Conducting Use Attainability Analyses (U.S. EPA, 1983b).

An equal effort should be expended at each sampling station each time sampling is done. The results of a fish fauna survey may be biased by the sampling method employed. For example, the gear used (trawl, gill net, trap net, seine), the mesh size and the area in which fishing occurs determine the sizes, numbers and kinds of fish caught (McHugh 1967, McErlean 1973). Sampling gear and technique are also important in benthic and planktonic investigations. Because of the many migratory organisms found intermittently in estuaries, sampling should occur during each season of the year.

A major concern in estuarine systems is biological change due to pollution, especially alterations to commercially important populations. The ratio of annelids to mollusks and annelids to crustaceans has been used as an indication of environmental stress. By comparing these ratios to the Contamination Index (C_I) and the Toxicity Index (T_I), described in Appendix A, areas highly contaminated by metals and organic chemicals can be characterized (U.S. EPA, 1983a).

Briefly, contaminant factors (C_f) indicate the anthropogenic concentration of individual contaminants, based on metal content and Si/Al ratios in sediment. The Contamination Index (C_I) is a sum of these contaminant factors, giving equal weight to all metals, and thus has no ecological significance until combined with biotoxicity data. The map of the Chesapeake Bay in Figure III-1 illustrates the degree of metal contamination based on C_I. The Toxicity Index (T_I) is calculated using contaminant factors and EPA "acute" criteria for the metals, i.e., the concentration that may not be exceeded in a given environment at any time. This index gives information pertinent to the toxicity of sediments to aquatic life. Figure III-2 illustrates the results of calculations of Toxicity Indices for the Chesapeake Bay.

The Toxicity Index ranges from values of 1 to 20 where to lowest values denote the least polluted conditions. Characteristics associated with various values of T_I may also be seen in Chapter IV, Table IV-3. The Contamination Index is based on the calculation of the quantity C_f (see Appendix A) where C_f=0 when observed and predicted metal concentrations in sediment are the same, C_f<0 when the observed is less than the predicted, and C_f>0 when the observed is greater than the predicted.

The juvenile index is often used to help predict future landings of certain commercially important fish in estuaries. The juvenile index is simply the number of first year fish of a species divided by the number of seine

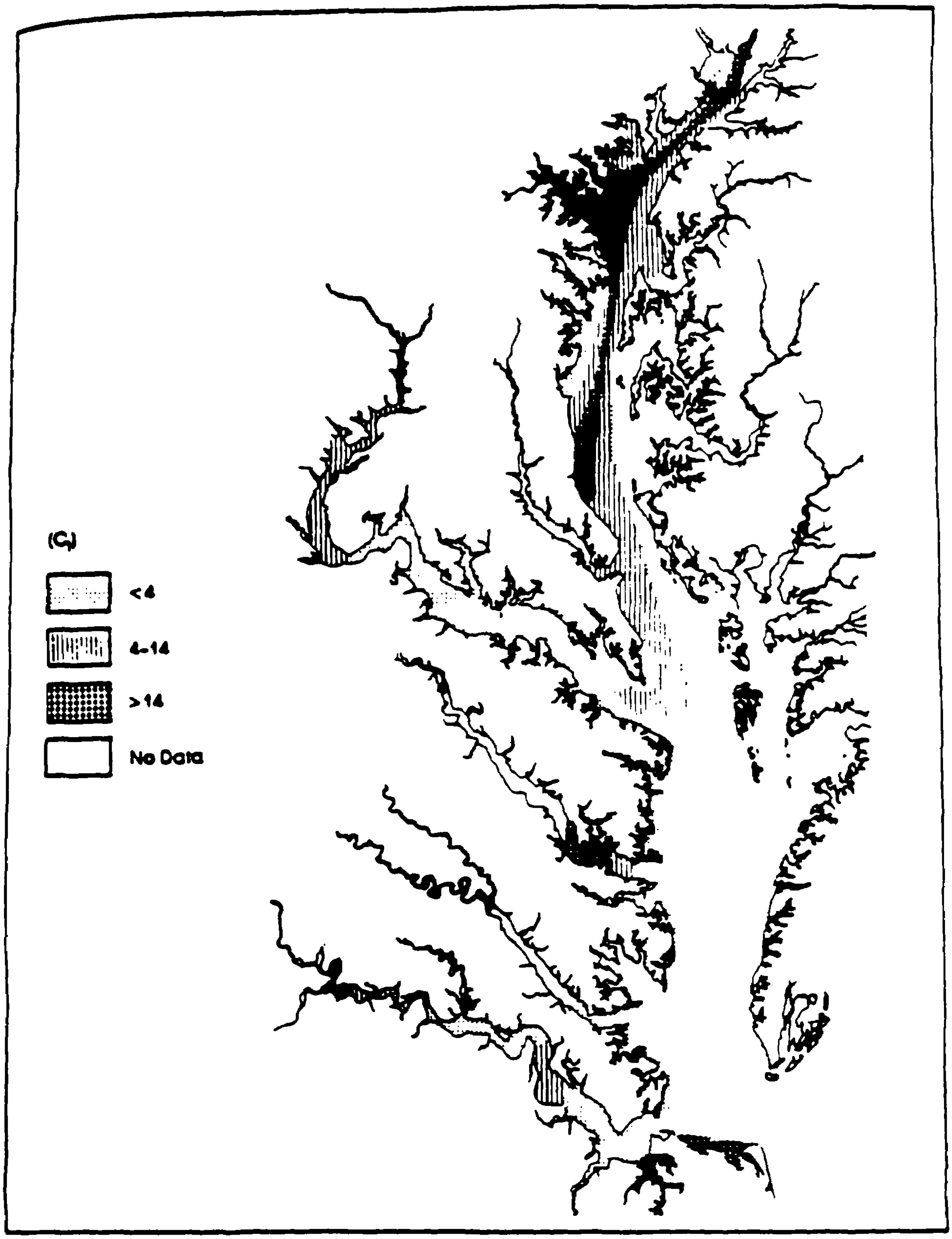


Figure III-1. Degrees of metal contamination in the Chesapeake Bay based on the Contamination Index (C_I). (from USEPA 1983c)

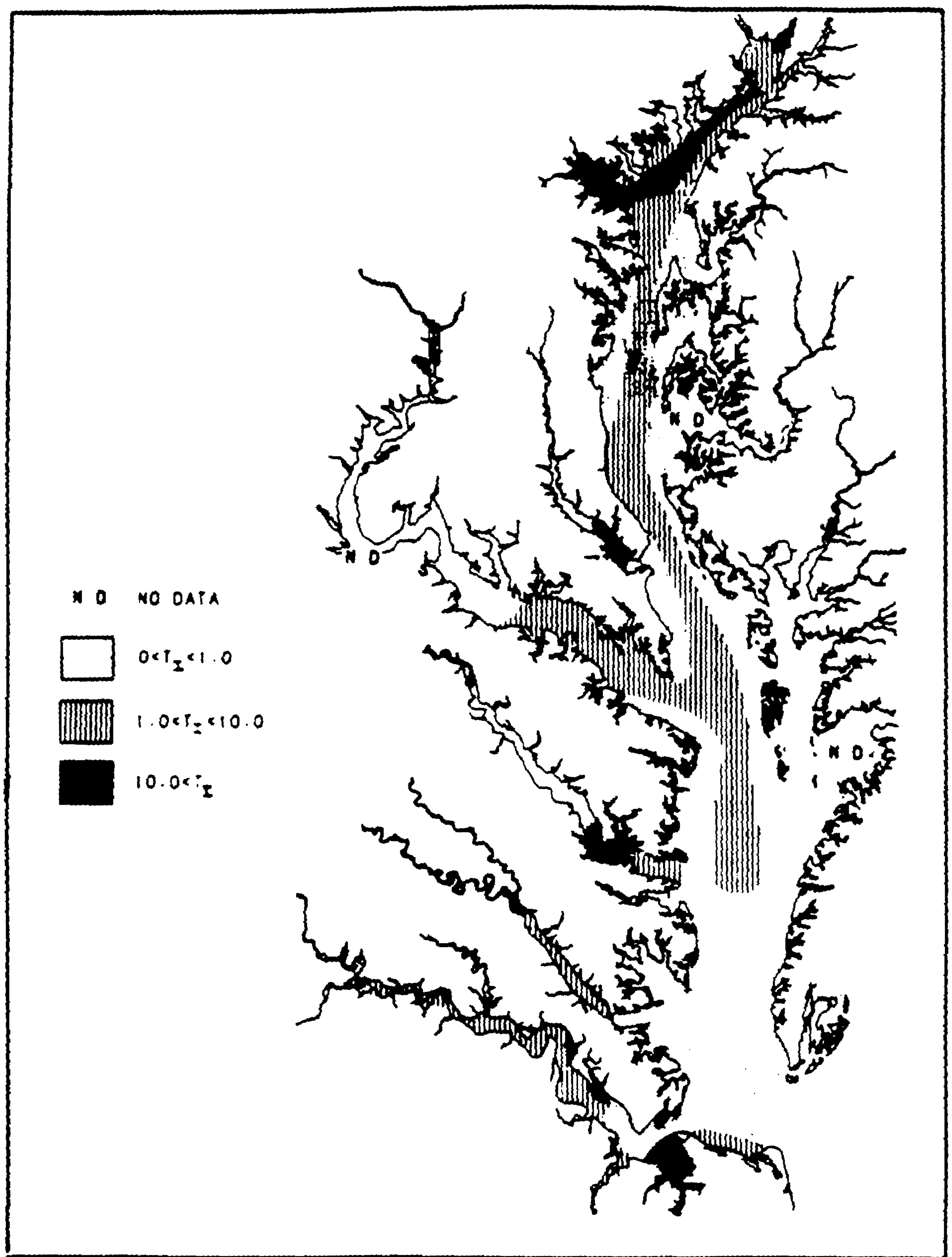


Figure III-2. Toxicity Index of surface sediments in Chesapeake Bay. (from USEPA 1983c) hauls. This index is then compared to juvenile indices from previous years along with commercial fisheries landings data.

In summary, species diversity in estuaries is generally lower than in adjacent freshwater or marine ecosystems. Either the changing environment or the youth of estuaries or perhaps a combination of both is responsible for this lack of species diversity. Indices of diversity that are used in estuaries are the same as those employed in freshwater studies and have been summarized in a previous document (U.S. EPA, 1983b).

ESTUARINE PLANKTON

Plankton include weak swimmers and drifting life forms. Most planktonic organisms are small in size, and although they may be capable of localized movement, their distribution is essentially governed by water movements. Because of their unique salinity conditions and currents, individual estuaries have characteristic plankton populations.

Phytoplankton

Three principal groups are included in the phytoplankton. They are diatoms, dinoflagellates and nanoplankton. Like the phytoplankton of freshwaters and oceans, estuarine phytoplankton require nutrients (such as phosphorus, nitrogen, silicon), vitamins, iron, zinc and other trace metals for growth. For photosynthesis to occur, adequate light must be available. Suitable salinities must also be present for phytoplankton populations to survive.

Nutrients generally are abundant in estuaries. Seasonal fluctuations in nitrogen and phosphorus levels are often evident, and are related to overland runoff and fertilizer application to agricultural lands. External sources are not entirely responsible for nutrient levels in estuaries. Cycling within estuaries also plays a role in plankton productivity. Thus the turnover, or replenishment time (R), of nutrients is significant in determining their availability. Replenishment time is defined as $R = [S]/Sp$, where [S] is the concentration of the nutrient in the phytoplankton and Sp is the daily production rate measured in terms of particulate content of that nutrient in the phytoplankton (Smayda 1983). Recycling mechanisms may be separated into (1) excretion of remineralized nutrients accompanying grazing by herbivorous zooplankton or benthic organisms, (2) release through sediment roiling and diffusive flux of nutrients from the interstitial water of sediments following microbial remineralization, and (3) kinetic, steady-state exchanges between nutrients present in the particulate phase (phytoplankton, bacteria, sedimentary particles) and in the dissolved phase. The importance of each of the preceding mechanisms is dependent upon characteristics, viz. depth and vertical mixing, of specific estuaries.

Although the phytoplankton of estuaries is an integral part of the ecosystem, its role is somewhat less important than in marine or freshwater lake ecosystems. This is due partly to the large quantities of detritus and bacteria that serve as an alternative food source for many primary consumers. Estimates of primary production are generally calculated from the utilization of nutrients (phosphates, C^{14} uptake, chlorophyll concentration) (Perkins 1974). The phytoplankton contribution to primary productivity is often minimal in many coastal plain estuaries. Although nutrients are abundant there, other factors limit phytoplankton production. At the compensation depth, the amount of oxygen produced by photosynthesis is equal to the amount utilized in respiration. Because of high turbidity, the compensation depth in estuaries is relatively shallow thus limiting the volume of water in which positive production occurs. Several authors maintain the importance of phytoplankton in supporting estuarine food webs, although the degree of contribution is controversial. Boynton, et al. (1982) provides a review of factors affecting phytoplankton production by comparing numerous estuarine systems.

The flushing time of an estuary also affects the phytoplankton population. Many estuaries have a relatively long flushing time and stable populations are able to develop. The Columbia River estuary has a stable system with a gradation from freshwater to brackish to marine plankton. In contrast, the Margaree River (the Gulf of St. Lawrence) is drained completely at low water and has no such gradation. Thus, high tide populations are typically marine, while a freshwater population is evident at low tide.

The species composition of an estuary may be unique. Narragansett Bay for example, is a shallow, well-mixed estuary located on the northeastern coast of the United States. Surface salinity ranges from 20.5 ppt near river mouths to 32.5 ppt at the mouth of the bay. Flushing time of the bay is estimated at thirty days (Smayda 1983). Because of tidal and wind-induced mixing, most of Narragansett Bay has neither a well-defined halocline or thermocline. Seasonal variation of plankton is evident, although the diatom *Skeletonema costatum* represents about 80% of total numerical abundance over the annual cycle (Smayda 1983). The major phytoplankton bloom occurs during December, coinciding with the minimum incident radiation and length of day. Blooms are regulated by temperature, light, nutrients, grazing, hydrographic disturbances and possibly species interactions. Neither blue-green algae nor dinoflagellates are important in Narragansett Bay due to its relatively high salinity. Planktonic blue-green algae tend to be more important in reduced salinities. Dinoflagellates (viz. *Prorocentrum triangulatum*, *Peridinium trochoideum*, *Massartia rotundata*, *Olisthodiscus luteus*) occur sporadically during the summer months, although diatoms continue to predominate. A succession of diatom species occurs seasonally, although *Skeletonema* is prevalent during all months. *Detonula confervacea* and *Thalassiosira nordenskioeldii*, important secondary species during the winter-spring bloom, are replaced by *Leptocylindrus danicus*, *L. minimus*, *Cerataulina pelagica*, *Asterionella japonica*, and *Rhizosolenia fragilissima*.

Phytoplankton in the Navesink River, New Jersey, were studied by Kawamura (1966). Based on salinity, several zones with characteristic phytoplankton were defined. Euglenoids dominated below 20 ppt. The zone in which salinity lay between 20 and 22 ppt was populated by *Rhizosolenia*. *Cerataulina bergonii* dominated in salinities ranging from 22 to 25 ppt. Dinoflagellates, including *Peridinium conicoides*, *P. trochoides*, and *Glenodinium danicum*, were prevalent in the outer region of the estuary. Open water beyond the mouth of the estuary was populated mostly by *Skeletonema costatum*. For regions with a fairly stable salinity gradient, Kawamura (1966) noted the dominant forms as presented in Table III-1.

TABLE III-1. DOMINANT PHYTOPLANKTON IN DEFINED SALINITY REGIONS

Salinity	Dominant Forms
2-5 ppt	*Anabaenopsis* sp., *Microcystis* sp., *Synedra ulna*, *Melosira varians*.
9-10 ppt	*Anabaena flos-aquae*, *Melosira varians*, *Chaetoceros* sp., *Biddulphia* spp., *Coscinodiscus* sp.
16 ppt	Euglenoids
20 ppt	*Melosira varians*, *Chaetoceros debilis*, *Ditylum brightwelli*, Peridinians.
24-31 ppt	*Skeletonema costatum*, *Rhizosolenia longiseta*, *Biddulphia aurita*, *Ditylum brightwelli*, Dinophyceans.

from Kawamura (1966).

Zooplankton

Zooplankton commonly found in estuarine reaches have been divided into the following groups based upon thei- origins and salinity tolerances: (1) Marine Coastal species, (2) Estuarine, and (3) Freshwater. One of the dominant copepods in estuaries is *Acartia tonsa*. Although it is not utilized directly by humans, *A. tonsa* is a major food source for fish or invertebrates that are consumed by humans (Jones and Stokes Assoc. 1981). Several surveys of the zooplankton in Narragansett Bay have been conducted and are summarized in Miller (1983). Copepods were the dominant group, comprising 80% or more of the individuals on an annual average. Important species were *Acartia clausi*, *A. tonsa*, *Pseudocalanus minutus* and *Oithona* spp. Rotifers were abundant in late winter, and cladocerans were abundant in early summer. Flushing reaches a peak in March-April, coinciding with a low in biomass.

Zooplankton have also been studied extensively in the Chesapeake and Delaware Bays, resulting in the following list of predominant species:

(1) Coastal:

copepods - *Centropages typicus*, *C. hamatus*, *Labidocera aestiva*, *Temora longicornis*, *Paracalanus parvus*, *Pseudocalanus minutus*;

cladocerans - *Penilia avirostris*, *Evadne nordmanni*.

(2) Estuarine:

copepods - *Acartia tonsa*, *Acartia clausi*, *Eurytemora affinis*, *Scottolana canadensis* (harpacticoid), and *Pseudodiaptomus coronatus*;

cladocerans - Podon polyphemoides.

(3) Freshwater:

copepods - Cyclops viridis;

cladocerans - Bosmina longirostris.

Grazing by zooplankton is an important factor in the control of phytoplankton populations, although the precise role played is not yet well-defined. The population dynamics of zooplankton on the east coast, including seasonal cycles and predation by ctenophores, is covered extensively by Miller (1983). Ctenophores have not been observed in Yaquina Bay, Oregon, and it is probable that fish predators limit zooplankton densities.

Comparatively less information is available on Gulf coast zooplankton distributions than for the Atlantic coast. Some references for zooplankton community structure and distributions in Louisiana estuaries and coastal waters are: Brice, 1983; Binford, 1975; Cuzon du Rest, 1963; Drummond, 1976; Gillespie, 1971.

Planktonic larval forms of organisms such as oysters and crabs are included in the temporary zooplankton. The veliger larvae of molluscs become part of the plankton during the spring and summer. Some estuarine worms also have planktonic larval forms. The occurrence of these forms is governed by the breeding season of the adults. Environmental tolerances of the larval forms of the blue crab (Callinectes sapidus) and the American oyster (Crassostrea virginica) are found in Appendix B (e,f).

To persist in an estuary, zooplankton, like phytoplankton, must have rates of population increase at least equal to the rates of loss due to tidal flushing and river flow. High flushing rates generally prohibit the development of an endemic plankton population, and the plankton found merely resemble those found in the ocean offshore. Studies of population budgets have been made on a few estuaries (Narragansett Bay, Great Pond, Moriches Bay) and are mentioned briefly by Miller (1983).

The following articles contain information on methods in zooplankton research: Computer and electronic processing of zooplankton (Jeffries 1980); Gear used (Schindler 1969, Josai 1970); Sampling for biomass-standing stock (Ahlstrom et al. 1969, Colebrook 1983, Tranter 1968); Fixation and preservation of zooplankton (Steedman 1976); Icthyoplankton (Smith and Richardson 1977).

ESTUARINE BENTHOS

Those organisms which live on or in the bottom of any water body are the benthos. Plants such as diatoms, macroalgae and seagrasses comprise the phytobenthos, while the zoobenthos includes the animals occupying this habitat. The estuarine zoobenthos will be discussed in this section. The zoobenthos is generally divided into macro-, meio- and microbenthos. Meiobenthos pass through a 1- or 2-mm sieve, but are larger than 100 um;

macro- and microbenthos are respectively larger and smaller than meiobenthos (Wolff 1983).

Although the diversity of the benthos in estuaries is low compared to other ecosystems, benthic production is relatively high. A high level of food (detritus and plankton) and shallow depths contribute to the characteristically high benthic production noted in estuaries. Detritus is readily available to the benthos because it sinks through the shallow water. In addition, waves and tidal currents promote resuspension of particles, making them available to filter-feeders. The predominance of relatively opportunistic species, with one or more generations per year, results in a high turnover of biomass and thus high production. Macrofauna have high biomass and low turnover times and hence have economic and commercial value. Meiofauna, with low biomass and high turnover rate, play an essential role as nutrient regenerators and food for higher trophic levels (Tenore et al. 1977, McIntyre and Murison 1973, Ajheit and Scheibel 1982).

Infaunal Forms

The benthos comprises invertebrates such as thread worms, bristle worms, ostracods, and copepods as well as commercially important species of crustaceans and molluscs. Nematodes (Nematoda, thread worms) dominate the shallow water meiofauna of estuarine sediments. In addition to nematodes, permanent meiofauna include copepods, gastrotrichs, oligochaetes, rotifers and turbellarians. Juvenile macrofauna comprise the temporary meiofauna. Generally, coarser sediments support a greater diversity of species than finer estuarine sediments (Ferris and Ferris 1979). Polychaetes (Polychaeta:Annelida, bristle worms) are abundant in the soft bottom, especially within the sediment of intertidal mud flats.

Studies have used polychaete populations to characterize water bodies as having healthy, polluted, or very polluted bottoms. The use of benthic organisms as indicator species is well-documented for freshwater studies whereas studies in the estuarine/marine environment are relatively few (Reish 1979). Although the species composition in freshwater is different than marine species composition, the concept of using benthic communities as indicators of pollution remains the same. In estuarine systems, polychaete species composition changes from zones characterized as healthy to those classified as polluted. As shown in Table III-2, there is a concurrent decrease in dissolved oxygen concentration, an increase in the organic carbon content of the soil, and a reduction in the number of organisms until all species are absent (Reish 1979). However, the validity of using polychaetes as indicator species has been questioned, since polychaetes such as Capitella capitata, an opportunistic organism whose presence has often been cited as an indication of pollution, also occur in pristine estuarine areas (Reish 1979). The following literature contributions also pertain to the use of benthos as indicators of pollution: Sediment bacteria as indicators (Erkenbrecher 1980); Meiofauna as indicators (Coull et al 1981, Raffaelli 1981, Warwick 1981); Macrofauna as indicators (Gray and Mirza 1979).

TABLE III-2. SUMMARY OF BIOLOGICAL, CHEMICAL AND PHYSICAL CHARACTERISTICS OF FIVE ECOLOGICAL AREAS OF THE LOS ANGELES-LONG BEACH HARBORS[a,b].

Characteristic	Healthy bottom, *Tharyx parvus* *Cossura candida* *Nereis procera*	Semihealthy bottom I, *Polydora paucibranchiata* *Dorvillea articulata*	Semihealthy bottom II, *Cirriformia luxuriosa*	Polluted bottom, *Capitella capitata*	Very polluted bottom, no animals
Number of animal species (average)					
Polychaetes	7	5	5	3	0
Nonpolychaetes	3	2	2	2	0
Dissolved oxygen (ppm) (median)					
Surface	6.0	2.5	2.5	3.5	1.6
20 ft depth	6.0	1.2	1.2	1.5	2.2
pH (median)					
Surface	7.8	7.3	7.4	7.6	7.5
20 ft depth	7.8	7.4	7.6	7.6	7.5
Substrate	7.2	7.2	7.2	7.3	7.1
Nature of substrate (in order of importance)	Gray mud, black mud, black sulfide mud	Black sulfide mud, gray clay, sand, and mud, black mud	Black sulfide mud, gray clay, black mud	Black sulfide mud	Black sulfide mud
Organic carbon of substrate (%) (median)	2.5	2.0	2.7	2.7	1.4

[a]Data from Reish (1959)
[b]Dominant species of polychaete

(from Reish 1979)

Crustaceans

Crustaceans include microorganisms such as ostracods, copepods and isopods along with commercially important macroorganisms such as crabs, shrimp and lobsters. The crabs (Arthropoda:Crustacea:Decapoda:Brachyura) that have successfully colonized North American estuarine systems are listed in Table III-3. Brachyuran crabs have a complex ontogeny. They are released from the female as zoeae, or free swimming larvae, into meso- to euhaline waters. The zoeae undergo a series of molts before reaching the megalopa stage. The megalopa metamorphoses into the first crab stage, which becomes the adult following successive molts (Williams and Duke 1983). It has been noted that above and below the preferred temperature range, the length of time required for larval development increases. Two species of Cancer that have commerical value, C. magister (Pacific Dungeness crab) and C. irroratus (Rock crab), normally enter estuaries only in high salinity regions. Larvae of C. magister and C. irroratus prefer conditions of 25-30 ppt, 10-13°C and 23.3-32.3 ppt, 13°-21°C, respectively.

Callinectes sapidus, the blue crab, supports a major fishery in the United States. The species lives in fresh water to salinities as high as 117 ppt (large males have been recorded in salt springs over 180 miles from the sea in Marion County, Florida) and from the water's edge to 35 meter depths. Appendix B (Table 1e) contains information pertaining to the life cycle of the blue crab. Additional information on general life histories of crabs and other commercially important shellfish in Gulf Coast waters is compiled by Benson (1982). The family Portunidae is also represented by Carcinus maenas in estuaries. The green or shore crab normally inhabits waters ranging in salinity from 10-33 ppt, and depths of less than 5-6 m (Williams and Duke 1979). Other crabs commonly found in North American estuaries are listed in Table III-3. Among the xanthid crabs, only Menippe mercenaria, the stone crab, has any fishery value. The major commercial fishery for stone crabs occurs in Florida, where its flesh is considered a delicacy.

Most of the information about shrimp pertains to the commercially valuable penaeid shrimp, Penaeus duorarum (pink shrimp), Penaeus aztecus (brown shrimp) and Penaeus setiferus (white shrimp). Penaeid shrimp are dependent upon estuaries during their transformation from the postlarval stage to the juvenile stage. Adults migrate from the estuarine environment to coastal and nearshore oceanic waters (Couch 1979). The life cycle of the penaeid shrimp is illustrated in Figure III-3. The range of the brown shrimp extends from Martha's Vineyard, Massachusetts, through the Gulf of Mexico to the Yucatan Peninsula, Mexico (Turner, 1983). Brown shrimp spawn in offshore marine waters deeper than 18 m (59 ft). Movement of postlarvae into estuaries has been observed from January through June in Louisiana. A peak migration from March to April was noted for Galveston Bay, Texas. Postlarval brown shrimp prefer salinities of 10 to 20 ppt, and temperatures above 15°C. Transformation from postlarvae to juveniles occurs four to six weeks after entering the estuary. Juveniles remain in shallow estuarine areas (near the marsh-water or mangrove-water interface or in seagrass beds) that provide feeding habitat and protection from predators until they reach 60 to 70 mm (2.4 to 2.8 inches) total length (TL). They move into deeper, open water, and begin gulfward migration when they reach 90 to 110 mm (3.5 to 4.3 inches) (Turner and Brody, 1983).

TABLE III-3. TAXONOMIC POSITION AND HABITAT OF DECAPOD CRUSTACEAN SPECIES, INFRAORDER BRACHYURA, OF CONCERN IN ESTUARINE POLLUTION STUDIES.

Taxon	Habitat
Infraorder Brachyura	
Section Cancridea	
Family Cancridae	
Cancer irroratus Say. Rock crab	Temperate-polyhaline
Cancer magister Dana. Dungeness crab	
Section Brachyrhyncha	
Superfamily Portunoidea	
Family Portunidae. "Swimming" crabs	
Subfamily Portuninae	
Callinectes sapidus Rathburn. Blue crab	Temperate-tropical-euryhaline
Carcinus maenas (Linnaeus). Green or shore crab	Temperate-polyhaline
Superfamily Xanthoidea	
Family Xanthidae	
Subfamily Xanthinae. "Mud" crabs	
Cataleptodius (=*Leptodius*) *floridanus* (Gibbes)	Tropical-polyhaline
Eurypanopeus depressus (S. I. Smith)	Temperate-mesohaline
Neopanope sayi (S. I. Smith)*	Temperate-mesohaline
Panopeus herbstii A. Milne Edwards	Temperate-tropical-mesohaline
Rhithropanopeus harrisii (Gould)	Temperate-oligo-mesohaline
Subfamily Menippinae	
Menippe mercenaria (Say). Stone crab	Warm temperate-subtropical-mesopolyhaline
Family Grapsidae	
Subfamily Varuninae	
Hemigrapsus nudus (Dana). Purple shore crab*	Temperate-polyhaline
Subfamily Sesarminae	
Sesarma cinereum (Bosc). Wharf crab*	Temperate-tropical-polyhaline-semiterrestrial
Sesarma reticulatum (Say). "Marsh crab"*	Temperate-polyhaline-semiterrestrial
Superfamily Ocypodoidea	
Family Ocypodidae	
Subfamily Ocypodinae	
Uca minax (Le Conte). Red jointed fiddler	Temperate-oligo-mesohaline-semiterrestrial
Uca pugilator (Bosc). Sand fiddler	Temperate-subtropical-mesopolyhaline-semiterrestrial
Uca pugnax (Smith). Mud fiddler	Temperate-mesopolyhaline-semiterrestrial

*Species intimately associated with communities reported here and pollution studies published elsewhere

(from Williams and Duke 1979)

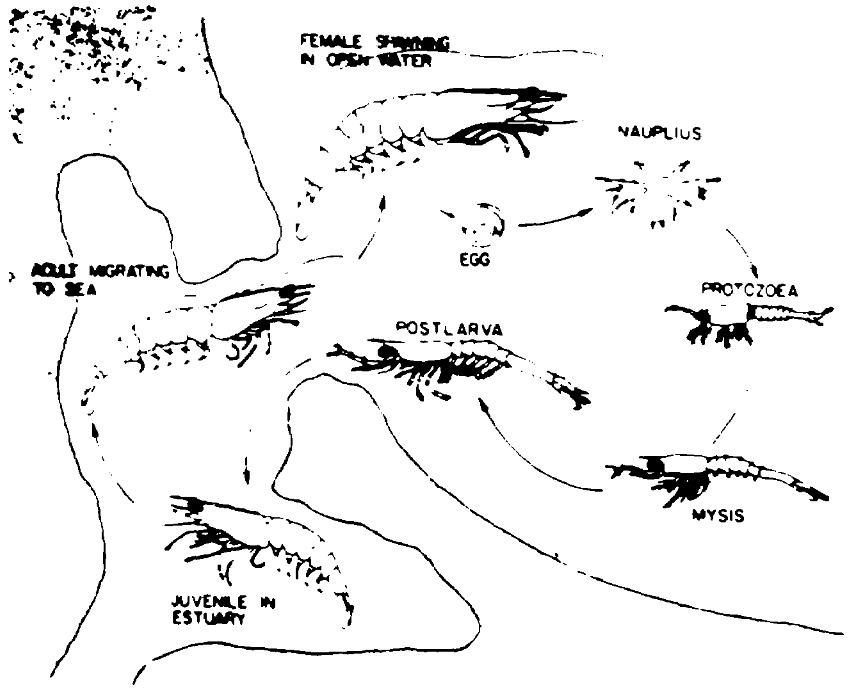


Figure III-3. Life Cycle of the Penaeid Shrimp. (from Couch 1979)

Postlarval white shrimp migrate into estuaries from late spring to early fall, and are most abundant in Louisiana estuaries from June through September. They are generally found in lower salinity waters than brown shrimp and prefer water temperatures higher than 15°C. White shrimp (120 to 140 mm) leave Gulf of Mexico embayments from September to December, as the water cools.

Finally, the grass shrimp (Paleomonetes sp.) of estuaries commonly live in patches of grasses growing in shallow water. Because of aquarium suitability, members of palaemonidae are often used in pollution studies.

Molluscs

The last major group in the estuarine benthos is the molluscs. The molluscs include clams, mussels, scallops, oysters and snails. Clams of major importance include Mya arenaria (soft shell clam), Mercenaria mercenaria (hard shell clam), and Rangia cuneata (brackish water clam).

The soft shell clam is common in bays and estuaries on both the east and west coasts of the United States, although it is commercially important only on the east coast. Soft shell clams can tolerate a wide range of salinities and temperatures. Larval development occurs at salinities from 16-32 ppt, and at temperatures of 17-23°C. Mya arenaria occurs in a variety of substrates, but prefers a mixture of sand and mud (Jones and Stokes Assoc. 1981). Hard clams (Mercenaria mercenaria) can tolerate high pollution and low oxygen levels; thus, they thrive where other species cannot compete. Hard clams prefer substrates of sand or sandy clay (Beccasio et al. 1980). The littleneck clam (Protothaca staminea) is a hardshell species found in estuaries, bays and open coastlines along the Pacific coast. It ranges from the Aleutian Islands to Socorro Island, Mexico. Minimum salinity for survival is 20.0 ppt (Rodnick and Li 1983). The brackish water clam is found in low salinity bays and estuaries from the Chesapeake Bay to Mexico (Haven 1978). Rangia cuneata can survive in fresh water, but needs brackish water for spawning (Menzel 1979).

The bay mussel (Mytilus edulis) is found worldwide in estuaries and bays. It is tolerant of variations in temperature, salinity and dissolved oxygen. Although the bay mussel is under stress at salinities less than 14-16 ppt, it can survive at 4 ppt for short periods of time. This mussel attaches to any hard substrate and may be found on rocks, stones, shingles, dead shells, ship bottoms, piers, harbor walls and compacted mud and sand (Jones and Stokes Assoc. 1981).

Bay scallops (Argopectin irradians) are usually found in shallow estuarine eelgrass beds, but may occur in depths to 18 m (Beccasio et al. 1980). They ingest detritus, bacteria and phytoplankton. The large amount of detritus consumed reflects its great availability in estuarine systems (McLusky 1981).

The American oyster (Crassostrea virginica) is a permanent resident of estuaries. It is a valuable component of east coast fisheries. Oysters prefer salinities between 14.1 ppt and 22.2 ppt, although they are able to tolerate a wider range, from 4-5 ppt to 35 ppt (Castagna and Chanley 1973). Within the range of distribution of C. virginica, the species lives in water temperatures from about 1°C (during the winter in northern states) to about 36°C (in Texas, Florida, and Louisiana) (Galtsoff 1964). Larvae develop well in depths from 2 to 8 meters at temperatures of 17.5 to 32.2°C. The oyster population in high salinities is limited by oyster drills (e.g. gastropod Urosalpinx cinerea) and parasites (MSX and Dermocystidium) (Haven 1978). Spawning by oysters is dependent upon temperature, and commences when the water reaches from 16-28°C depending upon geographic area (Bardach et al. 1972, Ingle 1951). After 6-14 days, the eggs hatch and the free-swimming larvae settle on a suitable hard substrate. Oysters filter food from the water column and deposit organic material (feces and pseudofeces) which is then available to other benthic organisms; thus, they play a valuable role in increasing the productivity of the area in which they live (McLusky 1981).

Temperature tolerances of American oysters differ with latitude. Oysters at latitudes north of Cape Hatteras can survive at temperatures less than 0°C for 4 to 6 weeks, while Gulf of Mexico oysters die if subjected to such low temperatures (Cake 1983). Temperatures required for mass spawning also differ with latitude. Apalachicola Bay reached temperatures of 26-28°C before mass spawning occurred, while a low of 16.4°C induced mass spawning in Long Island Sound, New York (Ingle 1951). Other oyster species commonly found in estuaries of the United States are Crassostrea gigas (Pacific oyster) and Ostrea edulis (flat oyster).

Snails (Gastropoda) have not been studied as extensively as the molluscs discussed above. In general, adult snails are slow moving, benthic, and able to endure a variety of temperatures and salinities. After the eggs are hatched, most snails have a planktonic stage; a few emerge as crawling juveniles. Many snails are vegetarians and scrape algae from surfaces. Some carnivorous snails use their radulas to drill holes in other shelled animals (e.g., oyster drills). Other snails consume gastropods whole, digesting the tissue and regurgitating the empty shells (Menzel 1979). More information about the distributions and habitats of NE Gulf gastropods is described in Heard (1982).

References on methodology for the study of estuarine microbiota and benthos include: Holme and McIntyre 1971, Hulings and Gray 1971, U.S. EPA 1978, Uhlig et al. 1973, de Jonge and Bouman 1977, Federle and White 1982, White et al. 1979, Montagna 1982.

In conclusion, the estuarine benthos play an important role in estuarine ecosystems. The nematodes and polychaetes, along with the commercially important shellfishes, contribute to the high productivity noted in most estuaries. The benthos are generally able to tolerate variations in temperature and salinity. Thus, they are able to live, and often thrive, in estuaries.

SUBMERGED AQUATIC VEGETATION

Submerged aquatic vegetation (SAV) plays an important role in the estuarine ecosystem, providing habitat, substrate stability and nourishment. These functions are the subject of discussion in this section. However, submerged aquatic vegetation also provides a valuable frame of reference against which to assess the health of an estuary, or portion of an estuary. The importance of SAV to an analysis of the uses of an estuarine waterbody will be discussed further in Chapter IV, Interpretation.

Role of SAV in the Estuary

Plants increase the stability of bottom sediments and reduce shoreline erosion. In addition, because the plants help to slow the tidal current, more materials may settle from suspension, augmenting the substrate and decreasing turbidity. Species differ in their ability to reduce turbidity. For example, areas dominated by Potamogeton perfoliatus (a highly branched species) were more instrumental in improving water clarity than areas where Potamogeton pectinatus (a thin-bladed single leaf species) dominated (Boynton et al. 1981).

Aquatic plants serve as both sources and sinks for nutrients. During the growing season, SAV absorbs nutrients from the water and sediments. Release of nutrients occurs when the vegetation dies. Submerged aquatic vegetation also provides valuable habitat for fish and crabs, along with molluscs and other epifauna. SAV provides shelter, spawning areas and shade for fish, while roots, stems and leaves provide firm bases for the attachment of mussels, barnacles, molluscs and other epifauna. Thus, vegetated bottoms exhibit a greater species richness than unvegetated bottoms (U.S. EPA 1982).

Stevenson and Confer (1978) cited a study (Baker 1918) which emphasized the large number of organisms associated with submerged aquatic vegetation. Over a 450 sq. mile area, Potamogeton sp. harbored 247,500 molluscs and 90,000 associated animals (total fauna, 337,500) and Myriophyllum sp. harbored 45,000 molluscs with 56,250 associated animals (total fauna, 101,250). Epiphytes and macroalgae constitute a significant and sometimes a dominant feature of SAV community production and biomass, as can be seen from Table III-4. Fish such as silversides (Menidia menidia), fourspine stickleback (Apeltes quadracus) and pipefish (Syngnathus fuscus) take advantage of this abundant epifauna for food.

Eelgrass beds also provide protection for amphipods from predatory finfish. Grass shrimp (Palaeomonetes pugio) seek protection from predatory killifish (Fundulus heteroclitus) in eelgrass beds. Young and molting crabs find shelter in areas of submerged aquatic vegetation as well.

Aquatic vegetation enters the food chain though grazing by waterfowl or as detritus passing through epifaunal and infaunal invertebrates to small and large fish. The extent to which SAV is used as a food source is determined mainly by two methods. The first is direct visual identification of material in an organism's digestive system. Such analyses are time-consuming, and the degree to which food items can be identified is often limited to larger items that are resistant to digestion. The second techique is based on $C^{12}:C^{13}$ ratios in plants and associated predators. This method assumes that animals feeding on a particular plant will, in time, reflect the food source ratio. Problems arise when animals feed on a variety of species, or if several plants have similar $C^{12}:C^{13}$ ratios. In addition, determination of $C^{12}:C^{13}$ ratios is a relatively expensive procedure.

Submerged aquatic vegetation also plays a role in nutrient cycling in estuaries. Since plants act as nutrient traps and sinks for dissolved minerals, SAV communities are capable of removing nutrients from the water column and incorporating them into biomass. Iron and calcium were found to be absorbed from the sediment by Myriophyllum spicatum. The release of nutrients and minerals occurs by excretion by living plants or by the death and decomposition of SAV.

Distribution of SAV

The distribution of SAV species is determined largely by salinity. The degree of flooding also affects vegetation distribution and is particularly important for Gulf Coast estuaries (Sasser 1977). In a study of the Chesapeake Bay, Steenis (1970, cited by Stevenson and Confer 1978) noted the following tolerance levels for Bay vegetation:

TABLE III-4. DATA FROM SELECTED SOURCES INDICATING THE PARTITIONING OF (a) PRODUCTION (Pa), $gCm^{-2}y^{-1}$ AND (b) BIOMASS gm^{-2} (ORGANIC) BETWEEN VARIOUS AUTOTROPHIC COMPONENTS OF SAV COMMUNITIES

a. Location	Species	Seagrass	Epiphytes	Benthic micro-algae	Macro-algae	Phytoplankton	Reference
Florida	Thalassia	1000	200	---	---	---	Jones 1968
Mass.	Zostera	---	20	---	---	---	Marshall 1970
Calif.	Ruppia	28	---------------267 ---------------			91	Wetzel 1964
N.Carolina	Zostera	330	73	---	---	---	Penhale 1977
Ches. Bay	Zostera[a]	0.48	0.17	-0.05	---	0.09	Murray (pers.comm.)
	P.pectinatus	0.5-2.2	---	---	---	0.3-1.0	Kaumeyer et al. 1981
	P.perfoliatus	1-3.0	---	---	---	0.5-1.0	Kaumeyer et al. 1981

a) Daily estimates in summer period.

b. Location	Species	Seagrass	Epiphytes	Benthic micro-algae	Macro-algae	Phytoplankton	Reference
Europe	Cymodocea	400-700	---	---	375	---	Gessner and Hammer 1960
Alaska	Zostera						
Kinzarof		1500	---	---	393		McRoy 1970
Klawak		415	---	---	29		
Others		113	---	---	2.4		
N.Carolina	Zostera	80	25	---	---	---	Penhale 1977
Ches. Bay	P.pectinatus	20-60	0.1-0.6	---	---	---	Staver et al. 1981
	P.perfoliatus	20-80	0.1-0.6	---	---	---	Staver et al. 1981

(from USEPA 1982)

3 ppt
Najas guadalupensis (southern naiad)

3-5 ppt
Chara spp. (muskgrass)
Vallisneria americana (wildcelery)

12-13 ppt
Elodea canadensis (elodea)
Myriophyllum spicatum (Eurasian watermilfoil)
Ceratophyllum demersum (coontail)

20-25 ppt
Potamogeton perfoliatus (redhead grass)
Potamogeton pectinatus (sago pondweed)
Zannichellia palustris (horned pondweed)

over 30 ppt
Ruppia maritima (widgeongrass)
Zostera marina (eelgrass)

The depth at which vegetation is able to survive is directly related to the penetration of incident radiation. Plants need light for photosynthesis, therefore turbidity affects their distribution by decreasing the amount of sunlight reaching greater depths. Temperature also affects the distribution of SAV, and exerts considerable influence upon its vegetative growth and flowering. These factors are considered in more detail in Appendix C for several east-coast species.

Three associations of submerged aquatic vegetation were described for the Chesapeake Bay, based on their co-occurrence in mixed beds. The first association tolerates fresh to slightly brackish water (upper reaches of the Bay) and includes bushy pondweed, coontail, elodea (waterweed), and wildcelery. The middle reaches of the Bay have associations of widgeongrass, Eurasian watermilfoil, sago pondweed, redhead grass, horned pondweed, and wildcelery. Finally, in the lower reaches of the Bay, eelgrass and widgeongrass predominate. The kinds of submerged aquatic vegetation encountered in the Chesapeake Bay from 1971 to 1981 are listed in Table III-5.

The major species of SAV found on the eastern coast of the United States (their distribution, environmental tolerances and consumer utilization) are listed in Appendix C. The species that are especially important as food items for waterfowl are coontail, muskgrass, bushy pondweed, sago pondweed, redhead grass, widgeongrass and wildcelery. Grazing by waterfowl is a primary force in the management of aquatic vegetation. Some aquatic vegetation, although it provides protective cover for wildlife, is considered a nuisance because of excessive growth and clogging of waterways. Elodea, Eurasian watermilfoil, and sago pondweed are among those considered to be pest species.

Information concerning aquatic vegetation in southern U.S. estuaries is found in literature by Chabreck and Condrey 1979, Beal 1977, and Correll and Correll 1972.

TABLE III-5. A LISTING OF THE SUBMERGED AQUATIC VEGETATION ENCOUNTERED IN THE CHESAPEAKE BAY FROM 1971 TO 1981.

Species	Vascular Plants[1]	Macro-Algae[1]
1. Redhead grass (*Potamogeton perfoliatus*)	X	
2. Widgeongrass (*Ruppia maritima*)	X	
3. Eurasian watermilfoil (*Myriophyllum spicatum*)	X	
4. Eelgrass (*Zostera marina*)	X	
5. Sago pondweed (*P. pectinatus*)	X	
6. Horned-pondweed (*Zanichellia palustris*)	X	
7. Wildcelery (*Vallisneria americana*)	X	
8. Common elodea (*Elodea canadensis*)	X	
9. Naiad (*Najas guadalupensis*)	X	
10. Muskgrass (*Chara spp.*)		X
11. Slender pondweed (*P. pusillus*)	X	
12. Coontail (*Ceratophyllum demersum*)	X	
13. Unidentified fragments	X	
14. Curly pondweed (*Potamogeton crispus*)	X	
15. Sea lettuce (*Ulva spp.*)		X
16. *Agardhiella spp.*		X
17. Unidentified filamentous green algae		X
18. Unidentified green algae		X
19. *Gracilaria spp.*	X	
20. Water-stargrass (*Heteranthera dubia*)	X	
21. Unidentified alga		X
22. *Enteromorpha spp.*		X
23. *Ceramium*		X
24. *Polysiphonia*		X
25. *Dasya spp.*		X
26. Unidentified red alga		X
27. Unidentified brown alga		X
28. *Champia parvula*		X

[1] An "X" in the column indicates the type of SAV.

(from USEPA 1982)

Adverse Impacts on SAV

Portions of the estuary may become enriched beyond their flushing and assimilative capacity and elevated levels of nitrogen and phosphorus begin to support abnormal algal growth and eutrophic conditions. Algal growths are important because they act to diminish to penetration of sunlight into the water. Submerged aquatic vegetation is dependent upon sunlight for photosynthesis, and when light penetration is diminished too much by algal growths, the SAV will be affected. These factors are discussed in detail in Chapter II.

Runoff may also introduce herbicides to the estuarine ecosystem. The magnitude of detrimental effects depends upon the particular herbicide, and its persistence in the environment and potential for leaching. Furthermore, several herbicides have a synergistic effect along with nutrients, its potential for leaching and persistence in the environment. Several pathogens may attack and diminish the size of submerged aquatic vegetation beds. Rhizoctonia solani is a fungus that attacks the majority of duck food plants, but is especially pathogenic to sago pondweed (Stevenson and Confer 1978). Lake Venice Disease causes a gradual wasting away of the host plant; it is manifested as a brownish, silt-like coating on leaves and stems. Milfoil is attacked by the Northeast Disease, which gradually causes the leaves to break off, leaving a blackened stem.

Survey Techniques

Aerial, surface and subsurface methods are used to prepare maps delineating vegetation types and percent cover. Plant growth stage (e.g. season) is critical when planning a plant survey. For example, early summer is the optimum time of year to record maximum plant coverage in the Chesapeake Bay but a different time of year may be more appropriate in other parts of the Country. Water transparency is also important to show plant growth. Aerial methods are useful in determining the distribution of plant associations, irregular features, normal seasonal changes and perturbations caused by pollutants. Mapping cameras are designed to photograph large areas without distortion. Areas of SAV beds may be derived from topographic quadrangles (Raschke 1983). The Earth Resources Observation System (EROS) Data Center may be used to obtain listings and photographs already available for a particular area.

Surface or ground maps can be prepared if the area is relatively small. Distances can be determined by ruled tapes, graduated lines, range finders, or, if more accuracy is required, surveyor's tools. Field observations of species may be supplemented by photographs. Divers can mark subsurface beds with bouys to facilitate determination of bed shapes and areas from the surface.

Regional surveys of flora give qualitative information, based upon visual observation and collection of plant types. To obtain more quantitative information, line transects, belt transects, or quadrats may be employed (Raschke 1983). Use of line transects involves placement of a weighted nylon or lead cord along a compass line and recording plant species and linear distance occupied. A belt transect can be treated as a series of quadrats, with each quadrat defined as the region photographed from a standard height or a marked area. The technique of sampling within a quadrat or plot of standard size is applicable to shallow and deep water. Where visibility is poor, epibenthic samplers can be used.

A fundamental characteristic of the community structure of submerged aquatic vegetation is the leaf area index (LAI). It is defined as the amount of photosynthetic surface per unit of biomass (U.S. EPA 1982). The photosynthetic area is measured by obtaining a two-dimensional outline of the frond, and determining the area with a planimeter. Leaf area index differences demonstrate the importance of light in regulating SAV communities and their adaptability to different light regimes. The greatest LAI values occur for mixed beds of Zostera and Ruppia; lower values were found for pure stands of Zostera and Ruppia (U.S. EPA 1982).

The information presented here is a brief overview of survey techniques used in the sampling of SAV. Supplementary discussions are found in literature by Kadlec and Wentz (1974), and Down (1983).

ESTUARINE FISH

Systems of Classification

Various authors have attempted to devise systems to classify estuarine organisms. Because salinity is the most dominant physical factor affecting the distribution of organisms, it is often used as the basis for classification systems. McLusky (1971, 1981) divides estuarine organisms into the following categories:

1. Oligohaline organisms - The majority of animals living in rivers and other fresh waters do not tolerate salinities greater than 0.1 ppt but some, the oligohaline species, persist at salinities up to 5 ppt.

2. True estuarine organisms - These are mostly animals with marine affinities which live in the central parts of estuaries. Most of them are capable of living in the sea but are not found there, apparently because of competition from other animals.

3. Euryhaline marine organisms - These constitute the majority of organisms living in estuaries with their distribution ranging from the sea into the central part of estuaries. Many disappear by 18 ppt but a few survive at salinities down to 5 ppt.

4. Stenohaline marine organisms - These occur in the mouths of estuaries at salinities down to 25 ppt.

5. Migrants - These animals, mostly fish and crabs, spend only a part of their life in estuaries with some, such as flounder (Platichthys) feeding in estuaries, and others, such as salmon (Salmo salar) or eels (Anguilla anguilla) using estuaries as routes to and from rivers and the sea.

A similar scheme of classification, shown in Table III-6, was defined by Remane. Components of fauna are separated according to the sources from which they arrived at their present-day habitat, e.g., from the sea, from freshwater and from the land. Marine and freshwater components are further divided based on salinity tolerances. The terrestrial component may be subdivided into those species which escape the effects of immersion by moving upwards when the tide floods the upper shore, and those species which remain on the shore and are able to survive submersion for several hours.

Day (1951, cited by Haedrich 1983) divided estuarine fishes into five categories: freshwater fishes found near the head of the estuary, stenohaline marine forms from the seaward end of the estuary, euryhaline marine forms occurring over wide areas, the truly estuarine fishes found only in the estuary, and migratory forms that either pass through the estuary or enter it only occasionally. A modified version of this classification was presented by McHugh (1967). His categories were:

1. Freshwater fish species that occasionally enter brackish waters.

2. Truly estuarine species which spend their entire lives in the estuary.

3. Anadromous and catadromous species.

4. Marine species which pay regular seasonal visits to the estuary, usually as adults.

5. Marine species which use the estuary primarily as a nursery ground, usually spawning and spending much of their adult life at sea, but often returning seasonally to the estuary.

6. Adventitious visitors which appear irregularly and have no apparent estuarine requirements.

Day's classification of biota and the Venice System of dividing estuaries into six salinity ranges were combined by Carriker (1967) to develop Table III-7. The right half of the table shows the biotic categories and the approximate penetration of animals relative to salinity zones in the estuary.

Salinity Preferences

Some freshwater fish species may occasionally stray into brackish waters. White catfish (Ictalurus catus) is a salt-tolerant freshwater form found in estuaries along the east coast of the United States. Three other species that are primarily freshwater, but have been captured in higher salinity areas are longnose gar (Lepisosteus osseus), bluegill (Lepomis macrochirus) and the flier (Centrarchus macropterus) (McHugh 1967).

Very few fish are considered to be truly estuarine. McHugh (1967) mentions only two species that he considers endemic to the estuarine environment. They are the striped killifish (Fundulus majalis) and the skilletfish

TABLE III-6. SUMMARY OF THE COMPONENTS OF AN ESTUARINE FAUNA

I. MARINE COMPONENT

The stenohaline marine component, not penetrating below 30 ppt
The euryhaline marine component

First grade, penetrate to 15 ppt
Second grade, penetrate to 8 ppt
Third grade, penetrate to 3 ppt
Fourth grade, penetrate to below 3 ppt

Brackish water component, lives in estuaries, but not in sea

II. FRESHWATER COMPONENT

The stenohaline freshwater component, not penetrating above 0.5 ppt
The euryhaline freshwater component

First grade, penetrate to 3 ppt
Second grade, penetrate to 8 ppt
Third grade, penetrate above 8 ppt

Brackish water component, lives in estuaries, but not in freshwater

III. MIGRATORY COMPONENT migrates through estuaries from sea to freshwater or vice versa

Anadromous, ascending rivers to spawn
Catadromous, descending to the sea to spawn

IV. TERRESTRIAL COMPONENT

Tolerant of Submersion
Intolerant of Submersion

(from Green 1967)

TABLE III-7. CLASSIFICATION OF ESTUARINE ZONES RELATING THE VENICE SYSTEM CLASSIFICATION TO DISTRIBUTIONAL CLASSES OF ORGANISMS.

Divisions of Estuary	Venice System: Salinity Ranges 0/00	Venice System: Zones	Ecological Classification: Types of Organisms and Approximate Range of Distribution in Estuary, Relative to Division and Salinities				
River	0.5	Limnetic		Limnetic			
Head	0.5-5	Oligohaline		Oligohaline			
Upper Reaches	5-18	Mesohaline	Mixohaline				
Middle Reaches	18-25	Polyhaline			True estuarine (estuarine endemics)		
Lower Reaches	25-30	Polyhaline					
Mouth	30-40	Euhaline		Stenohaline marine		Euryhaline marine	Migrants

(from Carriker 1967)

(Gobiesox strumosus). The fourspine stickleback (Apeltes quadracus) is a small fish that is abundant in estuaries but cannot be considered truly estuarine because it enters freshwater occasionally. Beccasio et al. (1980) included killifish, silverside, anchovy and hogchoker in the category of truly estuarine species. Other authors concede the existence of truly estuarine species although they fail to mention them as such. Instead, fish are categorized as spending a major portion of their life cycle in an estuary, as being dependent on the estuary at some time, or as being the dominant species present.

A listing of species commonly found in North American Atlantic/Gulf coast estuaries and their salinity tolerances/preferences as adults is contained in Table III-8. It should be noted, however, that salinity preferences of some fish may change at the time of migration. For example, adult stickleback (Gasterosteus aculeatus) prefer freshwater in March and saltwater in June/July (McLusky 1971). Salinity tolerances also differ depending on the organism's stage of life. Salinity tolerances or requirements of juveniles may be unlike those of the adult.

The Gulf of Mexico estuaries support populations of fish that are also found along the Atlantic coast. For example, spot (Leiostomus xanthurus) are abundant along the Gulf and the Atlantic coasts. The Atlantic croaker ranges from the New England States to South America, although it is basically a southern species important in the Gulf of Mexico and South Atlantic Bight. Gulf menhaden is an estuarine dependent species that primarily inhabits northern Gulf of Mexico waters. Southern kingfish (Menticirrhus americanus) have been collected along the coasts from Long

TABLE III-8. SALINITY TOLERANCE/PREFERENCE OF CERTAIN FISHES FOUND IN ATLANTIC/GULF COAST ESTUARIES

Scientific Name	Common Name	Salinity (ppt) (Tolerance/Preference)
Alosa spp.	Herring, shad, alewife	0-34/-
Brevoortia patronus	Gulf menhaden	5-35/5-10
Brevoortia tyrannus	Atlantic menhaden	1-36/5-18
Cynoscion regalis	Weakfish	-/10-34
Ictalurus catus	White catfish	<14.5/-
Ictalurus punctatus	Channel catfish	<21/<1.7
Leiostomus xanthurus	Spot	3-34/-
Menidia menidia	Atlantic silverside	0-35/-
Micropogonias undulatus	Atlantic croaker	0-40/10-34
Morone americana	White perch	0-30/4-18
Morone saxatilis	Striped bass	0-35/>12
Perca flavescens	Yellow Perch	0-13/5-7
Pomatomus saltatrix	Bluefish	7-34/-

(from U.S. EPA, 1983a)

Island Sound, New York, to Port Isabel, Texas (Sikora and Sikora 1982). They are estuarine dependent, and larval southern kingfish move from offshore spawning areas to estuarine nursery areas. Salinity preferences of southern kingfish varies with size. Only the smaller juveniles are found in waters with salinities of less than 10 ppt. Larger juveniles (>150 mm or 5.9 inches standard length, SL) are rarely taken in waters with salinities less than 20 ppt, and are usually found in deeper waters such as sounds, near the mouths of passes, or near barrier islands (Sikora and Sikora 1982). The most common fish found in Gulf of Mexico estuaries are listed in Table III-9, along with the range of salinities in which they were captured (Perret et al. 1971). Additional information on the environmental requirements of Gulf coast species is presented in Appendix D.

Appendix B contains a listing of habitat requirements of major Atlantic coast estuarine species during their life cycles. More detailed descriptions of habitat requirements of egg, larval and juvenile stages of fishes of the Mid-Atlantic bight are contained in several publications by the United States Fish and Wildlife service (1978, Volumes I-VI). Mansueti and Hardy (1967) also published information regarding fishes of the Chesapeake Bay region. These reports contain illustrations of the life stages for many species, along with pertinent information regarding preferred substrate, salinity and temperature. Although the books focus on egg, larval, and juvenile stages, the adult stage is also addressed.

Annual Cycles of Fish in Estuaries

Annual cycles and abundances of species are important in the ecology of estuaries. The composition of the estuarine fauna varies seasonally, reflecting the life histories of species. Anadromous fishes pass through

TABLE III-9. FISHES COLLECTED IN SAMPLES IN LOUISIANA ESTUARIES

Scientific Name	Common Name	Salinity (ppt) range at collection sites / range where greatest number of individuals captured
Anchoa hepsetus	Striped anchovy	7.0-29.9/>15.0
Anchoa mitchilli	Bay anchovy	0-31.5/-
Arius felis	Sea catfish	0->30.0/>10.0
Bagre marinus	Gafftopsail catfish	0-29.9/>5.0
Brevoortia patronus	Menhaden	0-30.0/5.0-24.9
Citharichthys spilopterus	Bay whiff	0->30.0/>15.0
Cynoscion nebulous	Spotted seatrout	0.2-30.0/>15.0
Dorosoma cepedianum	Gizzard shad	0-29.9/<10.0
Dorosoma pentenense	Threadfin shad	0-29.9/<5.0
Fundulus similis	Longnose killifish	0.5-30.7/>10.0
Ictalurus furcatus	Blue catfish	0-4.9/-
Leiostomus xanthurus	Spot	0.2->30.0/>10.0
Membras martinica	Rough silverside	2.0-29.9/>10.0
Menidia beryllina	Tidewater silverside	0->30.0/-
Menticirrhus americanus	Southern kingfish	2.0->30.0/>10.0
Micropogonias undulatus	Atlantic croaker	0->30.0/-
Mugil cephalus	Striped mullet	0->30.0/5.0-19.9
Paralichthys lethostigma	Southern flounder	0->30.0/-
Polydactylus ocfonemus	Atlantic threadfin	1.6-29.9/-
Prionotus tribulus	Bighead searobin	2.0->30.0/>15.0
Sciaenops ocellatus	Red drum	5.0-29.9/-
Sphaeroides nephelus	Southern puffer	1.7-30.9/>10.0
Synodus foetens	Inshore lizardfish	4.0-30.9/>10.0
Trinectes maculatus	Hogchoker	1.7-30.9/>10.0

(from Perret et al. 1971)

estuaries on the way to spawning grounds. In the Gulf of Mexico, the Alabama shad and the striped bass are important anadromous species (Beccasio et al. 1982). Both species are sought for sport. Anadromous species on the Pacific coast include chinook salmon, chum salmon, pink salmon, sockeye salmon, Dolly Varden, river lamprey and cutthroat trout (Beccasio et al. 1981, Beauchamp et al. 1983). Studies have shown that temperature is an important factor governing the timing of migrations and spawning for some species. Chinook salmon (Oncorhynchus tshawytscha) will not migrate when temperatures rise above 20°C. American shad live most of their lives at sea, but pass through estuaries to spawn in fresh water. Spawning of shad is dependent on temperature, and commences when the maximum daily water temperature reaches 16°C. It continues to about 24°C, peaking at 21°C (Jones and Stokes Assoc. 1980). Additional information on Pacific fishes is available in Hart (1973). Life history is presented along with certain environmental requirements of the species. However, salinity tolerances and preferences are noted infrequently.

Many of these anadromous species are major sport and commercial fish. Striped bass, for example, occur along the east coast of North America from the St. Lawrence River, Canada, to the St. Johns River, Florida; along the Gulf of Mexico; and from the Columbia River, Washington to Ensenada, Mexico, along the Pacific Coast (Bain and Bain 1982). Temperature was cited as a key factor in their distribution. Striped bass migrate to fresh or nearly fresh water to spawn. The optimum temperature for egg survival is 17° to 20°C. A minimum water velocity of 30 cm/s (1 fps) is necessary to prevent eggs from resting on the bottom. After hatching, the larvae remain in nearly fresh water. Striped bass larvae need a minimum of 3 mg/l dissolved oxygen. Optimum survival of larvae occurs when the temperature is between 18°C and 21°C (12°-23°C tolerated) and salinity ranges from 3-7 ppt (0-15 ppt tolerated). Juveniles are more tolerant of environmental conditions and migrate to higher salinity portions of the estuary, feeding on small prey fish. Optimum temperatures for juveniles are between 14°C and 21°C, but a range of 10°C to 27°C can be tolerated. Some adult striped bass may remain in estuaries, while others may embark on coastal migrations. Striped bass populations from Cape Hatteras, North Carolina to New England may travel substantial distances along the coast, while populations in the southern portion of the range and on the Pacific Coast tend to remain in the estuary or in offshore waters nearby (Bain and Bain 1982). It should also be noted that preferred temperatures vary depending on ambient acclimation temperatures. Striped bass acclimated to 27°C in late August avoided waters of 34°C, while 13°C was avoided by striped bass acclimated to 5°C in December.

Salmonids, numerous flatfishes and sturgeon are dependent upon Pacific coast estuaries at some time during their life cycles. For example, chum salmon spawn in rivers from northern California to the Bering Sea during October through December. Adults die after spawning. The young hatch in spring, and move to estuaries and bays where they remain for 3 to 4 months. They move to deeper waters gradually, as they grow (Beccasio et al. 1981). The sand sole, a sport species along the northwest Pacific coastline, spends up to its first year in bays and estuaries.

Some fish species utilize estuaries primarily as nursery grounds. Young fishes feed in the productive estuarine system and then migrate seaward or

TABLE III-10. FISHES THAT USE ESTUARIES PRIMARILY AS NURSERY AREAS

Scientific Name	Common Name
Alosa aestivalis	Blueback herring
Alosa pseudoharenga	Alewife
Brevoortia patronus	Gulf menhaden
Brevoortia tyrannus	Atlantic menhaden
Clupea harengus	Atlantic herring
Clupea harengus pallasii	Pacific herring
Cottus asper	Prickly culpin
Cynoscion regalis	Weakfish
Leiostomus xanthurus	Spot
Micropogonias undulatus	Atlantic croaker
Morone americana	White perch
Morone saxatilis	Striped bass
Mugil cephalus	Mullet (striped)
Mugil curema	Mullet (white)
Oncorhynchus gorbuscha	Pink salmon
Oncorhynchus kisutch	Coho salmon
Osmerus mordax	Rainbow smelt
Perca flavescens	Yellow perch
Platichthys stellatus	Starry flounder
Pseudopleuronectes americanus	Winter flounder
Salmo salar	Atlantic salmon
Trinectes maculatus	Hogchoker

(from U.S. EPA 1982, Jones and Stokes Assoc. 1981, Haedrich 1983, Beccasio et al. 1980)

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towards freshwater. Most of the fishes using estuaries as a nursery area are anadromous, the adults being principally marine. Table III-10 lists anadromous fishes (from both the east and west coasts of North America) which use estuaries primarily as nursery grounds. Although Table III-10 is not a comprehensive listing, it contains those fishes mentioned most frequently in the literature (U.S. EPA 1983a, Jones and Stokes Assoc. 1981, Haedrich 1983, Beccasio et al. 1980).

White perch (*Morone americana*), another commercially important fish, is also abundant in estuaries on the east coast of North America. Populations in the Chesapeake Bay area have been observed to inhabit the various tributaries, with some fish entering the Bay itself. The American eel (*Anguilla rostrata*) is the only catadromous species noted in the literature. It spawns in the Sargasso Sea, then migrates to and lives in estuaries or freshwaters for several years before returning to the sea.

Some fish take advantage of the complex circulation pattern of estuaries, spawning in offshore areas to allow eggs or larvae to drift up into the estuary. Most notably, the young of flatfishes (winter and starry flounder) and some of the drums (croaker, weakfish and spot) utilize the estuarine circulation system (U.S. Dept. of Interior 1970). The juveniles then feed and mature within the estuary. The gulf menhaden (*Brevoortia* patronus) supports the largest commercial fishery by weight (Christmas et al. 1982). It is an estuarine-dependent marine species that is found primarily in northern Gulf of Mexico waters. Gulf menhaden spawn from mid-October through March in marine waters. Currents transport planktonic larvae to estuarine areas, where they transform into juveniles. As they grow, juveniles migrate to deeper, more saline waters. Juveniles are able to tolerate water temperatures from 5°C to 34°C. Adults and juveniles may inhabit estuaries throughout the year. The Atlantic croaker also uses the estuary as a nursery area. Juveniles reside in salinities from 0.5 to 12 ppt, moving to higher salinity waters as they grow. They tolerate a wide range of temperatures, from 6°C to 20°C. The spot (Leiostomus xanthurus) is also estuarine dependent. Adults spawn in nearshore marine waters, but juveniles spend much of their lives in estuaries. Juvenile spot tolerate temperatures from 1.2°C to 35.5°C, preferring a range of 6°C to 20°C. They have been collected in salinities from 0 to 60 ppt, but tend to concentrate near the saltwater-freshwater boundary (Stickney and Cuenco 1982). Other estuarine-dependent species in the Gulf of Mexico are the bay anchovy, sea catfish, gafftopsoil catfish, spotted and sand seatrout, red drum, black drum, southern kingfish and southern flounder.

Some marine species enter the estuary seasonally. The spotted hake (Urophycis regins) enters the Chesapeake Bay in late fall, and exits before the warm weather. In Texas estuaries, Urophycis floridanus follows a similar migration pattern.

The bluefish (Pomatomus saltatrix) is often considered an adventitious visitor to Atlantic coast estuaries (McHugh 1967). Although the bluefish is a seasonal visitor, it may not appear if environmental conditions are not suitable. Other species may occasionally enter estuaries to feed on small fish, or if environmental conditions are suitable.

Difficulties often arise because sufficient information is not available on the life cycles of certain species to enable their classification. For this reason, and because of the many species of fish that enter estuaries only occasionally, a fully comprehensive list of species is not available. However, Haedrich (1983) compiled a listing of characteristic families found in estuaries, based upon faunal lists reported in various papers. He divided the fauna into families found in three zones, that of temperate, tropics/subtropics, and high latitudes. The families in Table III-11 include the few resident species, anadromous fisn and marine species that utilize the estuary as feeding and nursery areas.

Habitat Suitability Index Models

Habitat Suitability Index (HSI) models developed by the U.S. Fish and Wildlife Service consider the quality of habitats necessary for specific species during each life stage. The variables selected for study in a given model are known to affect species growth, survival, abundance, standing crop and distribution. Output from the models is used to determine the quantity of suitable habitat for a species. The HSI values produced by the models are relative, and should be used to compare two areas, or the same area at different times. Thus, the area with the greater HSI value is interpreted to have the potential to support a greater number of a species than that with the lower HSI. Values range from 0 to

TABLE III-11. CHARACTERISTIC FAMILIES OF ESTUARINE SYSTEMS

High Latitudes	Tropics/Subtropics
Salmonidae (salmon and trout)	Clupeidae (herrings)
Osmeridae (smelt and capelin)	Engraulidae (anchovies)
Gasterosteidae (sticklebacks)	Chanidae (milkfish)
Ammodytidae (sand lance)	Synodontidae (lizardfish)
Cottidae (sculpins)	Belonidae (silver gars)
	Mugilidae (mullets)
Temperate Zones	Polynemidae (threadfins)
Anguillidae (freshwater eels)	Sciaenidae (crockers)
Clupeidae (herrings)	Gobiidae (gobies)
Engraulidae (anchovies)	Cichlidae (cicheids)
Ariidae (saltwater catfishes)	Soleidae (flounders)
Cyprinodontidae (killifishes)	Cynoglossidae (flounders)
Gadidae (cods)	
Gasterosteidae (sticklebacks)	
Serranidae (basses)	
Sciaenidae (croakers)	
Sparidae (seabreams)	
Pleuronectidae (flounders)	

(from Haedrich 1983)

1, with 1 representing the most suitable conditions. HSI models can be used to provide one value for all life stages, or to calculate HSI values for each component (e.g. spawning, egg, larvae, juvenile, adult). There is some uncertainty in the use of the HSI models, both in the form of calculation and the fact that they are unverified models. They have not been tested to see if they work. The form of calculation leads to the possibility of their being insensitive to environmental changes. An area may have undergone great degradation before the HSI model drops in value. More information concerning HSI models can be found in Chapter IV-1 of the Technical Support Manual (U.S. EPA 1983b). Models are currently available for the following estuarine fish: striped bass (Bain and Bain 1982), juvenile Atlantic croaker (Diaz 1982), Gulf menhaden (Christmas et al. 1982), juvenile spot (Stickney and Cuenco 1982), Southern kingfish (Sikora and Sikora 1982), and alewife and blueback herring (Pardue 1983). Models have been developed for several other estuarine organisms. They are northern Gulf of Mexico brown shrimp and white shrimp (Turner and Brody 1983), Gulf of Mexico American oyster (Cake 1983), and littleneck clam (Rodnick and Li 1983).

SUMMARY

The preceding sections touch upon procedures that might be used and specific phenomena that might be evaluated during the field collection phase of a waterbody survey.

Strong seasonal changes in estuarine biological communities compound difficulties involved in collection of useful data. Because of annual cycles, important organisms can be totally absent from the estuaries for portions of the year, yet be dominant community members at other times. For example, brown and white shrimp spend part of the year in estuaries, and migrate to deeper, more saline waters as the season progresses. Furthermore, estuarine biological communities may also vary from year to year. Although it has not been mentioned explicitly, it is understood that, if at all possible, a reference site will have been identified and will have been studied in a manner that is consistent with the study of the estuary of interest. In addition to whatever field data is developed on the estuary and its reference site, it is also important to examine whatever information might exist in the historical record.

The importance of submerged aquatic vegetation has not been fully discussed in this Chapter, nor have any tools been presented by which to digest all the assessments so far presented. This will be done in Chapter IV, Interpretation.

CHAPTER IV

SYNTHESIS AND INTERPRETATION

INTRODUCTION

The basic physical and chemical processes of the estuary are introduced in Chapter II, with particular emphasis placed on a description of stratification and circulation in estuarine systems, on simplifying assumptions that can be made to characterize the estuary, on desktop procedures that might be used to define certain physical properties, and on mathematical models that are suitable for the investigation of various physical and chemical processes.

The applicability of desktop analyses or mathematical models will depend upon the level of sophisticaton required for a particular use attainability study. These types of analysis are important to the study in three ways: to help segment the estuary into zones with homogeneous physical characteristics, to help in the selection of a suitable reference estuary, and to help in the analysis of pollutant transport and other phenomena in the study area. Several case studies are presented to illustrate the use of measured data and model projections in the use attainability study. The selection of a reference estuary(ies) is discussed later in this Chapter.

Chapter II also offers a discussion of chemical phenomena that are particularly important to the estuary: the several factors that influence dissolved oxygen concentrations in surface and bottom layers and the impact of nutrient overenrichment on submerged aquatic vegetation (SAV). Other chemical evaluations are discussed in the Technical Support Manual (EPA, November 1983).

The biological characteristics of the estuary are summarized in Chapter III. Specific information on various species common to the estuary are presented to assist the investigator in determining aquatic life uses. Typical forms of estuarine flora and fauna are described and the overall importance of SAVs--as an indicator of pollution and as a source of habitat and nutrient for the biota--for the use attainability study is emphasized.

In this Chapter, emphasis is placed on a synthesis of the physical, chemical and biological evaluations which will be performed, to permit an overall assessment of uses, and of use attainability in the estuary. Of particular importance are discussions of the selection and analysis of a reference site, and the statistical analysis of the data that are developed during the use study.

USE CLASSIFICATIONS

There are many use classifications-navigation, recreation, water supply, the protection of aquatic life-which might be assigned to a water body. These need not be mutually exclusive. The water body survey as discussed in this volume is concerned only with aquatic life uses and the protection of aquatic life in a water body. Although the term "aquatic life" usually refers only to animal forms, the importance of submerged aquatic vegetation

(SAV) to the overall health of the estuary dictates that a discussion of uses include forms of plant life as well.

The use attainability analysis may also be referred to as a water body survey. The objectives in conducting a water body survey are to identify:

1. The aquatic life uses currently being achieved in the water body,

2. The potential uses that can be attained, based on the physical, chemical and biological characteristics of the water body, and

3. The causes are of any impairment of uses.

The types of analyses that might be employed to address these three points are summarized in Table IV-1. Most of these are discussed in detail elsewhere in this volume, or in the Technical Support Manual.

Use classification systems vary widely from State to State. Use classes may be based on geography, salinity, recreation, navigation, water supply (municipal, agricultural, or industrial), or aquatic life. Clearly, little information is required to place a water body into such broad categories. Far more information may be gathered in a water body survey than is needed to assign a classification, based on existing State classifications, but the additional data may be necessary to evaluate management alternatives and refine use classification systems for the protection of aquatic life in the water body.

Since there may not be a spectrum of aquatic protection use categories available against which to compare the findings of the biological survey; and since the objective of the survey is to compare existing uses with designated uses, and existing uses with potential uses, as seen in the three points listed above, the investigators may need to develop their own system of ranking the biological health of a water body (whether qualitative or quantitative) in order to satisfy the intent of the water body survey. Implicit in the water body survey is the development of management strategies or alternatives which might result in enhancement of the biological health of the water body. To do this it would be necessary to distinguish the predicted results of one strategy from another, in cases where the strategies are defined in terms of aquatic life protection.

The existing state use classifications may not be helpful at this stage, for one may very well be seeking to define use levels within an existing use category, rather than describing a shift from one use classification to another. Therefore, it may be helpful to develop an internal use classification system to serve as a yardstick during the course of the water body survey, which may later be referenced to the legally constituted use categories of the state.

A scale of biological health classes is presented in Table IV-2. This is a modified version of Table V-2 presented in the Technical Support Manual, and it offers general categories against which to assess the biology of an estuary. The classification scheme presented in Table IV-3, which was developed in conjunction with extensive studies of the Chesapeake Bay, associates biological diversity with various water quality parameters. The Toxicity Index (T_I) in the table was discussed in Chapter III.

Table IV-1. SUMMARY OF TYPICAL ESTUARINE EVALUATIONS
(adapted from EPA 1982, Water Quality Standards Handbook)

PHYSICAL EVALUATIONS	CHEMICAL EVALUATIONS	BIOLOGICAL EVALUATIONS
° Size (mean width/depth)	• Dissolved oxygen	• Biological inventory (existing use analysis)
° Flow/velocity	° Toxics	° Fish
° Total volume		- macroinvertebrates
° Reaeration rates	• Nutrients	- microinvertebrates
		° Plants
	- nitrogen	- phytoplankton
° Temperature	- phosphorus	- macrophytes
° Suspended solids	° Chlorophyll-a	
° Sedimentation	° Sediment oxygen demand	° Biological condition/ health analysis
		- diversity indices
° Bottom stability	° Salinity	
		- tissue analyses
° Substrate composition and characteristics	° Hardness	- Recovery Index
	° Alkalinity	
° Channel debris		
	° pH	° Biological potential analysis
° Sludge/sediment		
	° Dissolved solids	
° Riparian characteristics		- reference reach comparison

TABLE IV-2. BIOLOGICAL HEALTH CLASSES WHICH COULD BE USED IN WATER BODY ASSESSMENT (Modified from Karr, 1981)

Class	Attributes
Excellent	Comparable to the best situations unaltered by man; all regionally expected species for the habitat including the most intolerant forms, are present with full array of age and sex classes; balanced trophic structure.
Good	Fish invertebrate and macroinvertebrate species richness somewhat less than the best expected situation; some species with less than optimal abundances or size distribution; trophic structure shows some signs of stress.
Fair	Fewer intolerant forms of plants, fish and invertebrates are present.
Poor	Growth rates and condition factors commonly depressed; diseased fish may be present. Tolerant macroinvertebrates are often abundant.
Very Poor	Few fish present, disease, parasites, fin damage, and other anomalies regular. Only tolerant forms of macroinvertebrates are present.
Extremely Poor	No fish, very tolerant macroinvertebrates, or no aquatic life.

TABLE IV-3. A FRAMEWORK FOR THE CHESAPEAKE BAY ENVIRONMENTAL QUALITY CLASSIFICATION SCHEME

Class	Quality	Objectives	Quality	T_I	T_N	T_P
A	Healthy	supports maximum diversity of benthic resources, SAV, and fisheries	Very low enrichment	1	<0.6	<0.08
B	Fair	moderate resource diversity, reduction of SAV, chlorophyll occasionally high	moderate enrichment	1-10	0.6-1.0	0.08-0.14
C*	Fair to Poor	a significant reduction in resource diversity, loss of SAV, chlorophyll often high, occasional red tide or blue-green algal blooms	high enrichment	11-20	1.1-1.8	0.15-0.20
D	Poor	limited pollution-tolerant resources, massive red tides or blue-green algal blooms	significant enrichment	>20	>1.8	>0.20

Note: T_I indicates Toxicity Index
T_N indicates Total Nitrogen in mg l^{-1}
T_P indicates Total Phosphorus in mg l^{-1}

* Class C represents a transitional state on a continuum between classes B and D.

ESTUARINE AQUATIC LIFE PROTECTION USES

Even though the estuary characteristically supports a lesser number of species than the adjacent freshwater or marine systems, it may be considerably more productive. Accordingly, uses might be defined so as to recognize specific fisheries (and the different conditions necessary for their maintenance), and to recognize the importance of the estuary as a nursery ground and a passageway for anadromous and catadromous species. Currently the water body use classification systems of the coastal states distinguish between marine and freshwater conditions, occasionally between tidal and freshwater conditions, but seldom make reference to the estuary. Uses and standards written for marine waters presumably are intended to apply to estuarine waters as well.

It is common in these States to include as a use of marine or tidal waters the harvesting and propagation of shellfish, frequently with reference to the sanitary and bacteriological standards included in National Shellfish Sanitation Program Manual of Operations: Part 1, Sanitation of Shellfish Growing Areas, published by the Public Health Service (1965). The term shellfish applies to both molluscs and crustaceans. Other marine protection uses which may be applicable to the estuary are worded in terms such as the growth and propagation of fish and other aquatic life, preservation of marine habitat, harvesting for consumption of raw molluscs or other aquatic life, or preservation and propagation of desirable species.

In establishing a set of uses and associated criteria to be used in the water body survey, the investigator might wish to consider examples like the State of Florida's criteria for Class II (Shellfish Propagation or Harvesting) and Class III (Propagation and Maintenance of a Healthy, Well-Balanced Population of Fish and Wildlife) Waters published in the Water Quality Standards of the Florida Department of Environmental Regulation. The published criteria are extensive and include the following categories which are of importance to the estuarine water body survey:

Biological Integrity - the Shannon-Weaver diversity index of benthic macroinvertebrates shall not be reduced to less than 75 percent of established background levels as measured using organisms retained by a U.S. Standard No. 30 sieve and collected and composited from a minimum of three natural substrate samples, taken with Ponar type samplers with minimum sampling areas of 225 square centimeters.

Dissolved Oxygen - the concentration in all waters shall not average less than 5 milligrams per liter in a 24-hour period and shall never be less than 4 milligrams per liter. Normal daily and seasonal fluctuations above these levels shall be maintained.

Nutrients - In no case shall nutrient concentrations of a body of water be altered so as to cause an imbalance in natural populations of aquatic flora or fauna.

SELECTION OF REFERENCE SITES

General Approach. There is a detailed discussion of the selection of reference or control sites in Chapter IV-6 of the Technical Support Manual. Although this discussion was prepared in the context of stream and lake studies, much of the material is pertinent to the study of estuaries as well. Riverine water body surveys may range in scale from a specific well-defined reach to perhaps an entire stream. One might expect to find a similar range of scale in estuary studies. The lateral bounds of the riverine study area generally are delineated by but not necessarily limited to the stream banks. The specification of a reference reach is prescribed by the scale of the study. If a short reach is under study, the reference reach might be designated upstream of the study area. If an entire river is under review, another river will have to be identified that will serve as an appropriate control.

An estuarine study may focus on a specific area, but the bounds of the study area are not easily defined because a physical counterpart to the river bank may not exist. Other factors compound the difficulties in designing an estuary study compared to the design of a river study. A major difference is that estuary segments cannot be so easily categorized because of seasonal changes in the salinity profile. Partitioning the estuary into segments with relatively uniform physical characteristics is an important first step of a water body survey.

It may be possible to study a small estuary as a single segment, but it will be necessary to go elsewhere for a reference site. This may be easily accomplished among the many bar built estuaries of the southeastern coast. For the large estuary, one may need only to examine a well-defined segment which has been affected by a point source discharge. If the segment is an embayment tributary to the main stem of the estuary, it may not be difficult to find a suitable control embayment within the same estuary. As the scale of the study increases, however, the difficulties associated with the establishment of a reference site also increases. It may not make sense to treat the entire estuary as a single unit for the use attainability survey, especially if use categories are associated with salinity ranges, different depths, etc. In such a case one would segment the estuary based upon physical characteristics such as salinity levels and circulation patterns, and then define the reference site in similar fashion. As a practical matter, it may not make sense to examine an entire estuary as a single unit, especially a large one. For example, the Chesapeake Bay has been subjected to a form of use attainability studies for a number of years at a cost of many millions of dollars. However, Chesapeake Bay is so complex that, despite the intensity of study, clear explanations are not always possible for the many undesirable changes that have taken place. The Chesapeake Bay itself is unique and no suitable reference estuary exists. From the use attainability standpoint, an estuary such as the Chesapeake or the Delaware or the Hudson is best broken down into segments that are homogeneous in characteristics and manageable in size.

Statistical Comparisons of Impact Sites With Control Sites. Reference site comparisons typically rely upon either parametric or nonparametric statistical tests of the null hypothesis to determine whether water quality or any other use attainment indicator at the impact site is significantly different from conditions at the control site(s).

Parametric statistics, which are suitable for datasets that exhibit a normal distribution, include the F (folded)-statistic on the difference between the variances at the impact site and control site and the t-statistic on the difference between the means. In order to conclude that there is no significant difference between the water quality conditions (or another indicator) at the impact site and the control site, both the F-statistic and the t-statistic should exhibit probabilities exceeding the 0.05 probability cutoff for the 95 percent confidence interval. In cases where the impact site is being compared with multiple control sites, parametric procedures such as the Student-Newman-Keuls (SNK) test, the least significant difference (LSD) test, and the Duncan's Multiple Range test can be used to test for differences among the grouped means.

Since water quality datasets are often characterized by small sample sizes and non-normal distributions, it is likely that nonparametric statistical tests may be more appropriate for the monitoring database. Nonparametric statistics assume no shape for the population distribution, are valid for both normal and non-normal distributions, and have a much higher power than parametric statistical techniques for analyses of datasets which are characterized by small sample sizes and skewed distributions. The one-sided Kolmogorov-Smirnov (K-S) test can be used to quantify whether each dataset is normally (or lognormally) distributed, thereby governing the selection of either parametric or nonparametric procedures. If nonparametric procedures are selected, significant differences in distributions can be evaluated with the two-sided K-S test, while significant differences in the central value can be tested with the Wilcoxon Ranksum test. Both nonparametric tests should exhibit probability values exceeding the cutoff for the 95 percent confidence interval in order to conclude that there is no significant difference in water quality conditions at the impact site and a control site. For comparisons with multiple control sites, nonparametric procedures such as the Kruskal-Wallis test and the Friedman Ranksum test can be used to test for significant differences among medians (if it can be assumed that the distributions of each dataset are not significantly different.

The same types of statistical tests can be used to evaluate sediment and biological monitoring data to determine whether suitable conditions for use attainability exist at the impact site. Either parametric or nonparametric statistical procedures can be used to compare conditions at the impact site and control site(s) which are unaffected by effluent discharge or other pollution sources. In cases where there are no statistically significant differences in distributions and/or control values, it may be assumed that sediment and/or biological monitoring results at the impact site and control site(s) are similar.

CURRENT AQUATIC LIFE PROTECTION USES

The actual aquatic protection uses of a water body are defined by the resident flora and fauna. The prevailing chemical and physical attributes will determine what biota may be present, but little need be known of these attributes to describe current uses. The raw findings of a biological survey may be subjected to various measurements and assessments, as discussed in Section IV (Biological Evaluations) of the Manual. After performing an inventory of the flora and fauna and considering a diversity index or other indices of biological health, one should be able adequately to describe the condition of the aquatic life in the water body.

CAUSES OF IMPAIRMENT OF AQUATIC LIFE PROTECTION USES

If the biological evaluations indicate that the biological health of the system is impaired relative to a "healthy" reference aquatic ecosystem (e.g., as determined by reference site comparisons), then the physical and chemical evaluations can be used to pinpoint the causes of that impairment. Figure IV-1 shows some of the physical and chemical parameters that may be affected by various causes of change in a water body. The analysis of such parameters will help clarify the magnitude of impairments to attaining other uses, and will also be important to the third step in which potential uses are examined.

ATTAINABLE AQUATIC LIFE PROTECTION USES

A third element to be considered is the assessment of potential uses of the water body. This assessment would be based on the findings of the physical, chemical and biological information which has been gathered, but additional study may also be necessary. A reference site comparison will be particularly important. In addition to establishing a comparative baseline community, defining a reference site can also provide insight into the aquatic life that could potentially exist if the sources of impairment were mitigated.

The analysis of all information that has been assembled may lead to the definition of alternative strategies for the management of the estuary at hand. Each such strategy corresponds to a unique level of protection of aquatic life, or aquatic life protection use. If it is determined that an array of uses is attainable, further analysis which is beyond the scope of the water body survey would be required to select a management program for the estuary.

One must be able to separate the effects of human intervention from natural variability. Dissolved oxygen, for example, may vary seasonally over a wide range in some areas even without anthropogenic effects, but it may be difficult to separate the two in order to predict whether removal of the anthropogenic cause will have a real effect. The impact of extreme storms on the estuary, such as Hurricane Agnes on the Chesapeake Bay in 1972, may completely confound our ability to distinguish the relative impact of anthropogenic and natural influences on immediate effects and longterm trends. In many cases the investigator can only provide an informed guess. Furthermore, if a stream does not support an anadromous fishery because of dams and diversions which have been built for water supply and recreational purposes, it is unlikely that a concensus could be reached to restore the fishery by removing the physical barriers -- the dams -- which impede the migration of fish. However, it may be practical to install fish ladders to allow upstream and downstream migration. Another example might be a situation in which dredging to remove toxic sediments may pose a much greater

SOURCE OF MODIFICATION

WATER QUALITY PARAMETERS

	Acid Mine Drainage or Acid Precipitation	Sewage Treatment Plant Discharge (primary or secondary)	Agricultural Runoff (pasture or cropland)	Urban Runoff	Channelization	(Industries) Pulp and Paper	Textile	Metal Finishing and Electroplating	Petroleum	Iron and Steel	Paint and Ink	Dairy and Meat Products	Fertilizer Production and Lime Crushing	Plastics and Synthetics
pH	D					C	I	C		D	C		D,I	C
Alkalinity	D						I						D,I	
Hardness	I						I						I	
Chlorides		I		I								I		
Sulfates	I								I	I			I	I
TDS	I						I		I		I		I	
TKN		I	I	I					I	I		I	I	I
NH_3-N		I							I	I			I	I
Total-P		I	I	I				I				I	I	
Ortho-P		I		I				I					I	
BOD_5		I				I	I		I	I	I	I		I
COD	I	I		I		I	I			I	I	I		I
TOC		I	I	I		I			I	I	I	I		I
COD/BOD_5	I			I		I	I		D	I				
D.O.		D				D						D		
Aromatic Compounds			I	I		I			I					I
Fluoride								I					I	I
Cr				I		I	I	I	I		I			I
Cu	I			I		I		I			I			I
Pb				I				I	I	I	I			
Zn	I			I		I		I	I	I	I			I
Cd		I		I				I			I			I
Fe	I			I				I		I	I		I	I
Cyanide										I				I
Oil and Grease						I	I	I	I	I	I	I		I
Coliforms	D	I	I	I				D		D	D	I		
Chlorophyll	D	I	I		I	D		D	D	D	D	I	I	D
Diversity	D	D		D	D	D	D	D	D	D	D			D
Biomass	D	I	I		I		I	D	D	D	D	I	I	D
Riparian Characteristics					C									
Temperature					I									
TSS			I	I	I	I	I	I			I		I	I
VSS				I		I								
Color						I	I				I	I		I
Conductivity	I												I	
Channel Characteristics					C									

Figure IV-1. Potential Effects of Some Sources of Alteration on Water Quality Parameters; D = Decrease, I = Increase, C = Change threat to aquatic life than to do nothing. Under the do nothing alternative, the toxics may remain in the sediment in a biologically-unavailable form, whereas dredging might resuspend the toxic fraction, making it biologically available and also facilitating wider distribution in the water body.

The points touched upon above are presented to suggest some of the phenomena which may be of importance in a water body survey, and to suggest the need to recognize whether or not they may realistically be manipulated. Those which cannot be manipulated essentially define the limits of the highest potential use that might be realized in the water body. Those that can be manipulated define the levels of improvement that are attainable, ranging from the current aquatic life uses to those that are possible within the limitations imposed by factors that cannot be manipulated.

RESTORATION OF USES

Uses that have been impaired or lost in an estuary can only be restored if the conditions responsible for the impairment are corrected. Impairment can be attributed to pollution from toxics or overenrichment with nutrients. Uses may also be lost through such activities as the disposal of dredge and fill materials which smother plant and animal communities, through overfishing which may deplete natural populations, the destruction of freshwater spawning habitat which will cause the demise of anadromous species, and natural events in the sea, such as the shifting of ocean currents, that may alter the migration routes of species which visit the estuary at some time during the life cycle. One might expect losses due to natural phenomena to be temporary although man-made alterations of the estuarine environment may prevent restoration through natural processes.

Assuming that the factors responsible for the loss of species have been identified and corrected, efforts may be directed towards the restoration of habitat followed by natural repopulation, stocking of species if habitat has not been harmed, or both. Many techniques for the improvement of substrate composition in streams have been developed which might find application in estuaries as well. Further discussion on the importance of substrate composition will be found in the Technical Support Manual (EPA, November 1983).

Stocking with fish in freshwater environments, and with young lobster in northeastern marine environments, is commonly practiced and might provide models for restocking in estuaries. In addition, aquaculture practices are continually being refined and the literature on this subject (Bardach et al., 1972) should prove helpful in developing plans for the restoration of estuaries or parts of estuaries.

Submerged aquatic vegetation (SAV) is considered to be an excellent indicator of the overall health of an estuary because it is sensitive to environmental degradation caused by physical smothering, nutrient enrichment and toxics. Because SAV is so important as habitat and as a source of nutrient for a wide range of the estuarine biota, its demise signals the demise of its dependent populations. If uses in an estuary have been impaired or lost, it is likely that SAV will also have been affected.

Unfortunately, the cause of SAV degradation is not always clear. In the Chesapeake Bay for instance, controversy persists as to the cause of loss of SAV and the loss of biota which depend to whatever extent on SAV. Trends noted over time in the demise of these populations may conceivably be related to trends in toxic, sediment and nutrient loadings on the Bay, and to trends in the release of chlorinated wastewaters from POTWs, chlorinated effluents from industry and chlorinated cooling water from powerplants. Areas in which SAV has been adversely impacted are areas where there are toxics in the sediment and/or where algal blooms prevent light from reaching SAV communities.

The ability to restore areas of SAV will depend upon the initial causes of loss, and the ability to remove the causes. Toxics in sediment may be a particularly difficult problem because of the impracticality of dredging large areas to remove contaminated bottom substrate. An inabilty to remove toxic sediments which may have caused a decline in SAV and other benthic communities severely limits the likelihood that these populations may be restored to past levels.

The control of nutrients may be a much more tractable problem. If nutrient inputs to the estuary can be controlled, SAV populations may begin to expand on their own. In the Potomac River estuary, phosphorus removal at the Blue Plains wastewater treatment plant, which serves the greater Washington, D.C. area, has resulted in sharp reductions in algal blooms which are considered a major factor in the demise of SAV within the Chesapeake Bay system.

Apart from natural processes which result in the enlargement of areas of SAV, SAV may be restored through reseeding and transplanting, depending upon the species. Generally speaking, reseeding may not be a practical approach because of the cost of collecting seeds and because one would not expect all seeds to survive, although _Vallisneria_ (wild celery) shows some promise in using seeds to reestablish populations. Some areas may reseed naturally, but in many cases SAV populations may be too distant for the natural transport of seeds to be likely. In these cases, plants may be transplanted in order to restore SAV. Reestablishment is accomplished by transplanting shoots and rhizomes.

Although transplanting may be a more practical alternative, the outcome is not assured. In an effort to reestablish SAV, plugs of _Zostera_ (eelgrass) and _Potamogeton_ (sage pondweed, redhead grass) were planted in the Potomac River estuary. These beds showed some measure of success, depending mainly upon the substrate present. The transplanting of SAV is a labor intensive operation and as such would require a considerable cost in time and resources to restore even a small area.

In Tampa Bay, Florida, stress on the ecosystem, including the disposal of dredge spoils which have smothered SAV communities, has caused a significant loss (25,220 ha, or 81 percent) of submergent wetland vegetation. Efforts to reestablish _Spartina_ (cord grass) and _Thalassia_ (turtlegrass) have resulted in the restoration of about 11 ha of vegetation (the growth and spreading of rhizomateous material is increasing this figure) (Hoffman et al., 1982). The transplantation of _Thalassia_ and _Halodule_ (shoalgrass) near the discharge side of a powerplant was less successful, in that

Thalassia failed to survive for 30 days where the mean water temperature was 31°C or greater, and only small patches of shoalgrass survived near the outer edges of the thermal plume. These differences could not be attributed to differences in sediment composition (Blake et al., 1976). Nevertheless, other transplantation efforts emphasize the importance of substrate to plant survival. For example, *Thalassia* prefers a reduced environment while *Halodule* prefers an oxidized substrate.

Transplanting oyster spat from "seed" areas which are protected from harvesting to areas less favorable for reproduction is a relatively common practice. Seed areas ideally exhibit optimum salinity and temperature for oyster reproduction and spat set. Clean shell is deposited as substrate in seed areas and spat often become very densely populated. Spat are then moved to areas where an oyster population is desired. Steps may also be taken to prepare the bottom (often by depositing oyster shells) where an oyster reef exists, or where attempts will be made to establish an oyster reef.

Although there has been some progress in the aquacultural sciences towards rearing species that may be found in the estuary (clam, quahog, oyster, scallop, shrimp, crab, lobster, flatfish), techniques are not well-advanced and there is little likelihood that they could be successfully applied on any scale towards the repopulation of the estuary. As with SAV, the experiments and the successes with the reestablishment of species are limited, and the more important factor in the restoration of habitat is the control and reversal of the various forms of pollution which cause the demise of estuarine populations.

CHAPTER V

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APPENDIX A

DEFINITION OF THE CONTAMINATION INDEX (C_I) AND THE TOXICITY INDEX (T_I)

To assess the contribution of anthropogenic sources of metal contamination over time, sediment cores may be analyzed. The Wedepohl ratio compares the amount of metal in the sediment sample with the concentration in an average shale (or sandstone). In the Chesapeake Bay program, scientists have measured silicon and aluminum, then correlated metals with Si/Al ratios. A contamination factor (Cf) may be computed as follows:

$$Cf = (Co-Cp)/Cp$$

where: Co = surface sediment concentration
Cp = predicted concentration, derived from the statistical relation between the Si/Al ratio and the log metal content of old, pre-pollution sediments from the estuary.

Thus, Cf < 0 when the observed metal concentration is less than the predicted value; Cf = 0 when observed and predicted are the same; Cf > 0 when the observed is greater than the predicted value.

The Contamination Index (C_I) is found by summing contamination factors for metals in a given sediment.

Then,

$$C_I = \sum_{n=1}^{n} Cf = \sum_{n=1}^{n} (Co-Cp)/Cp$$

The Toxicity Index (T_I) is related to the Contamination Index and is expressed by the following equation:

$$T_I = \sum_{i=1}^{i} (M_1/M_i)\cdot Cf_i$$

where: M_i = the "acute" anytime EPA criterion for any of the metals,
but M_1 is always the criterion value for the most toxic of the metals.

The "acute" anytime EPA criterion is defined as the concentration of a material that may not be exceeded in a given environment at any time. When evaluating Toxicity Indices, sampling stations should be characterized by their minimum salinities. This is because the toxicity of metals is often *greater in freshwater than in saltwater.*

A more detailed discussion of the development of the Contamination Index may be found in the U.S. EPA publication, Chesapeake Bay: A Profile of Environmental Change (1983a) and A Framework for Action (1983c).

APPENDIX B

LIFE CYCLES OF MAJOR SPECIES OF ATLANTIC COAST ESTUARIES

Contents

(from U.S.EPA 1983*a*)

TABLE 1a. ENVIRONMENTAL TOLERANCES OF ALOSA AESTIVALIS (BLUEBACK HERRING) CANADIAN MARITIMES TO ST. JOHN'S RIVER, FL

LIFE STAGE	HABITAT REQUIREMENTS	FOOD AND FEEDING FACTORS	GROWTH & DEVELOPMENT FACTORS	BEHAVIOR	PREDATORS AND COMPETITORS	SELECTED REFERENCES
Eggs	Tidal-fresh and low-brackish water. Eggs are found in streams and rivers with swift currents and sandy or rocky substrate.	Not applicable	No information	Not applicable	No information	Burbidge 1974 Hudson and Hardy 1974 Jones et al. 1978 Lippson et al. 1979
Larvae	Tidal-fresh and brackish water. Larvae are found in tributary streams and upper portions of rivers. Optimum salinity 0-5 ppt.	- copepods	Growth occurs during warm temperatures.	Interspecific competition with Bay anchovy in brackish water causes larvae to select food items other than the preferred type.	Compete with Bay anchovy. Prey of predatory fish (striped bass, white perch)	Domermuth and Reed 1980 Raney and Massmann 1953
Juvenile	Tidal-fresh and brackish water. Juveniles are found primarily in surface waters. Tolerate salinity 0-28 ppt. Optimum salinity 0-5 ppt.	Selective feeder during daylight. - copepods - copepodites - Bosmina spp. - macrozooplankton	Growth occurs during warm temperatures; rate of growth is more rapid than for alewives.	Young juveniles remain in nursery area until the fall, then undertake a seaward migration. Young may remain in the lower Bay during first or second winter.	Prey of predatory fish (striped bass, white perch, bluefish)	
Adult	0-34 ppt salinity. Adults enter the Bay to spawn in freshwater; return to the ocean after spawning.	- zooplankton - crustaceans - crustacean eggs - insects - fish eggs and larvae	Blueback herring mature in 3-4 yrs., and reach a maximum length of 38.0 cm.	Schooling herring occur in a narrow band of coastal water; move to the bottom during winter. Herring are anadromous, migrating into the Bay to spawn in spring.	Prey of predatory fish (striped bass, bluefish, weakfish) in fresh, brackish, & salt water. Target of a commercial & recreational fishery.	

TABLE 1b. ENVIRONMENTAL TOLERANCES OF ALOSA PSEUDOHARENGUS (ALEWIFE) NEWFOUNDLAND TO SOUTH CAROLINA

LIFE STAGE	HABITAT REQUIREMENTS	FOOD AND FEEDING FACTORS	GROWTH & DEVELOPMENT FACTORS	BEHAVIOR	PREDATORS AND COMPETITORS	SELECTED REFERENCES
Eggs	0-0.5 ppt salinity. Eggs are released in slow, shallow portions of creeks and rivers over detritus or sandy substrate.	Not applicable	Hatching period 6 days. Mean water temp. 60°F.	Not applicable	No information	Jones et al. 1978 Shea et al. 1980 Lippson et al. 1979 Hildebrand and Schroeder 1928
Larvae	0-3 ppt salinity. Larvae remain in vicinity of spawning area at depths less than 3m.	- rotifers - copepod nauplii	No information	Form schools within 1-2 days after hatching.	Prey of predatory fish (white perch and striped bass)	
Juvenile	Tolerate salinity 0-34 ppt. Optimum salinity 0.5-5 ppt. Young juveniles are found in nursery areas from shore to shore; as the fish grow, there is a slow downstream movement.	- copepods - mysid shrimp	Grow very rapidly, possibly due to entering salt water, average 105 mm.	Young juveniles migrate toward the ocean in the fall, some overwinter in deep areas of the Bay.	Prey of predatory fish (bluefish, striped bass, white perch)	
Adult	0-34 ppt salinity. Adults enter the Bay to spawn in freshwater; return to ocean by mid-summer.	Mid-water feeder - copepods - young fish - zooplankton - mysids	Alewife mature in 3-4 yrs., measuring an average 25.0-30.0 cm in length.	Schooling alewife show regular anadromous Alosid coastal movements. Alewife are anadromous, migrating into the Bay to spawn in spring.	Prey of predatory fish (striped bass, bluefish, weakfish). In fresh, brackish, and salt water. Target of commercial and recreational fishery.	

TABLE 1c. ENVIRONMENTAL TOLERANCES OF *ALOSA SAPIDISSIMA* (AMERICAN SHAD) GULF OF ST. LAWRENCE TO FLORIDA

LIFE STAGE	HABITAT REQUIREMENTS	FOOD AND FEEDING FACTORS	GROWTH & DEVELOPMENT FACTORS	BEHAVIOR	PREDATORS AND COMPETITORS	SELECTED REFERENCES
Eggs	0-0.5 ppt salinity. Streams and rivers with swift currents and sandy or rocky substrate.	Not applicable	Temperatures above 21°C and low D.O. levels decrease hatching success.	Not applicable	No information	Hildebrand and Schroeder 1928 Shea et al. 1980 Domermuth and Reed 1980 Lippson et al. 1979 Ellis et al. 1947
Larvae	Optimum salinity 0-5 ppt. Larvae are found at depths greater than 3m.	No information	At D.O. levels of 5 ppm, some stress and mortality occurs; at D.O. levels of 4 ppm, high mortality may occur.	No information	Preyed upon by top predatory species (striped bass, bluefish, white perch, other herring spp.)	
Juvenile	Tolerate salinity 0.5-12 ppt. Optimum salinity 5-12 ppt. Young juveniles gradually move into more saline waters.	Feed at or beneath surface - daphnid cladocerans - bosminid cladocerans - other cladoceran spp. - copepods	Young grow rapidly during the first summer.	Juveniles remain in natal streams and rivers until the fall, then undertake a seaward migration. Some remain in the lower Bay during the first winter.	Competition with species such as the alewife or blueback herring influence location of feeding fish & selection of prey. Prey of top predatory species.	
Adult	Tolerate salinity 0-34 ppt. Adults enter the Bay to spawn in freshwater or on flats in tidal waters; return to ocean after spawning.	Feed in surface layer - copepods - small fish - planktivorous crustaceans - insects	Growth rate decreases after 3 years of age. Reach sexual maturity in 4-5 years.	Shad are anadromous, migrating into the Bay to spawn in spring. Nests are built, but no parental care is given to eggs.	Prey of top predatory fish (bluefish, striped bass). Target of a commercial and recreational fishery.	

TABLE 1d. ENVIRONMENTAL TOLERANCES OF <u>BREVOORTIA</u> <u>TYRANNUS</u> (ATLANTIC MENHADEN) NOVA SCOTIA TO GULF OF MEXICO

LIFE STAGE	HABITAT REQUIREMENTS	FOOD AND FEEDING FACTORS	GROWTH & DEVEL-OPMENT FACTORS	BEHAVIOR	PREDATORS AND COMPETITORS	SELECTED REFERENCES
Eggs	Eggs are released in the ocean, probably not far (as far as 64 km) from the mouth of the Bay.	Not applicable	No information	Not applicable	No information	Pristas and Willis 1973 Shea et al. 1980 June and Carlson 1971 Durbin and Durbin 1975 Lippson et al. 1979
Larvae	Early larvae tolerate 18-34 ppt salinity. Optimum salinity 25-34 ppt. Later they concentrate in tidal fresh to low brackish waters (0-3 ppt salinity).	Sight-selective feeders - copepods size of fish influences size of copepods taken.	No information	Larvae enter the Bay in spring when they are about 10-30 mm long; may reach nursery areas in larval or juvenile stage.	No information	
Juvenile	Tolerate salinity 0-34 ppt. Optimum salinity 0-15 ppt. Younger fish concentrate in tidal-fresh to low-brackish waters.	Filter feeder - phytoplankton	No information	Young-of-the-year juveniles remain in the Bay during summer; may leave in fall or over-winter in Bay.	Prey of top preda-tory fish including bluefish and striped bass.	
Adult	Tolerate salinity 1-36 ppt concentrate in areas of 5-18 ppt salinity where food patches occur. One and two year old adults utilize the Bay; older fish remain off the coast.	Filter feeder - zooplankton - larger phyto-plankton - longer chains of chain-forming diatoms. Feeding behavior is linked to food density and par-ticle size.	Some fish may reach maturity in one year; all fish are mature by age 3. Maximum length around 47.0 cm.	Schooling marine fish which enter the Bay in spring to feed; most migrate seaward in the fall, though some may overwinter in the lower Bay.	Prey of top pre-datory fish in-cluding bluefish and striped bass. Target of a com-mercial fishery.	

TABLE 1e. ENVIRONMENTAL TOLERANCES OF CALLINECTES SAPIDUS (BLUE CRAB) NEW JERSEY TO FLORIDA

LIFE STAGE	HABITAT REQUIREMENTS	FOOD AND FEEDING FACTORS	GROWTH & DEVELOPMENT FACTORS	BEHAVIOR	PREDATORS AND COMPETITORS	SELECTED REFERENCES
Eggs	Hatch at salinities of 10.3-32.6 ppt; optimum salinities for hatch are 23-30 ppt. Females carry the eggs until hatch occurs.	Not applicable	Salinity affects hatching success.	Not applicable	No information	Van Engel et al. 1973 Shea et al. 1980 Sulkin 1975 Van Engel 1958 Sandoz and Rogers 1944 Lippson 1971 Lippson et al. 1979
Zoeae	Tolerate salinities of 15.8-32.3 ppt; optimum salinities are 21-28 ppt. Zoeae are found in the upper surface water.	- rotifers - Nauplii larvae - sea urchin larvae - polychaete larvae	Zoeae molt at least three times, with the final mold producing a megalops. Molting is affected by salinity, temperature, larval concentrations, and light intensity.	Zoeae show an attraction to light.	No information	
Megalops	Optimum salinities of 20-35 ppt. Megalops may be found in surface waters or on the bottom.	Omnivorous - plants - fish and shellfish pieces - detritus Availability of prey affects diet.	Salinity and temperature affect the duration of the megalops stage. Megalops metamorphose into a small juvenile crab.	Megalops and juveniles move into the Bay through the entrainment in bottom waters, beginning in fall. In winter young crabs cease migrations and burrow into channel bottoms.	No information	
Juveniles and Adults	Juveniles concentrate in brackish water with salinities less than 20 ppt. Adult males concentrate in salinities of 3-15 ppt. Females concentrate in salinities of 10-28+ ppt.	- benthic organisms - small fish - plants - shellfish - small crustaceans - detritus Availability of prey affects diet.	Crabs reach sexual maturity in 12-20 months depending on timing of hatch. Growth occurs by shedding the shell, and is regulated by water temperature.	In warm weather, juveniles move inshore. When temperatures drop, juveniles move to channel areas to overwinter in semi-hibernation. Adults have similar movement patterns.	- predatory fish such as striped bass and bluefish - birds such as herons and herring gulls - a commercial and recreational fishery.	

TABLE 1f. ENVIRONMENTAL TOLERANCES OF CRASSOSTREA VIRGINICA (AMERICAN OYSTER) NEW ENGLAND TO GULF COAST

LIFE STAGE	HABITAT REQUIREMENTS	FOOD AND FEEDING FACTORS	GROWTH & DEVELOPMENT FACTORS	BEHAVIOR	PREDATORS AND COMPETITORS	SELECTED REFERENCES
Eggs	Optimum salinity of 22.5 ppt; below 10 ppt, survival is poor. Pelagic eggs released in open water.	Not applicable	Turbidity levels of 125 mg L^{-1} or more reduce development and survival of eggs.	Not applicable	No information	Galtsoff 1964 Haven and Morales-Alamo 1970 Korringa 1952 Davis and Calabrese 1964 Ukeles 1971 Andrews 1967, 1968 Haven, personal communication
Larvae	Optimal growth occurs at salinities of 12.5-25.0 ppt.	Filter feeder - phytoplankton - bacteria The size of food particles taken is a function of the mouth size.	Turbidity levels of 100 mg L^{-1} cause high larval mortality. Salinity, temperature, and available food influence larval development.	Oyster larvae move within the estuary by entrainment in bottom waters. Larvae search for suitable substrate on which to attach in about two weeks. At setting, larvae metamorphose to spat.	Prey of planktonic-feeding fish and invertebrates.	
Juveniles (spat)	Salinity 5-35 ppt. Oysters are found in shallow water less than 10 meters deep. Optimum survival of oysters occurs on hard substrate such as rocks, pilings, and oyster shells in the intertidal and sub-tidal zones.	Filter feeder - phytoplankton - bacteria - detritus	Spat exhibit rapid growth during the first year. Growth rates are affected by availability of food, salinity, and water temperature.	Oysters initially develop as males, yet by the second breeding season many change into females.	Competitors - boring sponges and clams - slipper shell - sea squirt - barnacles - spirochaetes - perforating algae Predators - oyster drills - blue crabs - starfish - birds - commercial fishery Diseases - *Perkinsus marinus* (Dermo) - *Menchinia nelsoni* (MSX)	
Adults		Filter feed on 1-12 micron prey - phytoplankton - bacteria - detritus Turbidity and low temperatures influence feeding and digestion.	Growth is affected by substrate type, salinity, temperature, tidal flow, and crowding. Oysters reach sexual maturity during the second year of growth. [A few reach maturity at one year (Haven)]	Epibenthic with frequent alternation of sex. Form communities or "bars." Oyster distribution in higher salinity areas is restricted by predators and parasites.		

TABLE 1g. ENVIRONMENTAL TOLERANCES OF CYNOSCION REGALIS (WEAKFISH) MASSACHUSETTS TO FLORIDA

LIFE STAGE	HABITAT REQUIREMENTS	FOOD AND FEEDING FACTORS	GROWTH & DEVELOPMENT FACTORS	BEHAVIOR	PREDATORS AND COMPETITORS	SELECTED REFERENCES
Eggs	Tolerate salinities of 5-34 ppt. Buoyant eggs are released in the near-shore and estuarine zones along the coast.	Not applicable	Eggs are susceptible to low D.O. levels and sudden changes in either salinity or temperature.	Not applicable	No information	Lippson et al. 1979 Daiber et al. 1976 Wilk 1978 McHugh 1978
Larvae	Tolerate salinities 12-31 ppt. Larvae remain in the general vicinity of spawning.	No information	Larvae cannot withstand sudden changes in either salinity or temperature; a 5°C change in temperature in either direction can be fatal.	No information	No information	
Juvenile	About 0-34 ppt salinity. Young-of-the-year fish move into low salinity areas over soft, muddy bottoms.	- shrimp - other crustacean spp. - bay anchovy - young menhaden - other small fish	Weakfish grow most rapidly during their first year, reaching an average length of 19 cm.	Young juveniles move into low salinity areas for the summer; migrate to the coast in fall, and move offshore and south in the winter. Begin schooling as pre-adults.	Preyed upon by bluefish, striped bass, and large weakfish.	
Adult	Tolerate salinities of 10-34 ppt. Adults remain in the lower portion of the Bay.	Primarily piscivorous - menhaden - herring spp. - bay anchovy - silversides - crustaceans - annelids	Weakfish are sexually mature in 2-3 years, and reach an average length of about 50.0 cm.	Adults school, arrive in Bay in spring, leave by late fall and head south and offshore for the winter; return north to inshore areas in spring.	Preyed upon by bluefish and striped bass. The target of a commercial and recreational fishery.	

TABLE 1h. ENVIRONMENTAL TOLERANCES OF CYNOSCION NEBULOSUS (SPOTTED SEATROUT) DELAWARE TO MEXICO

LIFE STAGE	HABITAT REQUIREMENTS	FOOD AND FEEDING FACTORS	GROWTH & DEVEL-OPMENT FACTORS	BEHAVIOR	PREDATORS AND COMPETITORS	SELECTED REFERENCES
Eggs	Spawning occurs at salinities of 30-35 ppt. Hatched in 40 hrs at 25°C. Eggs reported as both demersal and pelagic, released in deeper channels and holes adjacent to grassy bays and flats.	Not applicable	Eggs are susceptible to low D.O. and sudden changes in salinity or temperature.	Not applicable	No information	Tabb 1961 Arnold et al. 1978 Fable et al. 1978 Idyll and Fahy 1975 Lorio and Perret 1980
Larvae	Growth of larvae is rapid, about 4.5 mm in 15 days after hatching. Young fish spend their juvenile life in vegetated flats, moving to deeper water in winter.	Very small invertebrates, including copepods, mysid shrimp, and post-larval penaeid shrimp.	Highly sensitive to changes in temperature. Winter-time cold shock and high temperature changes causes kills.	Tend to remain close to site of spawning in grassy flats.	No information	
Juvenile	Fish larger than 2 inches show a tendency to congregate in schools. Remain in grassy, shallow water flats until colder weather causes them to move to deeper water.	As the trout grow, diet changes to include larger portions of caridean shrimp and then to penaeid shrimp.	Females grow faster than males but males attain sexual maturity at a smaller size. Growth is rapid in first year with lengths of 13 cm attained by the first winter and 25 cm their second winter.	Start to school as young fish but remain in general area of nursery grounds until cold weather causes them to move to deeper water.	Reported as highly cannibalistic in the post-larval stage.	

(continued)

TABLE 1h. (CONTINUED)

LIFE STAGE	HABITAT REQUIREMENTS	FOOD AND FEEDING FACTORS	GROWTH & DEVELOPMENT FACTORS	BEHAVIOR	PREDATORS AND COMPETITORS	SELECTED REFERENCES
Adult	While tagging studies show that some seatrout travel as much as 315 miles, most studies show that few fish leave their natal estuary. *C. nebulosus* occupies a more southern, warmer water habitat than does *C. regalis*.	Listed as the top carnivore in most estuarine communities. As an adult, will eat all other fish of a smaller size as well as shrimp and small crabs.	Longevity indicated to be 8 to 9 years of age. Generally mature at one to three years with 50% sexually mature by end of second year (25 cm in length). All fish appeared to have spawned by age three. A 1978 report cites the largest seatrout caught was 16 pounds.	Movement patterns have been traced to the presence or absence of penaeid shrimp. Seasonal movements correspond to water temperature and spawning season.	A top predator which would be in competition with other predators such as bluefish and striped bass. both commercial and recreational fisheries.	

TABLE 11. ENVIRONMENTAL TOLERANCES OF *ICTALURUS CATUS* (WHITE CATFISH) NEW YORK TO FLORIDA

LIFE STAGE	HABITAT REQUIREMENTS	FOOD AND FEEDING FACTORS	GROWTH & DEVELOPMENT FACTORS	BEHAVIOR	PREDATORS AND COMPETITORS	SELECTED REFERENCES
Egg	Freshwater Eggs deposited in nests built near sand or gravel banks in still or running water.	Not applicable	Eggs need to be aerated.	Not applicable	No information	Jones et al. 1978 Lippson et al. 1979
Larvae	In freshwater, may move into tidal water.	No information	Yolk sac larvae bypass larval stage, develop directly to juvenile stage.	No information	No information	Daiber et al. 1976 Kendall and Schwartz 1968
Juvenile	No information	No information	Growth continues at 11 ppt salinity or less.	Remain in schools until end of first summer; initially guarded by parents.	No information	
Adult	Maximum salinity of 14.5 ppt Widespread in Bay. Prefer heavily silted bottom. Inhabit river channels and streams with slow current, ponds, and lakes.	Omnivorous, solitary, bottom feeder -plant material -small fish -clams and snails -worms -insects -dead material	Fish mature in one to two years. Maximum length 61.0 cm.	Stay in waters greater than 3 m, overwinter in deeper water (15 m), move upstream to spawn in freshwater. Males guard and aerate egg masses.	No information	

TABLE 1j. ENVIRONMENTAL TOLERANCES OF ICTALURUS NEBULOSUS (BROWN BULLHEAD) SOUTHERN CANADA TO SOUTHERN FLORIDA

LIFE STAGE	HABITAT REQUIREMENTS	FOOD AND FEEDING FACTORS	GROWTH & DEVEL-OPMENT FACTORS	BEHAVIOR	PREDATORS AND COMPETITORS	SELECTED REFERENCES
Egg	Freshwater Eggs deposited in nests in sand or gravel at depths of several inches to several feet.	Not applicable	Eggs exposed to direct sunlight produce poor hatches. Eggs need to be agitated.	Not applicable	No information	Jones et al. 1978 Lippson et al. 1979 Daiber et al. 1976
Larvae	Freshwater Found at bottom	No information	Yolk-sac larvae bypass larval stage, develop directly to juvenile stage.	Grouped in a tight mass at bottom.	No information	
Juvenile	Found among vegetation or other cover over muddy bottoms.	No information	No information	Young juveniles herded in schools by parents; may remain in schools throughout first summer.	No information	
Adult	Adults are widespread throughout most of the Bay area, occurring in channels and shallow, muddy water around aquatic vegetation. Maximum salinity 10 ppt.	Omnivorous, solitary bottom feeder - plant material - small fish - clams and snails - worms - insects - dead material	Mature at 3 years. Maximum length around 50.8 cm.	A schooling bottom species which is active primarily at night. Fish may burrow in soft sediments. Adults attend eggs and orally agitate.	No information	

TABLE 1k. ENVIRONMENTAL TOLERANCES OF ICTALURUS PUNCTATUS (CHANNEL CATFISH) HUDSON BAY REGION TO NORTHERN MEXICO

LIFE STAGE	HABITAT REQUIREMENTS	FOOD AND FEEDING FACTORS	GROWTH & DEVELOPMENT FACTORS	BEHAVIOR	PREDATORS AND COMPETITORS	SELECTED REFERENCES
Egg	Eggs 1 to 2 days old tolerate salinity to 10 ppt; 3 days and older 16 ppt.	Not applicable	No information	Not applicable	No information	Jones et al. 1978 Lippson et al. 1979 Daiber et al. 1976
Larvae	Tolerate salinities up to 8 ppt.	No information	Abnormal development occurs at temperatures above 35°C. Yolk-sac larvae bypass larval stage, develop to juvenile stage.	Larvae guarded by male first few days after hatching.	No information	
Juvenile	Tolerate salinities up to 11-12 ppt.	Feed at surface	Growth continues at 11 ppt salinity or less.	Remain in schools up to several weeks. Show strong schooling and hiding tendencies in first year.	No information	
Adult	Maximum salinity of 21.0 ppt, prefer less than 1.7 ppt. Restricted distribution in Bay. - deeper channels of large rivers with sluggish or swift current.	Omnivorous, solitary, bottom feeder - plant material - small fish - clams and snails - worms - insects - dead material	Mature in 2 to 9 years. Maximum length around 120.2 cm.	Males construct nests and guard eggs.	No information	

TABLE I 1. ENVIRONMENTAL TOLERANCES OF LEIOSTOMUS XANTHURUS (SPOT) MASSACHUSETTS TO FLORIDA

LIFE STAGE	HABITAT REQUIREMENTS	FOOD AND FEEDING FACTORS	GROWTH & DEVEL-OPMENT FACTORS	BEHAVIOR	PREDATORS AND COMPETITORS	SELECTED REFERENCES
Egg	Eggs are released over the continental shelf.	Not applicable	No information	Not applicable	Jellyfish, such as the sea walnut (Mnemiopsis leidyi), predatory marine fish.	Hudson and Hardy 1974 Shea et al. 1980 Lippson et al. 1979 Thomas 1971 Chao and Musick 1977 Peters and Kjelson 1975
Larvae	Tolerate salinity 0-35 ppt. Optimum salinity 0-5 ppt in the estuary.	Sight-selective feeder - planktonic copepods	No information	No information	Prey of predatory fish and birds	
Juvenile	Tolerate salinity 0-34.2 ppt. Post-larvae and young fish concentrate at salinities of 0.5-5.0 ppt; during years of high population density young may move into freshwater. Prefer muddy substrate.	Bottom feeder - benthic harpacticoid copepods - annelids - plant material	Growth during first summer is rapid, juveniles may measure 13 cm by late fall.	Post-larvae are carried into the Bay in April through entrainment in bottom waters. School along shore during summer. Young move downstream as they grow.	Same as above	
Adult	8-34 ppt salinity. Occur at depths greater than 1 m over soft muddy bottom; larger fish prefer channel waters.	Bottom feeder - burrowing polychaetes - annelids - small crustaceans - molluscs - macrozooplankton	Reach sexual maturity by the third year; maximum length around 33-35 cm.	Adults enter the Bay in April and May, leave for spawning grounds offshore from Aug. through Nov.	Prey of large gamefish (striped bass), sharks, and the target of recreational and commercial fisheries.	

TABLE 1m. ENVIRONMENTAL TOLERANCES OF MERCENARIA MERCENARIA (HARD CLAM) NOVA SCOTIA TO YUCATAN

LIFE STAGE	HABITAT REQUIREMENTS	FOOD AND FEEDING FACTORS	GROWTH & DEVELOPMENT FACTORS	BEHAVIOR	PREDATORS AND COMPETITORS	SELECTED REFERENCES
Eggs	Tolerate 20-35 ppt salinity, prefer 26.5-27.5 ppt.	Not applicable	Salinity affects egg development.	Eggs are carried on currents in the Bay.	No information	Lippson 1973 Daiber et al. 1976
Larvae	Salinities greater than 17.5 ppt. Larvae are pelagic, found in the surface waters.	No information	Larval development is affected by salinity, temperature, turbidity, and circulation patterns.	Larvae are initially pelagic, but toward the end of this stage, they alternate between a planktonic and benthic existence.	Clam larvae are prey of other filter feeding organisms.	Shea et al. 1980 Castagna and Chanley 1973
Juvenile	Optimum salinity 24-28 ppt, survive salinities as low as 12.5 ppt.	Filter feeder - algae species - detritus	Growth rates vary with the type of substrate used; faster growth occurs in coarser sediments.	Young clams have bisexual gonads, usually dominated by male characteristics. After the first spawning season, about 50% of the juveniles become female.	Predators include - oyster drills - blue crabs - moon snails - conchs - horseshoe crabs - sea stars - puffers - waterfowl - cow nosed rays - drum fish - man	
Adult	Salinities greater than 15 ppt. Hard clams occur in subtidal or intertidal waters with solid substrate (shell or rock).	Filter feeder - algae species	Large clams measure 12-13 cm in length.	Adults spawn during neap tides; spawning may be both thermally and chemically stimulated.		

TABLE 1n. ENVIRONMENTAL TOLERANCES OF MICROPOGONIAS UNDULATUS (ATLANTIC CROAKER) CAPE COD, MA TO FLORIDA

LIFE STAGE	HABITAT REQUIREMENTS	FOOD AND FEEDING FACTORS	GROWTH & DEVELOPMENT FACTORS	BEHAVIOR	PREDATORS AND COMPETITORS	SELECTED REFERENCES
Eggs	Eggs are released in the ocean near the mouth of the Bay from August through December.	Not applicable	No information	Not applicable	No information	Shea et al. 1980 Hildebrand and Schroeder 1928 Lippson et al. 1979 Stickney et al. 1975 Chao and Musick 1977 Haven 1957 Joseph 1972 Wallace 1940
Larvae	Larvae which enter the Bay in fall remain in channel waters at depths greater than 3m; carried to the salt water interface.	No information	No information	Larvae begin entering the Bay in fall through entrainment in bottom waters.	No information	
Juvenile	Young juveniles are found in channel waters of 0-21 ppt salinity. Older fish tend to be down-river from the younger fish.	Juveniles less than 10 cm - harpacticoid copepods Older juveniles - polychaetes - crustaceans - fish - other invertebrates	No growth occurs during the winter season; young fish have been killed during intensive cold periods on the nursery grounds.	Yearling croaker leave in the fall.	Striped bass predation on overwintering juveniles may depress the population; juveniles also preyed on by bluefish.	
Adult	Tolerate salinity 0-40 ppt. Optimum salinity 10-34 ppt. Hard bottom at depths greater than 3m.	- small crustaceans - annelids - molluscs - small fish	Croaker reach a maximum length of around 50 cm.	Croaker enter the Bay in spring, remaining in the lower estuary until fall, then they migrate back to sea. Water temperature influences croaker migrations.	Prey of top predatory species (striped bass and bluefish). The target of a commercial and recreational fishery.	

TABLE 1o. ENVIRONMENTAL TOLERANCES OF *MORONE AMERICANA* (WHITE PERCH) NOVA SCOTIA TO SOUTH CAROLINA

LIFE STAGE	HABITAT REQUIREMENTS	FOOD AND FEEDING FACTORS	GROWTH & DEVELOPMENT FACTORS	BEHAVIOR	PREDATORS AND COMPETITORS	SELECTED REFERENCES
Egg	Tolerate salinity 0-6 ppt. Eggs are released in tidal-fresh to low-brackish waters in shallows along the shore.	Not applicable	Suspended sediment levels about 1500 ppm increase incubation period.	Not applicable	No information	Shea et al. 1980 Lippson et al. 1979 Hildebrand and Schroeder 1928
Larvae	Tolerate salinity 0-8 ppt, prefer 0-1.5 ppt. Maximum depth 12 ft. Larvae are found in shallow water over sand or gravel bars or mud bottom.	Sight-selective feeders - rotifers - cladocerans - copepods	Temperature and availability of rotifers affects development of yolk-sac larvae.	Remain in spawning area, settle to bottom. General downstream movement as larvae develop.	Compete with striped bass larvae in nursery areas. Preyed upon by fish (striped bass) and birds.	Hudson and Hardy 1974 Loos 1975 Mansueti 1961
Juvenile	Tolerate salinity 0-13 ppt, prefer 0-3 ppt. Found in shallow sluggish water over silt, mud, or vegetation; move to sandy shoals and beaches at night.	- copepods - cladocerans - insect larvae	Growth positively correlated with temperature and solar radiation. Growth influenced by population density.	Juveniles remain in nursery area at least until 20 mm long, may remain until 1 year old. Juveniles may form large schools.	Compete with striped bass juveniles. Preyed upon by fish (striped bass, bluefish) and birds.	
Adult	Tolerate salinity 0-30 ppt, prefer 4-18 ppt. In summer, concentrate near shoals, occasionally in channel areas. In winter, found in deeper water; move to channels during coldest periods.	Bottom oriented, piscivorous - smelt - yellow perch - young eels - young striped bass - insects - crustaceans	Growth rates decrease with age and high population density. Males mature in 2 years, females in 3.	Schooling adults are resident to the Bay. White perch are semi-anadromous, making spawning migrations upstream in spring.	Preyed on by larger fish (striped bass, bluefish). Also the target of a commercial and recreational fishery.	

TABLE 1p. ENVIRONMENTAL TOLERANCES OF *MORONE SAXATILIS* (STRIPED BASS) ST. LAWRENCE RIVER, CANADA TO ST. JOHN'S RIVER, FL

LIFE STAGE	HABITAT REQUIREMENTS	FOOD AND FEEDING FACTORS	GROWTH & DEVELOPMENT FACTORS	BEHAVIOR	PREDATORS AND COMPETITORS	SELECTED REFERENCES
Egg	Tolerate salinity 0-10 ppt. 1.5-3 ppt optimal. 1.0-2.0 m sec^{-1} optimum flow rate. Semi-buoyant eggs released in fresh to brackish water.	Not applicable	Salinity and temperature influence development.	Not applicable	Prey of white perch.	Setzler et al. 1980 Boynton et al. 1981 Beaven and Mihursky 1980
Larvae	Tolerate salinity 0-15 ppt. 5-10 ppt optimal. 0.3-1.0 m sec^{-1} optimal flow rate. - open waters - at 13 mm, move inshore for first summer	Sight selective feeder - copepods - rotifers - cladocerans High prey concentrations necessary for successful first feeding.	Temperature and adequate food influence growth.	Positively phototrophic; newly-hatched larvae sink between swimming efforts; at 2-3 days of age larvae can swim continuously.	Compete with white perch larvae in nursery area.	Hollis 1952 Doroshev 1970 Shea et al. 1980 Md. Dept. Nat. Res. 1981
Juvenile	Juveniles 50-100 mm. Tolerate salinity 0-35 ppt. Optimal 10-20 ppt. 0-1 m sec^{-1} optimal flow rate. - prefer sandy substrate but found over gravel bottoms as well in shallow waters.	Non-selective feeder - insect larvae - polychaetes - larval fish - amphipods - mysids	Temperature and population density influence growth.	Downstream movement of young-of-the-year fish. Yearlings school in rivers or move into lower estuary in summer.	Compete with white perch in nursery. Prey of predatory fish, birds, mammals, and man.	
Adult	Tolerate 0-35 ppt, usually in salinities greater than 12 ppt. Summer habitat includes high energy shorelines with a current. Overwinter in channels in estuary or offshore at depths below 6 m.	Piscivorous - alewife - blueback herring - white perch - spot - menhaden - bay anchovy - croaker	Temperature, age, population density, and oxygen levels influence growth.	Andromous, migrate to freshwater to spawn, return to lower estuary or ocean after spawning. Young females (2-3 yr) migrate along coast in summer with older fish.	Compete with bluefish, weakfish, and white perch. Commercial and recreational fishery for striped bass.	

TABLE 1q. ENVIRONMENTAL TOLERANCES OF MYA ARENARIA (SOFT SHELL CLAM) LABRADOR TO NORTH CAROLINA

LIFE STAGE	HABITAT REQUIREMENTS	FOOD AND FEEDING FACTORS	GROWTH & DEVELOPMENT FACTORS	BEHAVIOR	PREDATORS AND COMPETITORS	SELECTED REFERENCES
Eggs	Eggs are released by sedentary adults in two spawning peaks, spring and fall.	Not applicable	No information	Not applicable	No information	Shea et al. 1980 Lucy 1977 Merrill and Tubiash 1970 Wallace et al. 1965 Castagna and Chanley 1973 Matthiessen 1960
Larvae	Minimum salinity for larval survival is 8 ppt.	Filter feeder - naked flagellates - other microscopic plankton spp.	Temperature influences larval development; at 10°C, larval development is slow.	After the planktonic larvae develop sufficiently, they metamorphose to adult form and settle to the bottom.	No information	
Juvenile	Juveniles occur over a broader depth range than adults.	Suspension feeder - phytoplankton - microzooplankton - bacteria - detritus	Juvenile clams are sensitive to salinity fluctuations.	Juveniles can move about by using the muscular foot or by currents. They establish a permanent burrow when one inch long.	Predators include: - blue crab - oyster drills - horseshoe crabs - cow-nosed rays - herring gulls - waterfowl - bottom feeding fish - commercial and recreational fisheries.	
Adult	Tolerate salinity 3-35 ppt. Optimum 16-32 ppt. Clams occur on shallow subtidal beds in stable substrates at depths less than 6-10 m.	Suspension feeder - phytoplankton - microzooplankton - bacteria - detritus	Clams reach sexual maturity in one year. Growth is influenced by water currents, food supply, temperature, and sediment type.	Adults occur in deep, permanent burrows in shallow water.		

TABLE 1r. ENVIRONMENTAL TOLERANCES OF PERCA FLAVESCENS (YELLOW PERCH) EAST COAST RANGE OF NOVA SCOTIA TO SOUTH CAROLINA

LIFE STAGE	HABITAT REQUIREMENTS	FOOD AND FEEDING FACTORS	GROWTH & DEVELOPMENT FACTORS	BEHAVIOR	PREDATORS AND COMPETITORS	SELECTED REFERENCES
Egg	0-0.5 ppt salinity. Non-tidal and tidal-fresh water.	Not applicable	Low temperatures during spawning season cause an extended incubation period (2-3 wks); larvae more developed at hatch than other anadromous species.	Not applicable	No information	Setzler et al. 1980 Lippson et al. 1979 Auld and Schubel 1974 Daiber et al. 1976 Muncy 1962
Larvae	Tolerate salinity 0-2 ppt. Optimum 0-0.5 ppt. Shallow, freshwater; survival reduced when sediment concentrations exceed 500 mg L^{-1}.	- plankton	Salinities greater than 2 ppt interfere with larval development.	Larvae move downstream after hatching; concentrate near surface, form schools.	Preyed upon by white perch, striped bass, chain pickerel.	
Juvenile	0.5-10 ppt, concentrate at salinities of 5-7 ppt in summer. Found in vegetated areas near shore.	- small crustaceans - insects - worms - molluscs	Grows quickly during first year; growth rate decreases with age. Females have greater growth rate than males.	Initially concentrate at surface, become demersal at about 25 mm.	Preyed upon by fish such as white perch and striped bass, birds, mammals. Compete with white perch and striped bass.	
Adult	Tolerate 0-13 ppt salinity, prefer 5-7 ppt in summer. Prefer higher salinity, tidal waters with muddy substrate.	- bay anchovies - silversides - minnows - isopods - amphipods - snails - crustaceans	Males mature at 1 year of age, females mature at age 2 or 3; grow to 53 cm. Large populations cause stunting of adults.	Spring migration upstream to spawn; return downstream after spawning.	Competes with smaller fish and invertebrates for food. Preyed upon by birds (mergansers), fish (gars and pikes), and man.	

TABLE 1a. LIFE HISTORY OF POMATOMUS SALTATRIX (BLUEFISH) NOVA SCOTIA TO ARGENTINA

LIFE STAGE	HABITAT REQUIREMENTS	FOOD AND FEEDING FACTORS	GROWTH & DEVEL-OPMENT FACTORS	BEHAVIOR	PREDATORS AND COMPETITORS	SELECTED REFERENCES
Eggs	Eggs released off-shore in two distinct waves; spring spawning occurs in the Gulf Stream, while summer spawning occurs over the continental shelf.	Not applicable	No information	Not applicable	No information	Lippson et al. 1979 Hildebrand and Schroeder 1928 Jones et al. 1978 Daiber et al. 1976
Larvae	No information	No information	No information	No information	No information	
Juvenile	0-37.5 ppt salinity. The larger the juvenile population, the greater the penetration into the Bay.	- copepods - molluscs - planktivorous crustaceans - any fish smaller than themselves	Juveniles grow quickly during the first summer.	Juveniles from spring spawning enter the Bay in early summer; leave the Bay by late fall, heading offshore and southward.	No information	
Adult	7-34 ppt salinity. Both sexually mature and immature adults enter the Bay; the larger the adult population, the greater the penetration into the Bay.	Voracious predator - menhaden - silversides - bay anchovy - herring spp. - crustaceans - annelids	Bluefish are sexually mature at about 30.0 cm, and reach a maximum length of 93.4 cm.	Bluefish, a marine species, enters the Bay in spring and summer to feed. Schools of bluefish move seasonally in relation to food abundance.	Compete with other top predators such as striped bass. Target of a commercial and recreational fishery.	

APPENDIX C

SUBMERGED AQUATIC VEGETATION

Compiled from Stevenson and Confer 1978.

APPENDIX C

SUBMERGED AQUATIC VEGETATION

Ceratophyllum demersum (Coontail)

Characea: _Chara_, _Nitella_, _Toypellas_

Elodea canadensis (Common elodea)

Myriophyllum spicatum (Eurasian watermilfoil)

Najas guadalupensis (Bushy pondweed)

Potamogeton pectinatus (Sago pondweed)

Potamogeton perfoliatus (Redhead grass)

Ruppia maritima (Widgeongrass)

Vallisneria americana (Wild celery)

Zannichellia palustris (Horned pondweed)

Zostera marina (Eelgrass)

Ceratophyllum demersum (Coontail)

References

Distribution

Frequents quiet, freshwater pools and slow streams. Also in the Maryland portion of the Chesapeake Bay. — Mason 1969

Temperature

Critical minimum temperature for vegetative growth of 20°C, with optimum growth at 30°C. — Wilkinson 1963

Salinity

Essentially freshwater, but grows normally in salinities under $6.5^{\circ}/_{\circ\circ}$. — Bourn 1932

Substrate

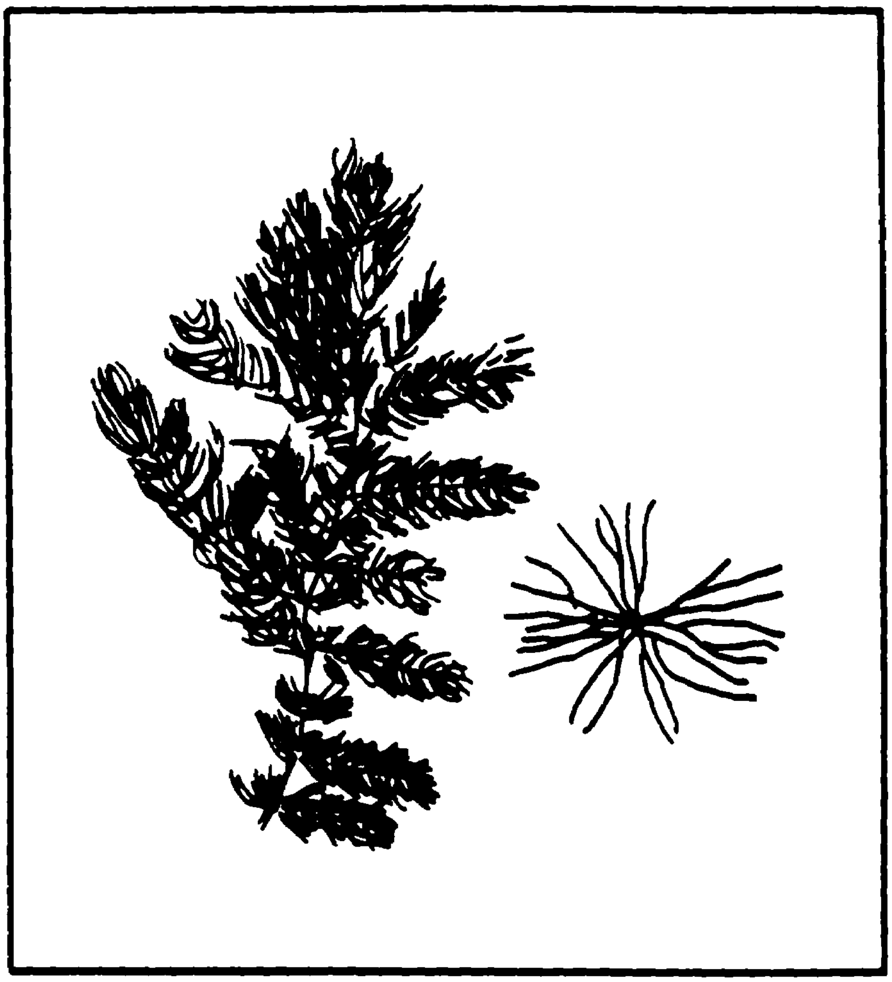
Often grows independently of substrate material. — Sculthorpe 1967

Light, Depth and Turbidity

Shade tolerant, requiring a minimum of 2 percent full sunlight for optimum growth. Not considered to be depth limited due to its rootless nature. Turbidity is not as detrimental for coontail as for rooted vegetation because of shade tolerance and water surface habitat. — Chapman et al. 1974

Ceratophyllum demersum (Coontail)
Continued

	References
Consumer Utilization	
Foliage and seeds rated as having great importance to ducks, coots, geese, grebes, swans, waders, shore and game birds. Moderate importance as fish food, shade, shelter and spawning medium.	Sculthorpe 1967



(copied from Hotchkiss 1967)

Figure 1. Coontail (<u>Ceratophyllum demersum</u>)

Characea: Chara, Nitella, Tolypellas

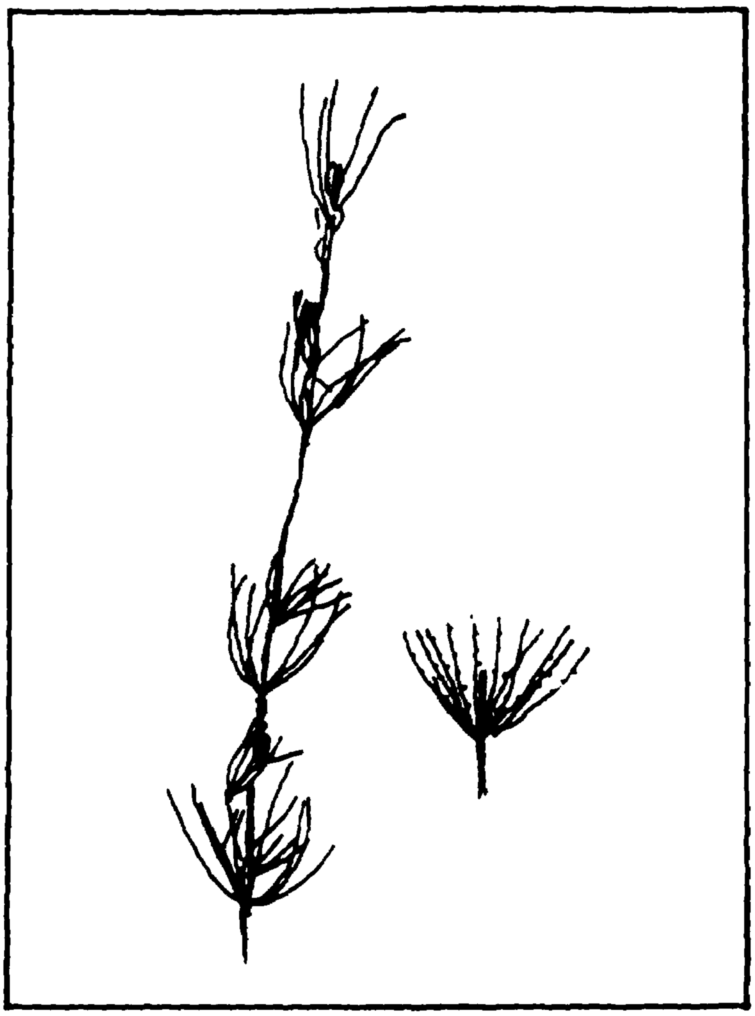
	References
Distribution	
Primarily found in freshwater environments. Some species inhabit brackish waters but are not found in truly marine environments. Found in temperate and tropical regions of all the continents.	Hutchinson 1975 Cook et al. 1974
Temperature	
Germination of Characea occurs after maintenance at 40°C for one to three months.	Hutchinson 1975
Salinity	
Certain species ranged in salinities up to 15°/₀₀ with growth cessation and limited survival at 20°/₀₀.	Dawson 1966
Substrate	
Most species of Characea grow in silt or mud substrate though a small number of species tend to grow in shallow water on sandy bottoms.	Hutchinson 1975
Light, Depth and Turbidity	
The Characea are capable of surviving in low light intensities. Have been found inhabiting fresh water at depths up to 65.5 m (Lake Tahoe), with incident	Hutchinson 1975

Characea: <u>Chara</u>, <u>Nitella</u>, <u>Tolypellas</u>
Continued

	References
radiation of slightly more than 2 percent of that reaching the lake surface.	

Consumer Utilization

Consumed by many kinds of ducks, especially diving ducks. Also provides habitat for aquatic fauna.	Martin and Uhler 1939



(copied from Hotchkiss 1967)

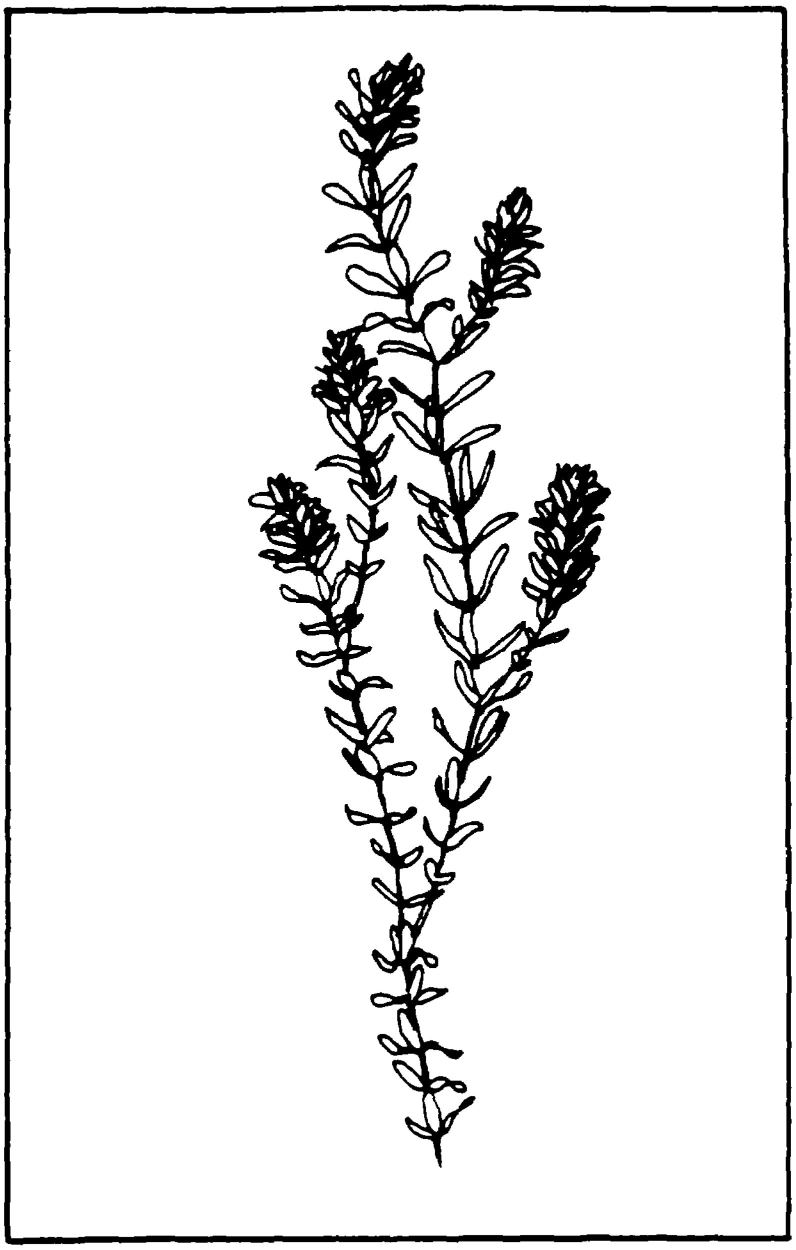
Figure 2. Muskgrass (Chara sp.)

Elodea canadensis (Common elodea)

	References
Distribution	
Endemic to North America and naturalized to many industrialized nations of Europe and the southern hemisphere.	
Temperature	
Water temperatures of 15 to 18°C are necessary for successful growth.	Yeo 1965b
Salinity	
Salinity range of fresh water to brackish water of 10°/₀₀.	U.S. Army Corps of Engineers 1974
Substrate	
Prefers a soil to sand substrate. Grows better when rooted than when suspended.	Yeo 1965b Hutchinson 1975
Light, Depth and Turbidity	
Maximum frequency of elodea is between 3.0 m and 7.5 m depth. Capable of quickly growing up through covering layers of silt.	Hutchinson 1975

Elodea canadensis (Common elodea)
Continued

	References
Consumer Utilization	
Has little value to water fowl. Generally unpalatable to aquatic insects. Epiphytes grow abundantly between the teeth on the leaf margins and on the upper leaf surfaces.	Martin and Uhler 1939 Hutchinson 1975



(copied from Hotchkiss 1967)

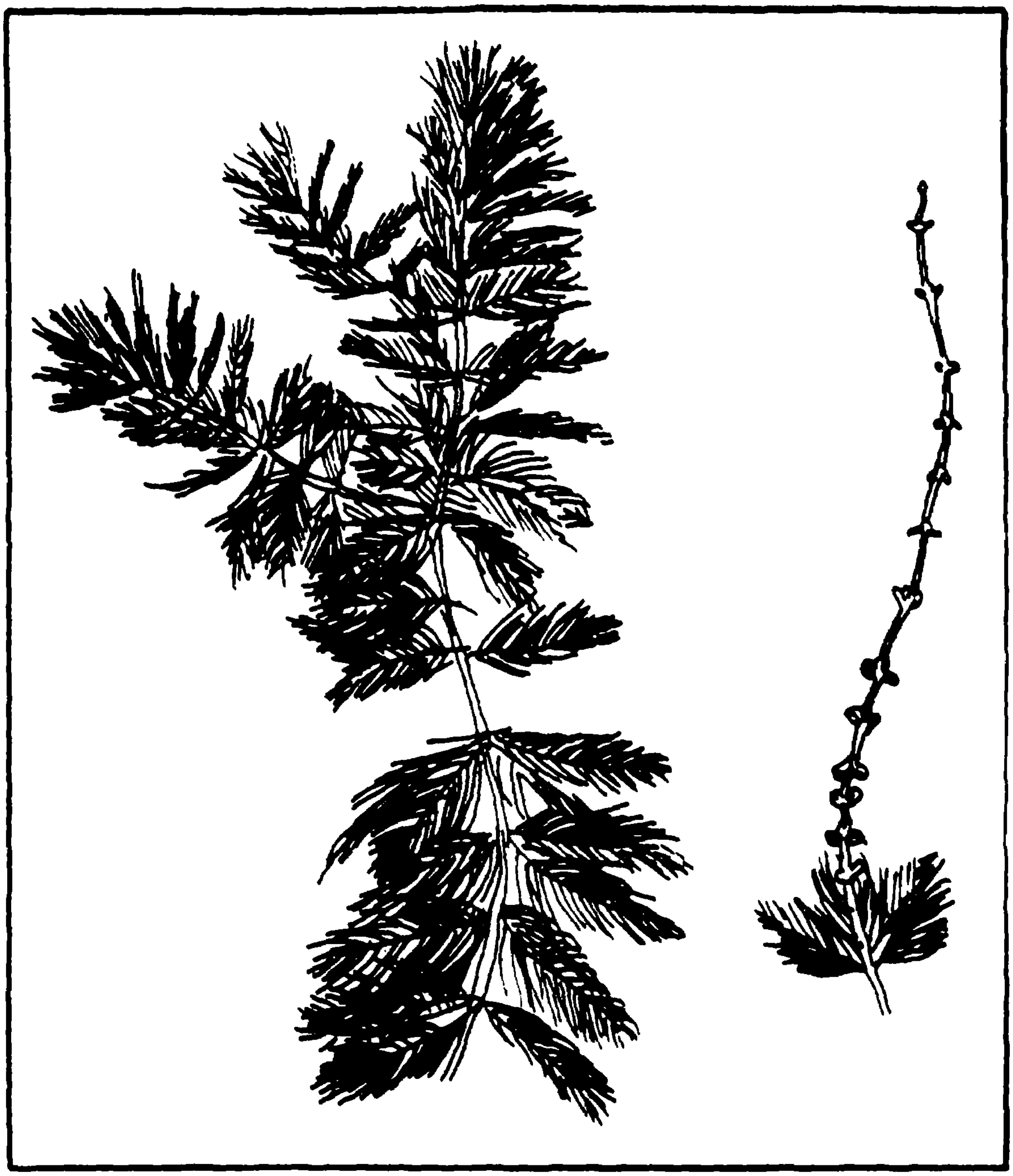
Figure 3. Common elodea (Elodea canadensis)

Myriophyllum spicatum (Eurasian watermilfoil)

	References
Distribution	
Native to Europe and Asia, is widespread in Europe, Asia and parts of Africa. Found in Chesapeake Bay area, also infested many lakes in New York, New Jersey and Tennessee.	Anonymous 1976 Springer 1959 Springer et al. 1961 Stotts 1961
Temperature	
Found growing in temperatures ranging from 0.1° to 30°C.	Anderson 1964 Anderson et al. 1965
Salinity	
Found in salinities ranging from 0 to 20°/₀₀. Grows best in salinities of 0 to 5 °/₀₀. Inhibition starts at 10°/₀₀ and becomes severe from 15 to 20°/₀₀.	Rawls 1964 Boyer 1960
Substrate	
Grows best in soft muck or sandy muck bottoms. Maximum density coincides with fine organic ooze while minimum density is found in sand.	Patten 1956 Anderson 1972 Steenis et al. 1967 Philipp and Brown 1965 Springer 1959
Light, Depth and Turbidity	
Sensitive to turbidity and grows in water more than 2 m deep, if clear. Limited to 1.5 m in extremely turbid waters.	Southwick 1972 Titus et al. 1975

Myriophyllum spicatum (Eurasian watermilfoil)
Continued

	References
Consumer Utilization	
Low grade duck food. Found in digestive tracts of 27 Canada Geese, 6 species of dabbling ducks, 4 species of divers and 31 coots in the vicinity of Back Bay and Currituck Sound. Offers support for aufwuchs which later become food for higher life forms. Crowds out more desirable foods.	Florschutz 1973 Martin et al. 1951 Springer 1959 Springer et al. 1961

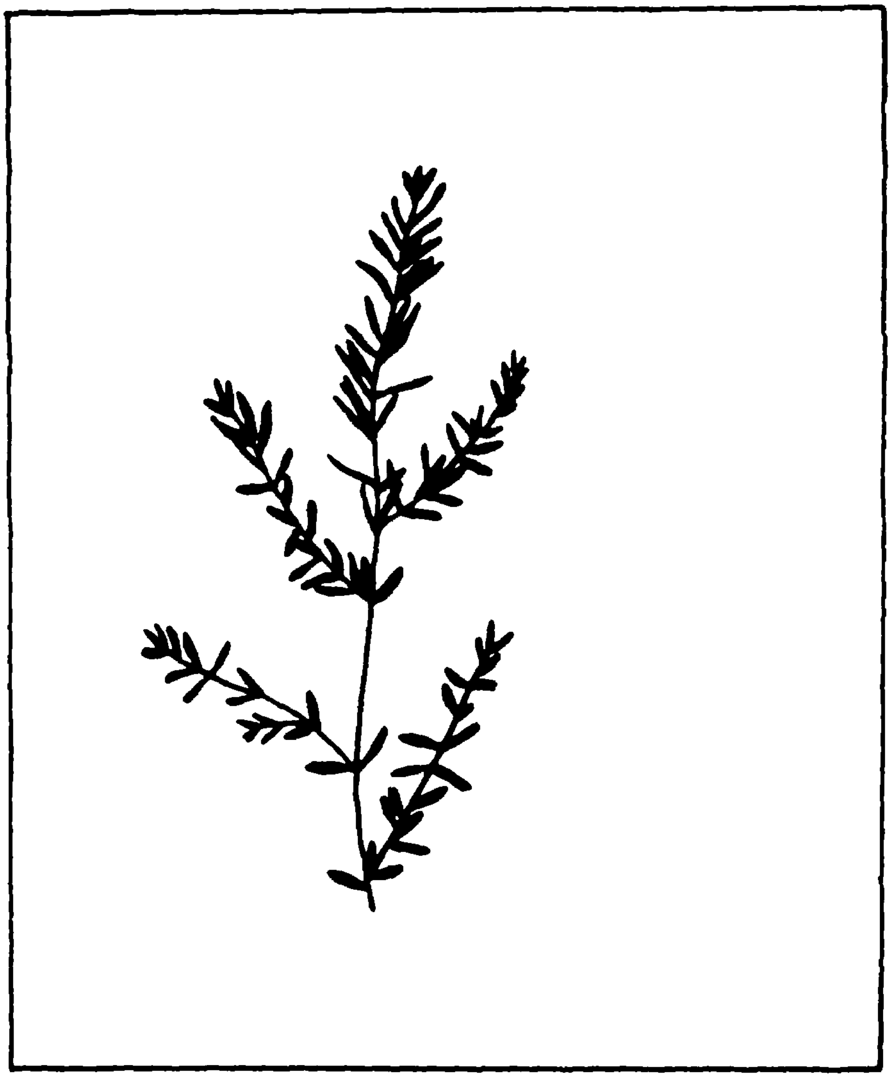


(copied from Hotchkiss 1967)

Figure 4. Eurasian watermilfoil (Myriophyllum spicatum)

Najas guadalupenses (Bushy pondweed)

	References
Distribution	
Essentially freshwater or brackish water species, ranging from Oregon to Quebec, and California to Florida.	Hotchkiss 1967 Martin and Uhler 1939
Temperature	
No information	
Salinity	
Prefers 3°/oo salinity. Found in Potomac River at salinities of 6 to 9°/oo.	Steenis 1970
Substrate	
Prefers soils containing a predominance of sand, but tolerates substrate of pure muck.	USDI 1944 Martin and Uhler 1939
Light, Depth and Turbidity	
Usually found in depths ranging from 0.3 to 1.2 m, but has been recorded at depths over 6 m.	Martin and Uhler 1939
Consumer Utilization	
Excellent in food value for waterfowl. Birds eat both the seeds and the leafy plant parts.	Martin and Uhler 1939



(redrawn after Hotchkiss 1967)

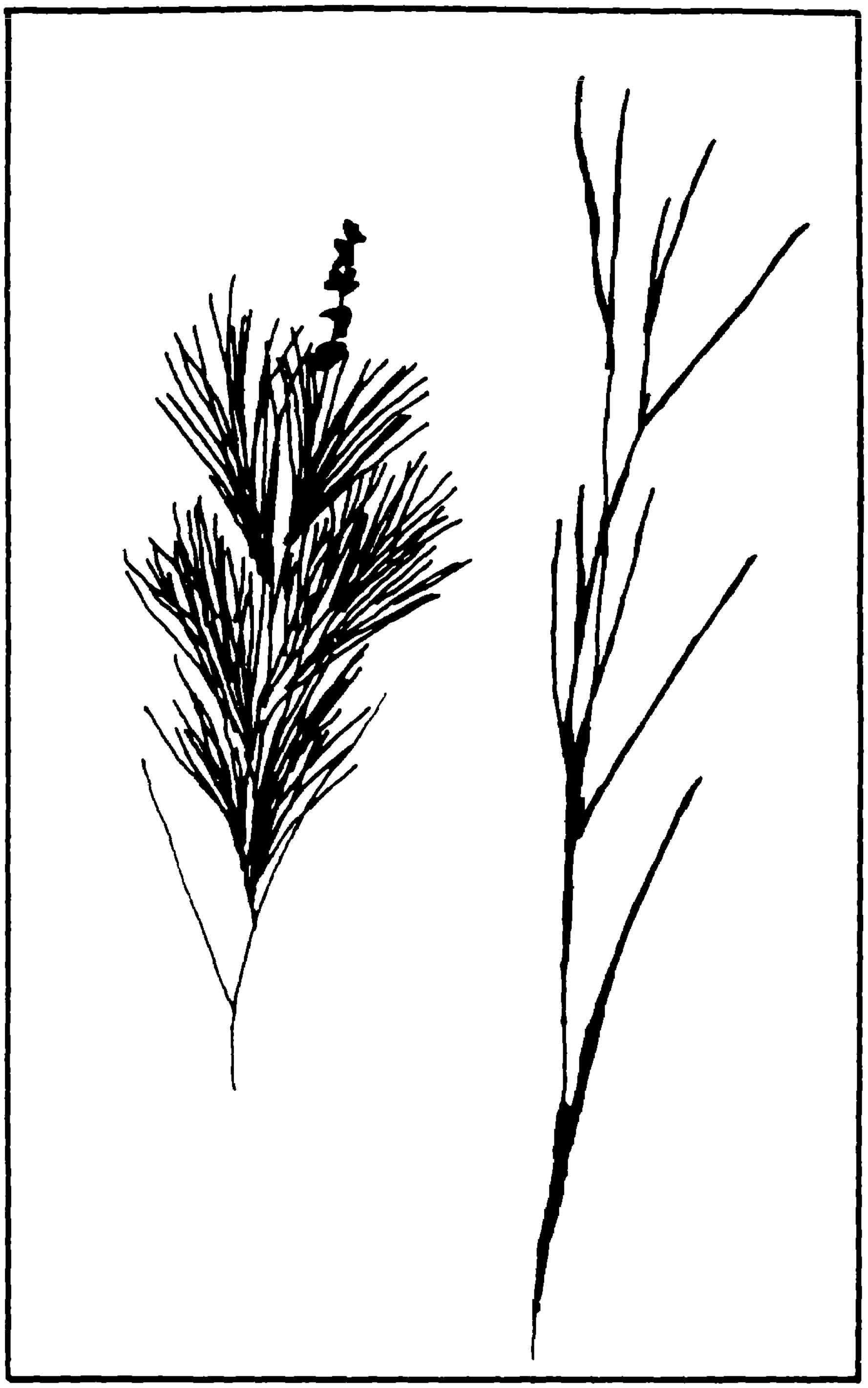
Figure 5. Naiad (Najas sp.)

Potamogeton pectinatus (Sago pondweed)

	References
Distribution	
Range includes freshwater streams and ponds, also brackish coastal waters of the United States and portions of Canada. Most abundant in the northwestern states and the Chesapeake Bay in the United States. Reported to be a pest species of irrigation systems in the west, and in cranberry bogs of Massachusetts.	Martin and Uhler 1939 Hodgeson and Otto 1963 Devlin 1973
Temperature	
Germination shown to occur when water temperature reaches 15 to 18°C.	Yeo 1965b
Salinity	
Maximum seed production, seed germination and vegetative growth occurs in freshwater. Salinities of 8 to 9‰ generally decreased growth and germination rates by 50 percent.	Teeter 1965
Substrate	
Grows on both mud and sand bottoms. Prefers silty bottoms.	Sculthorpe 1967 Rickett 1923
Light, Depth and Turbidity	
Requires at least 3.5 percent total sunlight for growth. Shading produces yellowed, sparse foliage, elongated nodes and rigid unbranched stems.	Bourn 1932

Potamogeton pectinatus (Sago pondweed)
Continued

	References
Consumer Utilization	
One of the more important waterfowl plant foods. Nutlets and tubers reported to be excellent food source for ducks; rootstocks and stems are consumed to a lesser degree. Also provides protective habitat for fish, oysters, and benthic creatures.	Martin and Uhler 1939 Fassett 1960



(copied from Hotchkiss 1967)

Figure 6. Sago pondweed (Potamogeton pectinatus)

Potamogeton perfoliatus (Redhead grass)

	References
Distribution	
Fresh and moderately brackish waters. It has been found in Labrador, Quebec, New Brunswick and extends to Eurasia, northern Africa and Australia. Its presence has been recorded in the Chesapeake Bay through 1976.	Ogden 1943 USFWS Migratory Bird and Habitat Research Laboratory 1976
Temperature	
Experiments showed that respiration and O_2 consumption increased as temperatures increased from 25 to 40°C, with death occurring at 45°C.	Anderson 1969
Salinity	
1.5 to 19°/₀₀, tolerant to 25°/₀₀.	Anderson 1969
Substrate	
Grows best on a mixture of organic material and silt with a minimum carbon to nitrogen ratio, a high capacity to recycle ammonia and a low redox potential. Moderately organic muds fairly rich in nitrogen and exchangeable calcium are more suitable than highly organic muds.	Misra 1938

Potamogeton perfoliatus (Redhead grass)

Continued

	References
Light, Depth and Turbidity	
Usually found in still or standing water ranging from 0.6 to 1.5 m depth. Maximum rate of photosynthesis attained where light intensity was about 1.1 g cal/cm^2.	Felfoldy 1960 Martin and Uhler 1939
Consumer Utilization	
Seeds, rootstocks and portions of the stem are consumed by Black Ducks, Canvasbacks, Redheads, Ringnecks and other duck species. Also eaten by geese, swans, beaver, deer, muskrat. Provides protective cover for various aquatic organisms.	Martin and Uhler 1939 Fassett 1960

(copied from Hotchkiss 1967)

Figure 7. Redhead grass (Potamogeton perfoliatus)

Ruppia maritima (Widgeongrass)

References

Distribution

Inhabits a wide range of shallow, brackish pools, rivers and estuaries along the Atlantic, Gulf and Pacific Coasts. Also occurs in fresh portions of estuaries, alkaline lakes, ponds and streams and in shallow, saline ponds and river deltas of the Great Salt Lake region.

Martin et al. 1951
Radford et al. 1964
Ungar 1974
Chrysler et al. 1910

Temperature

R. maritima appeared to have two growing seasons within the temperature range of 18° to 30°C. Growth ceased outside this range although some fruiting and flowering occurred at temperatures higher than 30°C.

Joanen and Glasgow 1965

Salinity

Tolerant of a broad salinity range, from 5.0 to 40.0°/₀₀. Tension zone of over 30°/₀₀. Flowering and seed set occurs in range of tapwater to 28°/₀₀.

Steenis 1970
Anderson 1972
McMillan 1974

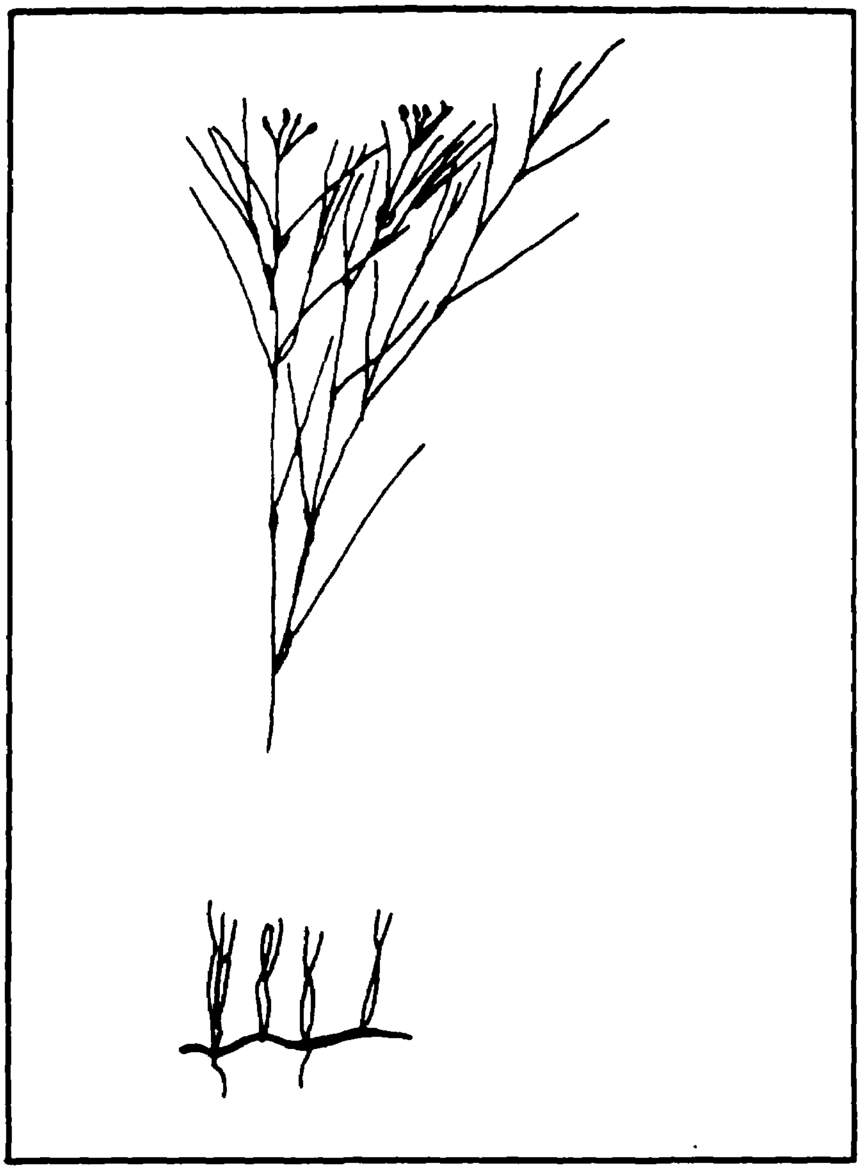
Substrate

Prefers soft bottom muds or sand. Has been found growing on shallow sand shell gravel soils in Russian rivers and streams.

Anderson 1972
Zenkevitch 1963

Ruppia maritima (Widgeongrass)
Continued

	References
Light, Depth and Turbidity	
Optimum production in laboratory studies occurred at depth of 60 cm. Is found at depths of a few inches to several feet. Turbidity tolerance less than 25-35 ppm in small ponds; turbidity is especially harmful to young plants prior to the stems reaching the surface.	Joanen and Glasgow 1965
Consumer Utilization	
Serves as food for numerous species of ducks, coots, geese, grebes, swans, marsh and shore birds of the Atlantic, Pacific and Gulf Coasts. Also used as nursery grounds and as a fish spawning medium and cover for marine organisms.	Sculthorpe 1967 Martin and Uhler 1939 Kerwin 1975b



(copied from Hotchkiss 1976)

Figure 8. Widgeongrass (Ruppia maritima)

Vallisneria americana (Wildcelery)

References

Distribution

Freshwater macrophyte occurring in the tidal streams of the Atlantic Coastal Plain. — Martin and Uhler 1939

Temperature

Grows best in temperature range of 33 to 36°C. Arrested growth occurs below 19°C. — Wilkinson 1963

Salinity

Laboratory tests showed that *Vallisneria* could not be maintained in salinities greater than $4.2^{\circ}/_{\circ\circ}$. — Bourn 1934

Substrate

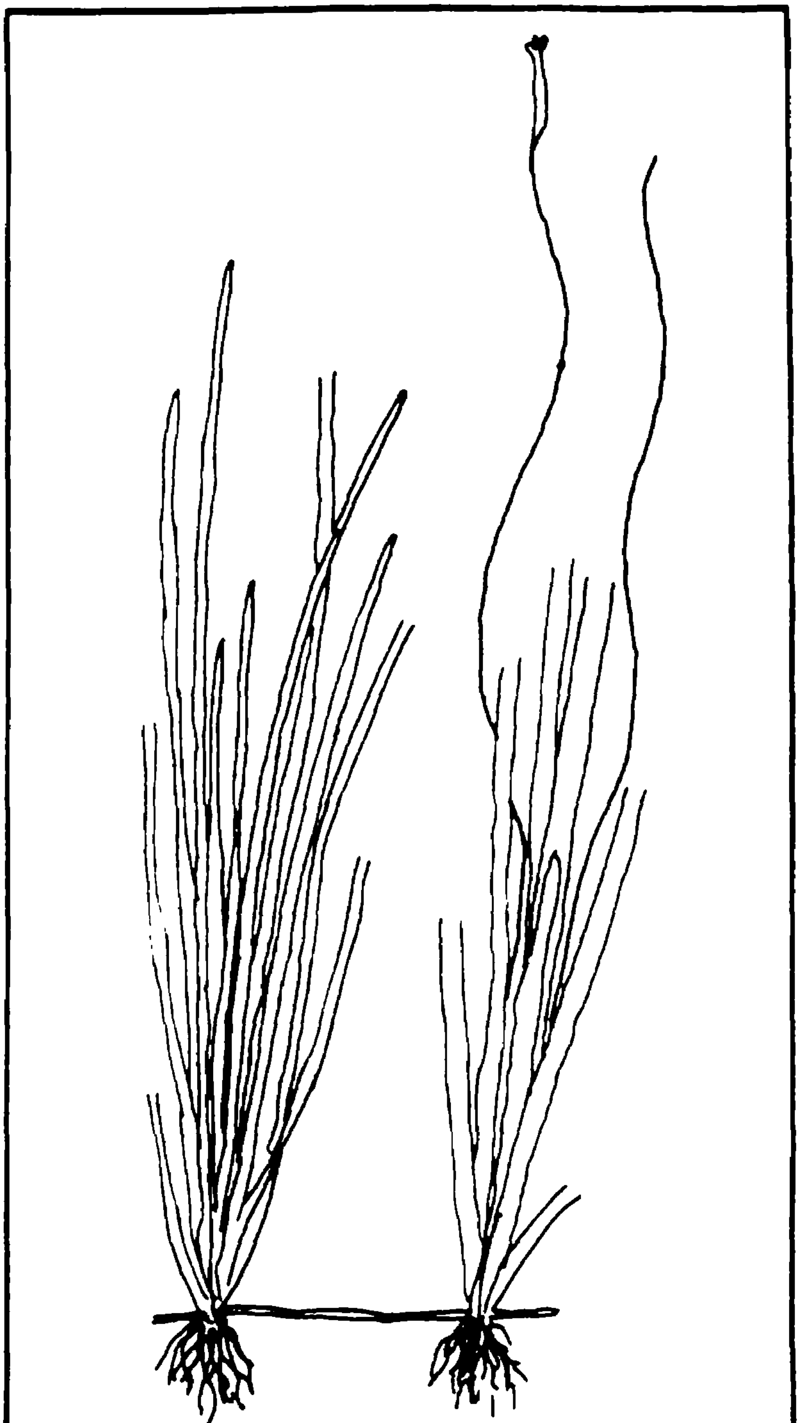
Grows equally well in sandy soil and mud. Hutchinson (1975) found that *V. americana* thrived best in a soil of 6.5 percent organics, 8.78 percent gravel, 21.46 percent sand, 47.90 percent silt, 14.26 percent clay. — Schuette and Alder 1927; Hutchinson 1975

Light, Depth and Turbidity

Able to tolerate muddy, roiled water. Usually found in shallow water (0.5 to 1.0 m). — Steenis 1970

Vallisneria americana (Wildcelery)
Continued

Consumer Utilization	References
All parts of the plant structure are consumed by fish, ducks, coots, geese, grebes, swans, waders, shore and game birds. Also serves as a shade, shelter and spawning medium for fish.	Sculthorpe 1967



(copied from Hotchkiss 1967)

Figure 9. Wildcelery (<u>Vallisneria americana</u>)

Zannichellia palustris (Horned pondweed)

References

Distribution

This species has been documented in every state in continental United States; however, it is not a commonly occurring submerged aquatic. Reported occasionally in brackish marshes along the New England coast, rarely found inland. Recorded in Chesapeake Bay and south to Currituck and Pamlico Sound area, North Carolina.	Deane 1910 Fassett 1960

Temperature

In the Chesapeake Bay, the Zannichellia populations decline rapidly when temperatures reach 30°C. Reported to exist in temperatures as low as 10.5 to 14.8°C.	Tutin 1940

Salinity

Tolerates freshwater, but prefers brackish waters to 20°/₀₀.	Radford et al. 1964

Substrate

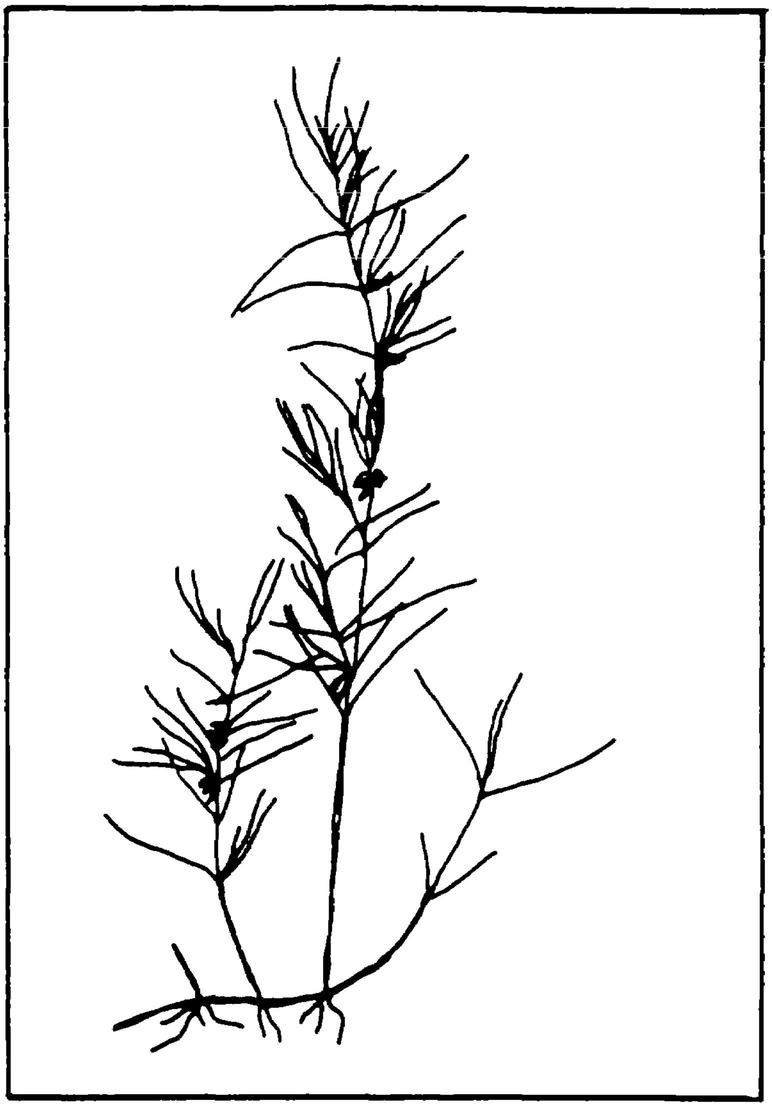
Tends to grow in clay to sandy sediments.

Light, Depth and Turbidity

Prefers shallower water than other submerged aquatics. May need higher light intensities than others; good growth obtained at 4 to 7 percent of the maximum noon summer sunlight.	Correll et al. 1977

Zannichellia palustris (Horned pondweed)
Continued

	References
Consumer Utilization	
Fruits and sometimes foliage are good for waterfowl in brackish pools.	Fassett 1960



(copied from Hotchkiss 1967)

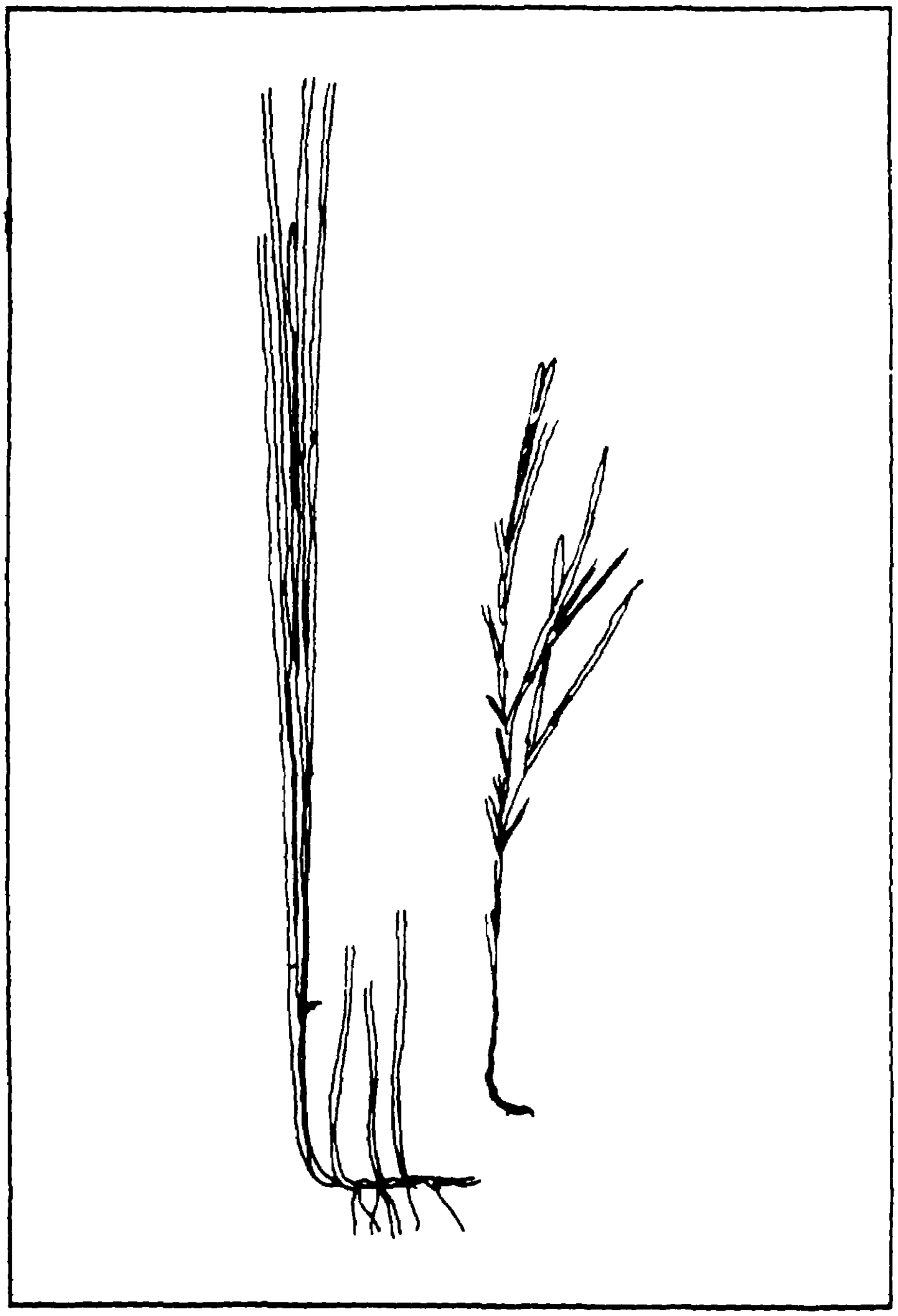
Figure 10. Horned pondweed (Zannichellia palustris)

<u>*Zostera marina*</u> (Eelgrass)

	References
Distribution	
On the Pacific Coast of North America, eelgrass extends from Grantly Harbor, Alaska, to Agiahampo Lagoon in the Gulf of California. On the Atlantic Coast of North America, eelgrass extends from Hudson Bay, Canada, the southern tip of Greenland, and one locality in Iceland, to Bogue Sound, North Carolina.	McRoy 1968 Steinbeck and Picketts 1941 Cottam 1934b Ostenfeld 1918 Phillips 1974a
Temperature	
Tolerate temperatures from -6°C to 35°C. Photosynthesis decreased sharply above 35°C. Death occurred after exposure to -9°C.	Biebel and McRoy 1971
Salinity	
Can tolerate salinities ranging from 8°/₀₀ to full strength seawater (35°/₀₀).	Phillips 1974a Arasaki 1950a, 1950b Martin and Uhler 1939
Substrate	
Found growing on a wide variety of substrates, from pure firm sand to pure firm mud.	Phillips 1974a

Zostera marina (Eelgrass)
Continued

	References
Light, Depth and Turbidity	
Has been found growing from about 2 m above MLW (minimum low water) to depths down to 30 m. Low light intensity conditions inhibit flowering and turion (young branch) density is decreased in shaded plots.	Cottam and Munro 1954 Phillips 1974a Backman and Barilotti 1976
Consumer Utilization	
The only groups of animals that consume eelgrass directly are waterfowl and sea turtles. Eelgrass beds provide important habitats and nursery areas for many forms of invertebrates and vertebrates, which then serve as food sources of species at higher levels.	Cottam 1934b Addy and Aylward 1944 Gutsell 1930



(copied from Hotchkiss 1967)

Figure 11. Eelgrass (*Zostera marina*)

APPENDIX D

Environmental Requirements of certain fish in Gulf of Mexico estuaries

Contents

from Benson 1982

Anchoa hepsetus (striped anchovy)

The distribution of all life stages of striped anchovy appears to be limited primarily by salinity. Christmas and Waller (1973) reported this species in salinities ranging from 5.0 ppt to 3.5 ppt. Perry and Boyes (1978) collected 95.6% of their specimens in salinities between 20 and 30 ppt, largely in waters south of the Gulf Intracoastal Waterway. This fish is most abundant at temperatures ranging from 20° to 30°C (68° to 86°F) (Perry and Boyes 1978).

Anchoa mitchilli (bay anchovy)

Although the distribution of the bay anchovy in Mississippi Sound waters is not greatly affected by differences in salinities, low winter temperatures appear to cause some movement to deeper, warmer offshore waters (Springer and Woodburn 1960; Christmas and Waller 1973). Swingle (1971) found them to be nearly equally distributed in salinities between 5 and 19 ppt in Alabama coastal waters. Highest catches were in salinities ranging from 20.0 to 29.9 ppt. In Mississippi Sound, Christmas and Waller (1973) established no relationships between the distribution of anchovies and salinities above 2 ppt. Perry and Christmas (1973) found larvae in Mississippi waters in salinities ranging from 16.6 to 27.8 ppt. Bay anchovies were taken at temperatures from 5.0° to 34.9°C (41.0° to 94.8°F), but the largest numbers were in water temperatures between 10.0° and 14.9°C (50.0° and 58.8°F) (Christmas and Waller 1973).

Arius felis (sea catfish)

Sea catfish in estuaries in the summer are most abundant in water temperatures from 19° to 25°C (66° to 77°F). Year round, they have been taken in the range of 5.0° to 34.9°C (41.0° to 94.8°F) (Perret et al. 1971; Adkins and Bowman 1976; Drummond and Pellegrin 1977; Johnson 1978). This euryhaline species is common in salinities from 0 to 45 ppt, but some tolerate 60 ppt. A preference of higher salinities has been suggested (Gunter 1947; Johnson 1978; Lee et al. 1980). Breeding occurs in waters having a salinity range of 13 to 30 ppt.

The developmental stage of larvae incubating in the oral cavity may determine the location of the parent male (Harvey 1971). Younger larvae tolerate salinities up to 12.8 ppt, but more developed larvae tolerate salinities of 16.7 to 28.3 ppt (Harvey 1971). Juveniles are most numerous in low salinities (Johnson 1978).

Although minimum dissolved oxygen requirements of sea catfish are not known, this fish sometimes lives in dredged semiclosed and closed canals that are characterized by low oxygen concentrations (Adkins and Bowman 1976). They are found in moderately turbid water (Gunter 1947; Lee et al. 1980).

Sea catfish principally live at depths from 4 to 7 m (13 to 23 ft), but may occupy waters as deep as 36 m (118 ft) (Lee 1937; Johnson 1978). Major substrates are muddy or sandy bottoms rich in nutrients (Etchevers 1978; Shipp 1981).

Paralichthys lethostigma (southern flounder)

The southern flounder is euryhaline, occurring in waters with salinities from 0 to 60 ppt. The normal range is from about 10 to 31 ppt. They live at water temperatures from 9.9° to 30.5°C (49.8° to 86.9°F), but are most common between 14.5° and 21.6°C (58.1° and 70.9°F) (Stokes 1973). The temperatures and salinities where southern flounder were collected in Mississippi Sound by Christmas and Waller (1973) ranged from 5.0° to 34.9°C (41.0° to 94.8°F) and 0.0 to 29.9 ppt. The juveniles may live in freshwater for short periods.

Juveniles are usually most abundant in shallow areas with aquatic vegetation (shoal grass and other sea grasses) on a muddy bottom. Adults also tend to favor aquatic vegetation such as *Spartina alterniflora*. Some flounders overwinter in the deeper holes and channels of estuaries, but most (adults and second-year juveniles) migrate to Gulf waters in the fall (Gunter 1945).

Mugil cephalus (striped mullet)

Striped mullet live in freshwater and in salinities up to 75 ppt. In Texas estuaries the mullet were about equally distributed in water of all salinities (Gunter 1945). They have been taken in Mississippi in salinities ranging from 0.0 to 35.5 ppt (Christmas and Waller 1973).

Fish less than 3.6 cm (1.4 inches) long are most abundant in salinities from 0.0 to 14.9 ppt. Juveniles (up to 7.9 cm or 3.1 inches long) prefer lower salinities and warmer waters than larger fish. Juveniles are mostly taken in salinities from 0 to 10 ppt when temperatures range from 25° to 30°C (77° to 86°F). Fish up to 11 cm (4 inches) long are abundant at salinities from 0 to 20 ppt at temperatures of 7° to 30°C (45° to 86°F) (Etzold and Christmas 1979). Highest catches in samples from Mississippi Sound were in the range of 7° to 20°C (45° to 68°F). Mullet are often killed in water temperatures less than 5°C (41°F) (J.C. Parker 1971), and they tend to aggregate in sheltered areas before the arrival of cold weather.

Pomatomus saltatrix (bluefish)

Temperature and salinity are the only factors cited by Wilk (1977) as determinants of the distribution of bluefish on the Atlantic coast. Extensive data from egg and larval collections on the outer continental shelf of Virginia showed that maximum spawning occurred at 25.6°C (78.1°F) with none below 18°C (64°F) (Norcross et al. 1974). Minimum spawning temperature is about 14°C (57°F) (Hardy 1978). Bluefish seem to prefer salinities from 26.6 to 34.9 ppt. Limited larvae collections in the Gulf of Mexico were found in a temperature range of 23.2° to 26.4°C (73.8° to 79.6°F) and a surface salinity range of 35.7 to 36.6 ppt (Barger et al. 1978). In estuaries they rarely live in salinities below 10 ppt. Hardy (1978) suggested 7 ppt as the minimum salinity. Lacking are data on the effects of substrate, turbidity, tides, or dissolved oxygen on bluefish distribution. Bluefish activity patterns are highly oriented to vision (Olla and Studholme 1979), however, and bluefish are not likely to frequent turbid areas.

Pogonias cromis (black drum)

Black drum are euryhaline during all life stages, i.e., they occur in salinities from 0 to 35 ppt. The species is most common at salinities ranging from 9 to 26 ppt (Gunter 1956; Etzold and Christmas 1979), but some inhabit water with salinities as high as 80 ppt. The black drum is usually taken at water temperatures from 12° to 30°C (54° to 86°F). This fish inhabits areas with sand or soft bottoms as well as brackish marshes and oyster reefs (Etzold and Christmas 1979). The preferred habitat of juveniles during the first 3 months are muddy, nutrient-rich, marsh habitats such as tidal creeks.

Sciaenops ocellatus (red drum)

The general salinity range for red drum is 0 to 30 ppt, but some tolerate salinities up to 50 ppt (Theiling and Loyacano 1976). Larvae and juveniles were taken at salinities between 5.0 and 35.5 ppt in one study (Christmas and Waller 1973), but most occur at salinities from 9 to 26 ppt. The larger fish seem to prefer higher salinities. Red drum are most abundant in salinities from 20 to 25 ppt (Etzold and Christmas 1979), and from 25 to 30 ppt (Kilby 1955). Overall, red drum prefer moderate to high salinities.

Red drum have been observed in water temperatures ranging from 2° to 29°C (36° to 84°F). Some young fish were found in a temperature range of 20.5° to 31°C (68.9° to 87.8°F). The highest catches were at temperatures between 20° and 25°C (68° and 77°F) (Etzold and Christmas 1979). Large numbers of red drum have been reported killed in severe cold spells (Adkins et al. 1979).

Red drum thrive in waters over sand, mud, or sandy mud bottoms and occasionally in and among aquatic vegetation.

United States
Environmental Protection
Agency

Office of Water
Regulations and Standards
Washington, DC 20460

Water

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EPA

Technical Support Manual: Waterbody Surveys and Assessments for Conducting Use Attainability Analyses

Volume III: Lake Systems

TECHNICAL SUPPORT MANUAL:
WATERBODY SURVEYS AND ASSESSMENTS FOR CONDUCTING USE ATTAINABILITY ANALYSES

VOLUME III: LAKE SYSTEMS

U.S. ENVIRONMENTAL PROTECTION AGENCY
OFFICE OF WATER REGULATIONS AND STANDARDS
CRITERIA AND STANDARDS DIVISION
WASHINGTON, D.C. 20460

NOVEMBER 1984

FOREWORD

The Technical Support Manual: Water Body Surveys and Assessments for Conducting Use Attainability Analyses, Volume III: Lake Systems contains guidance prepared by EPA to assist States in implementing the revised Water Quality Standards Regulation (48 FR 51400, November 8, 1983). This document addresses the unique characteristics of lake systems and supplements the two previous Manuals for conducting use attainability analyses (U.S. EPA, 1983b, 1984). The purpose of these documents is to provide guidance to assist States in answering three central questions:

(1) What are the aquatic protection uses currently being achieved in the water body?

(2) What are the potential uses that can be attained based on the physical, chemical and biological characteristics of the water body?

(3) What are the causes of any impairment of the uses?

Consideration of the suitability of a water body for attaining a given use is an integral part of the water quality standards review and revision process. EPA will continue to provide guidance and technical assistance to the States in order to improve the scientific and technical bases of water quality decisions. States are encouraged to consult with EPA at the beginning of any standards revision project to agree on appropriate methods before the analyses are initiated, and to consult frequently as they are conducted.

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

Elliot Lomnitz
Criteria and Standards Division (WH-585)
401 M Street, S.W.
Washington, D.C. 20460

Edwin L. Johnson

Edwin L. Johnson, Director
Water Regulations and Standards

CONTENTS

CHAPTER I

INTRODUCTION

EPA's Office of Water Regulations and Standards has prepared guidance to accompany changes to the Water Quality Standards Regulation (48 FR 51400). This guidance has been compiled and published in the Water Quality Standards Handbook (U.S. EPA, December 1983a). Sections in the Handbook present discussion of the water quality review and revision process; general guidance on mixing zones, and economic considerations pertinent to a change in the use designation of a water body; the development of site specific criteria; and the elements of a use attainability analysis.

One of the major pieces of guidance in the Handbook is "Water Body Surveys and Assessments for Conducting Use Attainability Analyses." This guidance presents a general framework for designing and conducting a water body survey whose objective is to answer the following questions:

1. What are the aquatic life uses currently being achieved in the water body?

2. What are the potential uses that can be obtained, based on the physical, chemical and biological characteristics of the water body?

3. What are the causes of impairment of the uses?

In response to requests from several states for additional information, technical guidance on conducting water body surveys and assessments has been provided in two documents:

1. Technical Support Manual: Water Body Surveys and Assessments for Conducting Use Attainability Analyses (U.S. EPA, November 1983b);

2. Technical Support Manual: Water Body Surveys and Assessments for Conducting Use Attainability Analyses, Volume II: Estuarine Systems (U.S. EPA, June 1984).

The first volume is oriented towards rivers and streams and presents methods for freshwater evaluations. The second volume stresses those considerations which are unique to the estuary. The current Manual, Volume III, focuses on the physical, chemical and biological phenomena of lakes and is presented so as not to repeat information that is common to other freshwater systems that already appears in one of the earlier volumes. Apart from the rare impoundment that is fed only by surface runoff or underground springs, rivers and lakes are linked physically and exhibit a transition from riverine habitat and conditions to lacustrine habitat and conditions. Because of this physical link, the biota of the lake will be essentially the same as the biota of the stream, although there are few species that are primarily lake species. Given the ties that exist between lake and stream under natural conditions, it is important that those who will be conducting lake use attainability studies refer to Volume I on rivers and streams for additional perspective.

Each of the Technical Support Manuals provides extensive information on the plants and animals characteristic of a given type of water body, and provides a number of assessment techniques that will be helpful in performing a water body survey. The methods offered in the guidance documents are optional, however, and states may apply them selectively, or may use their own techniques for designing and conducting use attainability studies.

Consideration of the suitability of a water body for attaining a given use is an integral part of the water quality standards review and revision process. The data and other information assembled during the water body survey provide a basis for evaluating whether or not the water body is suitable for a particular use. Since the complexity of an aquatic ecosystem does not lend itself to simple evaluations, there is no single formula or model that will serve to define attainable uses. Rather, many evaluations must be performed, and the professional judgment of the evaluator is crucial to the interpretation of data that is reviewed.

This Technical Support Manual on lakes will not tell the biologist or engineer how to conduct a use attainability study, per se, rather, it will lay out those chemical, physical and biological phenomena that are characteristic of lakes, and point out factors that the investigator might take into consideration while designing a use study, and while preparing an assessment of uses from the information that has been assembled. The chapters in this Manual focus on the following aspects of lakes:

Chapter II. Physical and Chemical Characteristics

- Circulation, stratification, seasonal turnover
- Nutrient cycling
- Eutrophication processes
- Computer and desktop procedures for lake evaluations

Chapter III. Biological Characteristics

- Benthos
- Zooplankton
- Phytoplankton
- Macrophytes
- Fish

Chapter IV. Synthesis and Interpretation

- Aquatic life use classifications
- Impairment of uses
- Reference site comparisons
- Preventive and remedial techniques

Chapter V. References

CHAPTER II

PHYSICAL AND CHEMICAL CHARACTERISTICS

INTRODUCTION

The aquatic life uses of a lake are defined in reference to the plant and animal life in the lake. The types and abundance of the biota are largely determined by the physical and chemical characteristics of the lake. Other contributing factors include location, climatological conditions, and historical events affecting the lake.

Each lake characteristic such as depth, length, inflow rate and temperature contributes to the physical processes of the water body. For example, circulation may be the dominant physical process in a lake that is large and shallow while for a deep medium size lake the dominant process may be the annual cycle of thermal stratification.

The chemical characteristics of a lake are affected by inflow water quality and by various physical, chemical and biological processes which provide the biota with its sustaining nutrients and required dissolved oxygen. Overenrichment with nutrients may accelerate the natural processes of the lake, however, and lead to major upsets in plant growth patterns, dissolved oxygen profiles, and plant and animal communities. The physical and chemical attributes of lakes as well as the influence of physical processes on chemical characteristics are discussed in this chapter.

In addition to a discussion of physical parameters and processes, and the chemical characteristics of lakes, several techniques for use attainability evaluations are presented in this chapter. These include empirical input/output models, computer simulation models, and data evaluation techniques. For each of these general categories specific methods and models are presented with references. Illustrations of some techniques are also presented.

The objective in discussing the physical and chemical properties of lakes is to assist the states to characterize a lake and select assessment methodologies that will enable the definition of attainable uses.

PHYSICAL CHARACTERISTICS

Physical Parameters

The physical parameters which describe the size, shape and flow regime of a lake represent the basic characteristics which affect physical, chemical and biological processes. As part of a use attainability analysis, the physical parameters must be examined in order to understand non-water quality factors which affect the lake's aquatic life.

Lakes can be grouped according to formation process. Ten major formation processes presented by Wetzel (1975) include:

- o Tectonic (depression due to earth movement)
- o Volcanos
- o Landslides
- o Glaciers
- o Solution (depressions from soluble rock)
- o River activity
- o Wind-formed basins
- o Shoreline activity
- o Dams (man-made or natural).

The origins of a lake determine its morphologic characteristics and strongly influence the physical, chemical and biological conditions that will prevail.

Physical (morphological) characteristics whose measurement may be of importance to a water body survey include the following:

- o Surface area, A (measured in units of length squared, L^2)
- o Volume, V (measured in units of length cubed, L^3)
- o Inflow and outflow, Q_{in} and Q_{out} (measured in units of length cubed per time, L^3/T)
- o Mean depth, $\bar{d}$
- o Maximum depth
- o Length
- o Length of shoreline
- o Depth-area relationships
- o Depth-volume relationships
- o Bathymetry (submerged contours).

Some of these parameters may be used to calculate other characteristics of the lake. For example:

- The mass flow rate of a chemical, say phosphorus, may be calculated as the product of concentration $[P_{in}]$ and inflow, Q_{in}, provided the units are compatible.

  $$\text{mass flow rate} = [P_{in}, M/L^3] \times (Q_{in}, L^3/T) = M/T$$

  where M denotes units of mass

- The surface loading rate is calculated as the quotient of inflow and surface area, or the quotient of mass flow rate and area, e.g.,

  $$\text{liquid surface loading rate} = (Q_{in}, L^3/T)/(A, L^2) = L^3/L^2\text{-}T$$

  $$\text{mass surface loading rate} = [C_{in}, M/L^3] \times (Q_{in}, L^3/T)/(A, L^2) = M/L^2\text{-}T$$

- The detention time is given by the quotient of volume and flow rate, e.g.,

  $$\text{detention time} = (V, L^3)/(Q_{in}, L^3/T) = T$$

  The reciprocal of the detention time is the flushing rate, T^{-1}

- Mean depth is the quotient of volume and surface area, e.g.,

  $$\bar{d} = (V, L^3)/(A, L^2) = L$$

The first seven parameters of the above list describe the general size and shape of the lake. Mean depth has been used as an indicator of productivity (Wetzel, 1975; Cole, 1979) since shallower lakes tend to be more productive. In contrast, deep and steep sided lakes tend to be less productive.

Total lake volume and inflow and outflow rates are physical characteristics which indirectly affect the lake aquatic community. Large inflows and outflows for lakes with small volumes produce low detention times or high flow through rates. Aquatic life under these conditions may be different than when relatively small inflows and outflows occur for a large lake volume. In the latter case the detention time is much greater.

Hand (1975) has recommended a shape factor--the lake length divided by the lake width--for lake studies. This shape factor was applied by Hand and McClelland (1979) as a variable in a regression equation used to predict chlorophyll-a in Florida lakes. Other parameters in that regression equation are phosphorus, nitrogen, and the mean depth.

For the requirements of a more detailed lake analysis, information describing the depth-area and depth-volume relationships and information describing the bathymetry may be required. An example of a bathymetric map is shown in Figure II-1 for Lake Harney, Florida (Brezonik and Fox, 1976). The roundness of this particular lake is typical of many lakes in Florida whose morphometry has been affected by limestone solution processes (Baker, et al., 1981). A typical representation of the depth-area and depth-volume relationships for a lake is shown in the graph of Figure II-2 for the Fort

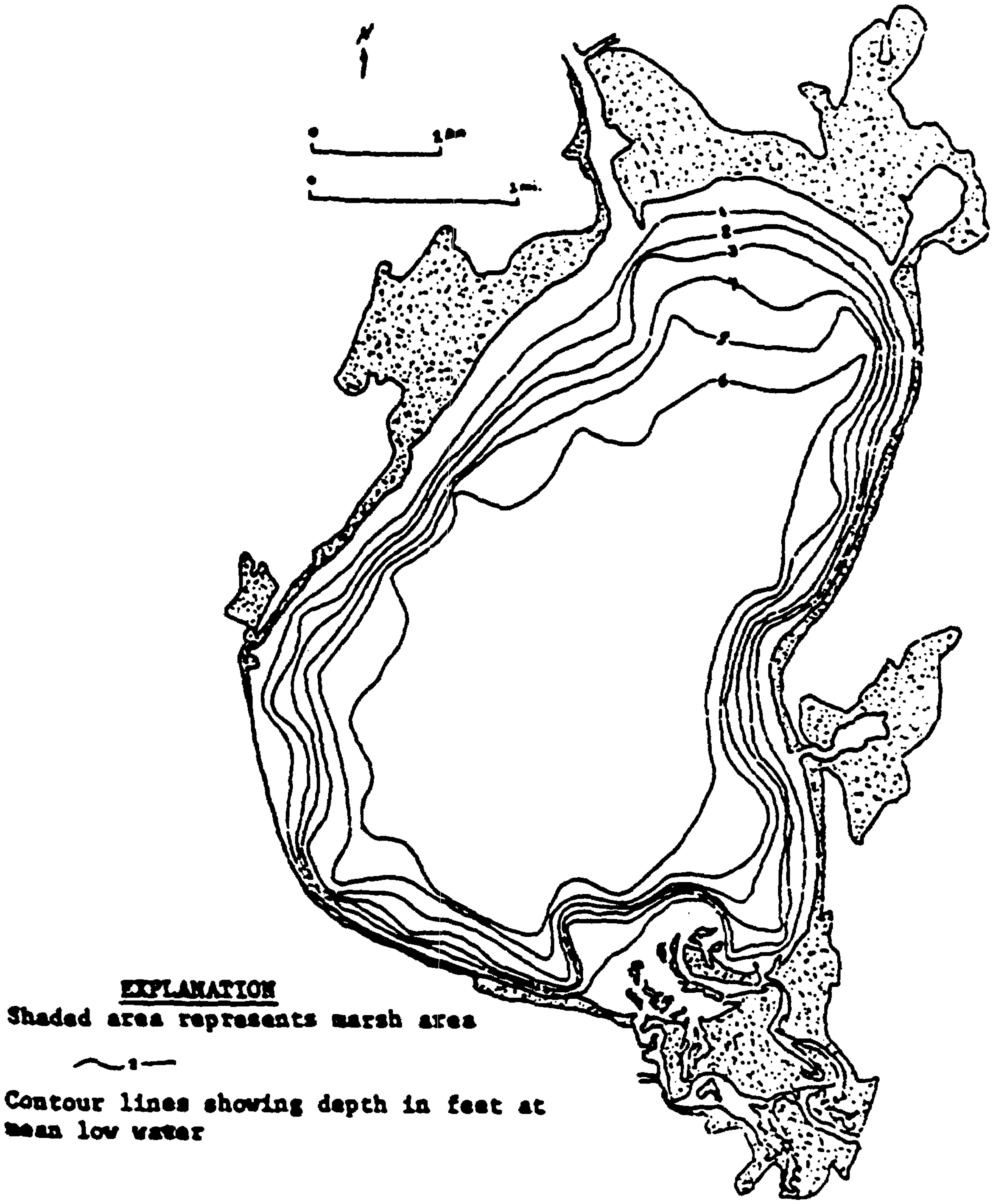

Figure II-1. Bathymetric Map of Lake Harney, Florida (from Brezonik, 1976)

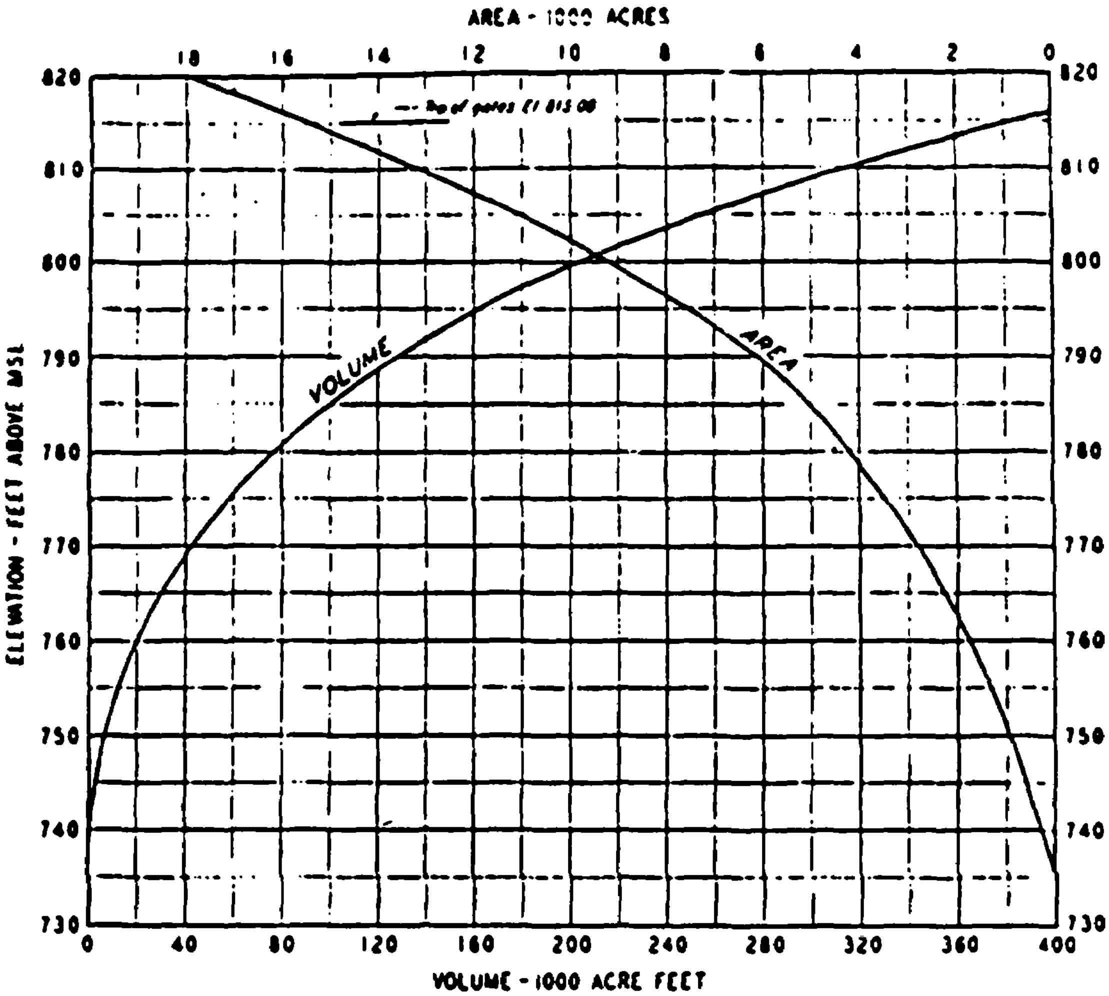

Figure II-2. Fort Loudoun Reservoir Areas and Volumes (from Water Resources Engineers, 1975)

Loudoun Reservoir, Tennessee (Hall, et al., 1976). Depth-area relationships can be important to the biological activity in a lake. If the relationship is such that with a slight increase in depth the surface area is greatly increased, this then produces greater bottom and sediment contact with the water volume which in turn could support increased biological activity.

In addition to the physical parameters listed above, it is also important to obtain and analyze information concerning the lake's contributing watershed. Two major parameters of concern are the drainage area of the contributing watershed, and the land use(s) of that watershed. Drainage area will aid in the analysis of inflow volumes to the lake due to surface runoff. The land use classification of the area around the lake can be used to predict flows and also nonpoint source pollutant loadings to the lake.

The physical parameters presented above may be used to understand and analyze the various physical processes that occur in lakes. They can also be used directly in simplistic relationships which predict productivity to aid in aquatic use attainability analyses.

Physical Processes

There are many complex and interrelated physical processes which occur in lakes. These processes are highly dependent on the lake's physical parameters, geographical location and characteristics of the contributing watershed. Individual physical processes are usually highly interdependent. Five major processes--lake currents, heat budget, light penetration, stratification and sedimentation--are discussed below. Each process can affect the ecological system of a lake, especially the biota and the distribution of chemical species.

Lake Currents

Water movement in a lake affects productivity and the biota because it influences the distribution of nutrients, microorganisms and plankton (Wetzel, 1975). Lake currents are propagated by wind, inflow/outflow and Coriolis force (a deflecting force which is a function of the earth's rotation). The types of currents developed in lakes are dependent upon the lake size and its density structure.

For small, shallow lakes (especially those that are long and narrow), inflow/outflow characteristics are most important and the predominant current is a steady-state flow through the lake. For very large lakes, wind is the primary generator of currents and, except for local effects, inflow and outflow have a relatively minor affect on lake circulation. The Coriolis force is another important determinant of circulation in larger lakes such as the Great Lakes (Lick, 1976a).

Wind. Wind induced turbulence on the lake surface results in a variety of current patterns that are characteristic of the lake's physical properties. For shallow lakes, the wind induces vertical mixing throughout the water column. Steady-state currents formed in deep lakes that have a constant density are characterized by top and bottom boundary layers where vertical mixing is important, and by horizontal boundary layers near the shore where horizontal mixing is important (Lick, 1976a).

Under severe or prolonged wind conditions, the stress on the water surface can cause circulation in the upper epilimnion region of a stratified lake because of the inclination of the water surface. This then can cause a counter flow in the lower hypolimnion region of the reservoir. This condition is demonstrated by Fischer (1979) in Figure II-3. The flow patterns are turbulent enough to disrupt the thermocline by tilting it toward the leeward side of the lake. After the wind stops, internal water movement causes the tilted upper and lower water regions, which are separated by the thermocline, to oscillate back and forth until the pre-wind stress steady-state condition returns (Wetzel, 1975). This type of water movement caused by wind stress and subsequent oscillations is known as a seiche.

Simply stated, an external seiche is a free oscillation of water, in the form of long standing surface wave, reestablishing equilibrium after having been displaced. The external seiche attains its maximum amplitude at the surface while the internal seiche, which is associated with the density gradient in stratified lakes, attains it maximum amplitude at or near the thermocline (Figure II-4). In stratified waterbodies, the layers of differing density oscillate relative to each other, and the amplitude of the internal standing wave or internal seiche of the metalimnion is much greater than that of the external or surface seiche. Because of the extensive water movement associated with internal seiches, the resulting currents lead to vertical and horizontal transport of heat and dissolved substances (including nutrients) and significantly affect the distribution and productivity of plankton (Wetzel, 1975).

Inflow and Outflow. Lake currents and the resultant mixing and horizontal transport of the water mass may also be a function of inflow and outflow patterns and volumes. Influent velocity generally decreases as the flow enters the lake. Inflowing water of a given temperature and density tends to seek a level of similar density in the lake. Three types of currents may be generated by river influents, as shown in Figure II-5. Overflow occurs when inflow water density is less than lake water density. Underflow occurs when inflow density is greater than lake water density. Interflow occurs when there is a density gradient in the lake, as during periods of stratification, where inflow is greater in density than the epilimnion but is less dense than the hypolimnion.

For a completely mixed lake where no density gradient exists, the outflow draws on the totally mixed volume with little consequence to the net flow within the lake. In stratified impoundments, where outflows could be from different levels (e.g., reservoir release or withdrawal operations), the discharge comes from only a limited zone (or layer) within the lake or reservoir. The thickness of the withdrawal layer is a function of the density gradient in the region of the outlet.

Coriolis Effect. For very large lakes, like the Great Lakes, the Coriolis effect can influence the currents within the lake. This effect is caused by the inertial force created by the earth's rotation. It deflects a moving body (water in this case) to the right (of the line of action of the

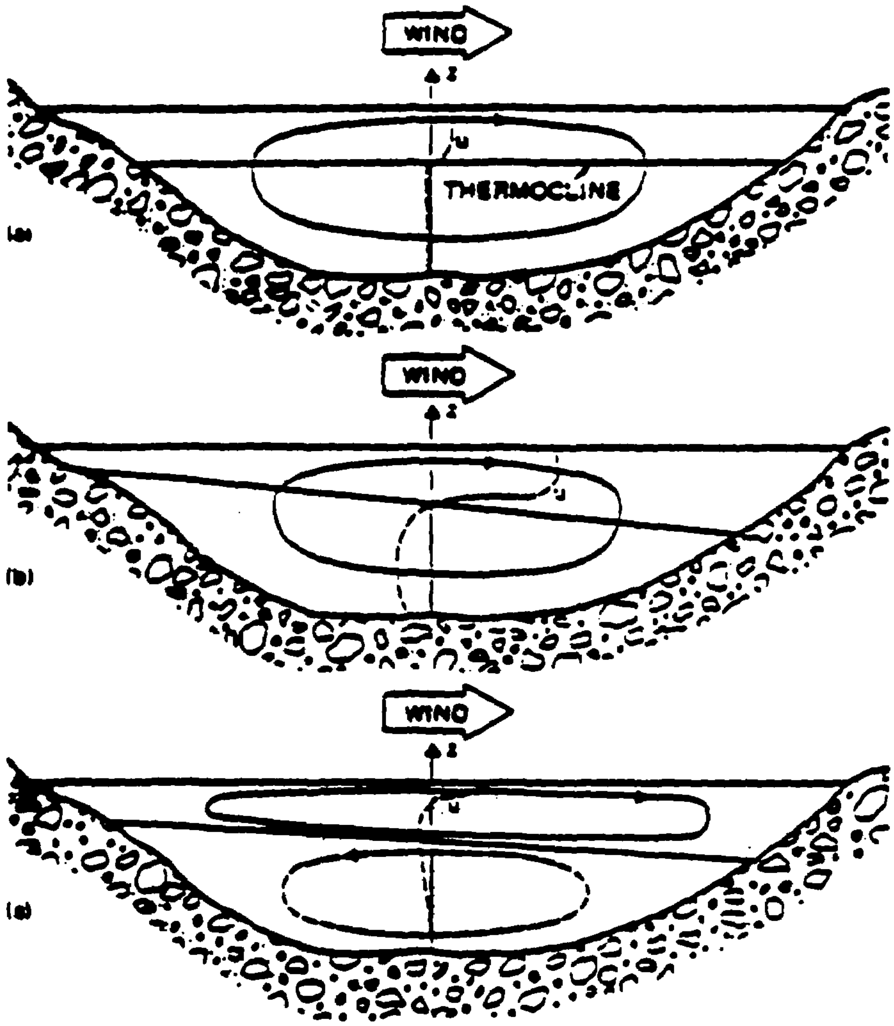

Figure II-3. Formation of baroclinic motions in a lake exposed to wind stresses at the surface: (a) initiation of motion, (b) position of maximum shear across the thermocline (c) steady-state baroclinic circulation (from Fischer, 1979)

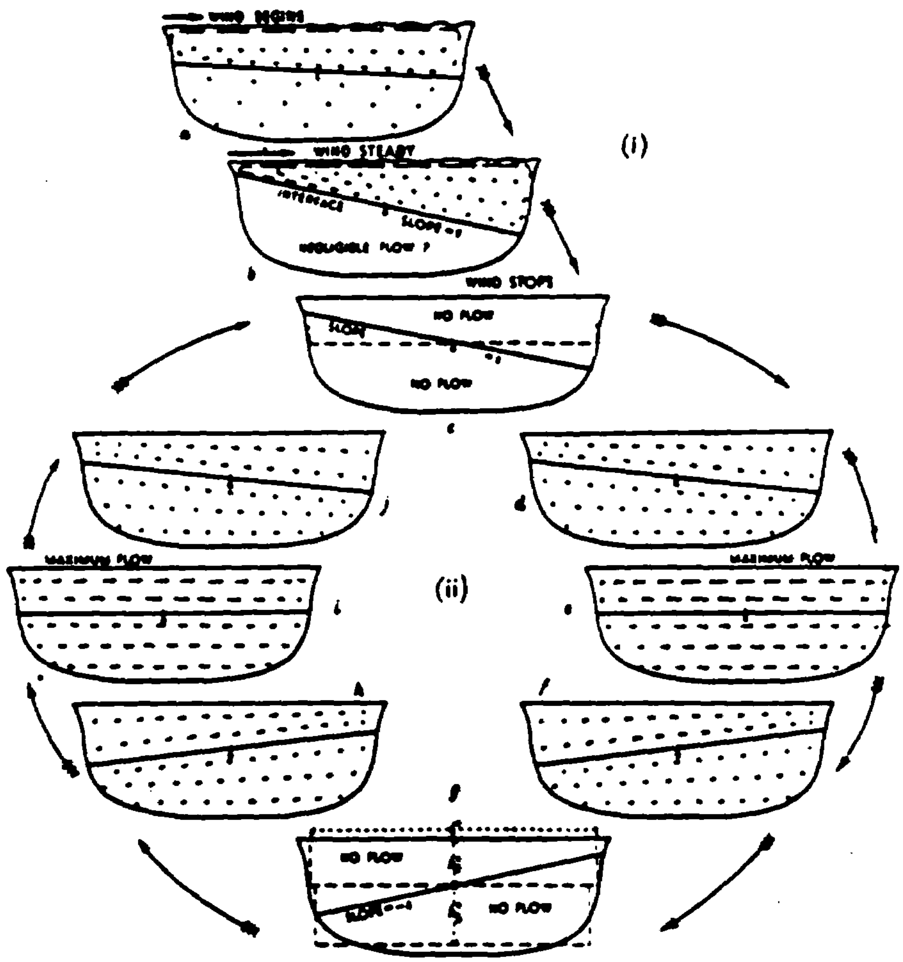

Figure II-4. Movement caused by (i) wind stress and (ii) a subsequent internal seiche in a hypothetical two-layered lake, neglecting friction. Direction and velocity of flow are approximately indicated by arrows. o = nodal section. (from Mortimer, 1952)

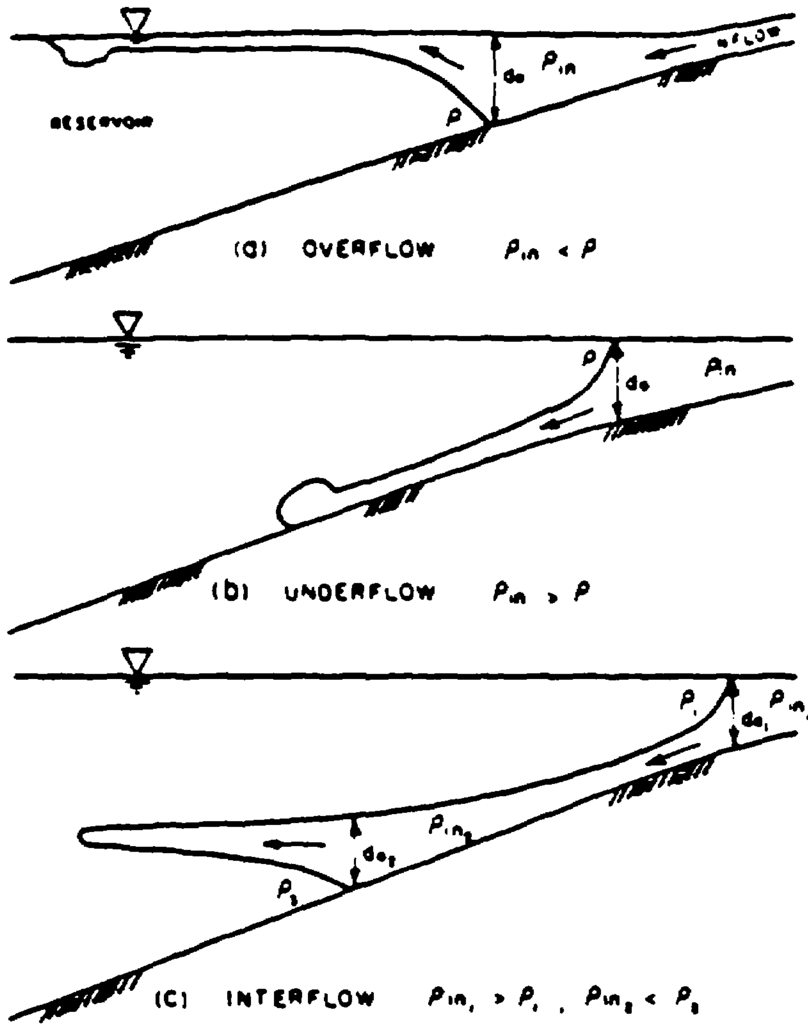

FIGURE II-5. Types of inflow into lakes and reservoirs (from Wunderlich, 1971) earth's rotation) in the Northern Hemisphere and to the left in the Southern Hemisphere. The Coriolis effect causes the surface water to move to the right of the prevailing direction of the wind. Under these conditions in a stratified lake, less dense water tends to form on the right side of the predominant current while denser water collects on the left side of the current (Wetzel, 1975).

Heat Budget

The temperature and temperature distribution within lakes and reservoirs affect not only the water quality within the lake but also the thermal regime and quality of a river system downstream of the lake. The thermal regime of a lake is a function of the heat balance around the body of water. Heat transfer modes into and out of the lake include: heat transfer through the air-water interface, conduction through the mud-water interface, and inflow and outflow heat advection.

Heat transfer across the mud-water interface is generally insignificant while the heat transfer through the air-water interface is primarily responsible for typical annual temperature cycles in lakes.

Heat is transferred across the air-water interface by three different processes: radiation exchange, evaporation, and conduction. The individual heat terms associated with these processes are shown in Figure II-6 and are defined in Table II-1 along with typical ranges of their magnitudes in northern latitudes.

The expression that results from the summation of these various energy fluxes is:

$$H_N = H_{sn} + H_{an} - (H_b + H_e \pm H_c) \qquad (1)$$

where

H_N = net energy flux through the air-water interface, Btu/ft^2-day

H_{sn} = net short-wave solar radiation flux passing through the interface after losses due to absorption and scattering in the atmosphere and by reflection at the interface, Btu/ft^2-day

H_{an} = net long-wave atmospheric radiation flux passing through the interface after reflection, Btu/ft^2-day

H_b = outgoing long-wave back radiation flux, Btu/ft^2-day

H_c = convective energy flux passing back and forth between the interface and the atmosphere, Btu/ft^2-day

H_e = energy loss by evaporation, Btu/ft^2-day

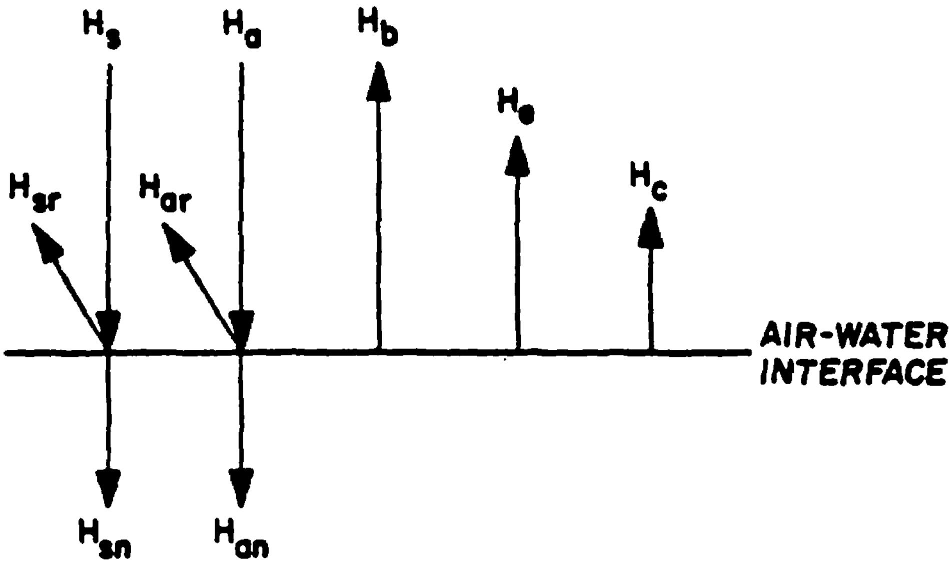

Figure II-6. Heat Transfer Terms Associated with Interfacial Heat Transfer (from Roesner, 1981)

TABLE II-1

DEFINITION OF HEAT TRANSFER TERMS
ILLUSTRATED IN FIGURE II-6

Heat Term	Units	Magnitude ($BTU\ ft^{-2}\ day^{-1}$)
H_s = total incoming solar or short-wave radiation	$HL^{-2}T^{-1}$	400-2800
H_{sr} = reflected short-wave radiation	$HL^{-2}T^{-1}$	40-200
H_a = total incoming atmospheric radiation	$HL^{-2}T^{-1}$	2400-3200
H_{ar} = reflected atmospheric radiation	$HL^{-2}T^{-1}$	70-120
H_b = back radiation from the water surface	$HL^{-2}T^{-1}$	2400-3600
H_e = heat loss by evaporation	$HL^{-2}T^{-1}$	150-3000
H_c = heat loss by conduction to atmosphere	$HL^{-2}T^{-1}$	-320 to +400

where

H = units of heat energy (e.g., BTU)

L = units of length

T = units of time

SOURCE: Roesner, et al., 1981.

These mechanisms by which heat is exchanged between the water surface and the atmosphere are fairly well understood and are documented in the literature (Edinger and Geyer, 1965). The functional representation of these terms has been defined by Water Resources Engineers, Inc. (1967).

The heat flux of the air-water interface is a function of location (latitude, longitude and elevation), season of the year, time of day and meteorological conditions in the vicinity of the lake. Meteorological conditions which affect the heat exchange are cloud cover, dew-point temperature, barometric pressure and wind.

Light Penetration

The heat budget discussed above is also descriptive of the light flux at the air-water interface. The transmission of light through the water column influences primary productivity, growth of aquatic plants, distribution of organisms and behavior of fish.

The reduction of light through the water column of a lake is a function of scattering and absorption where absorption is defined as light energy transformed to heat. Light transmission is affected by the water surface film, floatable and suspended particulates, turbidity, dense populations of algae and bacteria, and color.

The intensity at a given depth is a function of light intensity at the surface and the parameters mentioned above which attenuate the light. Attenuation is usually represented by the use of a light extinction coefficient.

An important physical parameter based on the transmission of light is the depth to which photosynthetic activity is possible. The minimum light intensity required for photosynthesis has been established to be about 1.0 percent of the incident surface light (Cole, 1979). From the depth at which this intensity occurs to the surface is called the euphotic zone. Percent light levels can be measured by a subsurface photometer which can be used to establish the depth of 1.0 percent illumination. A simple measurement of light penetration depth is made with the Secchi disc which is lowered into the water to record the depth at which it disappears to the observer. The depth of the 1.0 percent surface light intensity may be estimated as 2.7 to 3.0 times the Secchi disk transparency (Cole, 1979).

The percent of the surface incident light which reaches different depths is highly variable for individual lakes. Cole (1979) presents examples of the percent incident light by depth for various bodies of water, as shown in Figure II-7.

Lake Stratification

Lakes in temperate and northern latitudes typically exhibit vertical density stratification during certain times of the year. Stratification in lakes is primarily due to temperature differences (i.e., thermal stratification), although salinity and suspended solids concentration may also affect density.

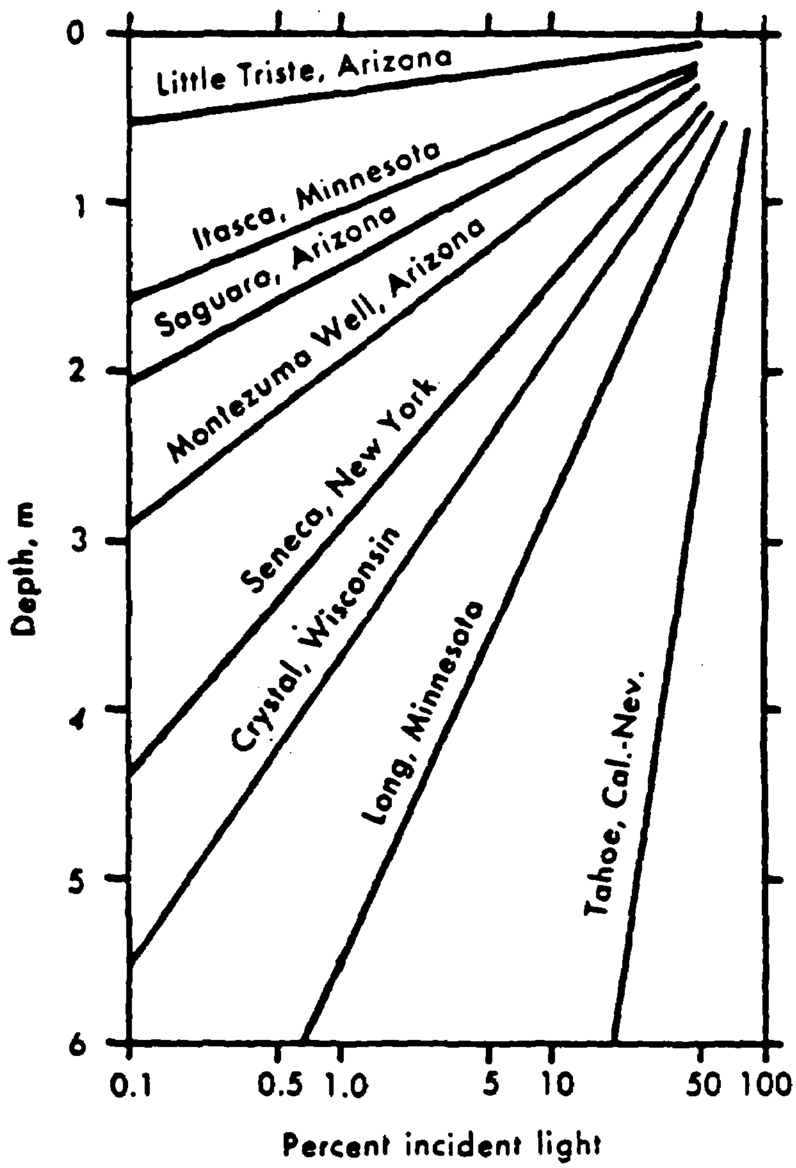

FIGURE II-7. Vertical penetration of light in various bodies of water showing percentage of incident light remaining at different depths (from Cole, 1978)

Lake stratification is best explained by a discussion of a generalized annual temperature cycle. For a period in spring, lakes commonly circulate from surface to bottom, resulting in a uniform temperature profile. This vernal mixing has been called the spring overturn. As surface temperatures warm further, the surface water layer becomes less dense than the colder underlying water, and the lake begins to stratify. This stratified condition, called direct stratification, exists throughout the summer, and the increasing temperature differential between the upper and lower layers increases the stability (resistance to mixing) of the lake.

The upper mixed layer of warm, low-density water is termed the epilimnion, while the lower, stagnant layer of cold, high-density water is termed the hypolimnion. The transition zone between the epilimnion and hypolimnion has been called, among other names, the metalimnion. This narrow transition zone is characterized by rapidly declining temperature with depth, and it contains the thermocline which is the plane of maximum rate of decrease in temperature. The region in which the temperature gradient exceeds 1°C per meter may be used as a working definition of the thermocline. A diagram of the three zones and the thermocline is presented in Figure II-8, and Figure II-9 is a diagram of an annual temperature cycle in which direct stratification occurs.

As surface water temperatures cool in the fall, the density difference between isothermal strata decreases and lake stability is weakened. Eventually, wind-generated currents are sufficiently strong to break down stratification and the lake circulates from surface to bottom (fall overturn). In warmer temperate regions, a lake may retain this completely mixed condition throughout the winter, but in colder regions, particularly following the formation of ice, inverse stratification often develops resulting in winter stagnation. In this condition, the most dense, 4°C water constitutes the hypolimnion which is overlied by less dense, colder water between 0°C and 4°C. The difference in density between 0°C and 4°C is very small, thus inverse stratification results in only a minor density gradient just below the surface. Hence, the stability of inverse stratification is low and, unless the lake is covered with ice, is easily disrupted by wind mixing.

During stratification, the presence of the thermocline suppresses many of the mass transport phenomena that are otherwise responsible for the vertical transport of water quality constituents within a lake. The aquatic community is highly dependent on the thermal structure of such stratified lakes.

Retardation of mass transport between the hypolimnion and the epilimnion results in sharply differentiated water quality and biology between the lake strata. For example, if the magnitude of the dissolved oxygen transport rate across the thermocline is low relative to the dissolved oxygen demand exerted in the hypolimnion, vertical stratification of the lake will occur with respect to the dissolved oxygen concentration. Consequently, as ambient dissolved oxygen concentrations in the hypolimnion decrease, the life functions of many organisms are impaired and the biology and biologically mediated reactions fundamental to water quality are altered. Major changes occur if the dissolved oxygen concentration goes to zero and anaerobic conditions result. Large diurnal fluctuations of

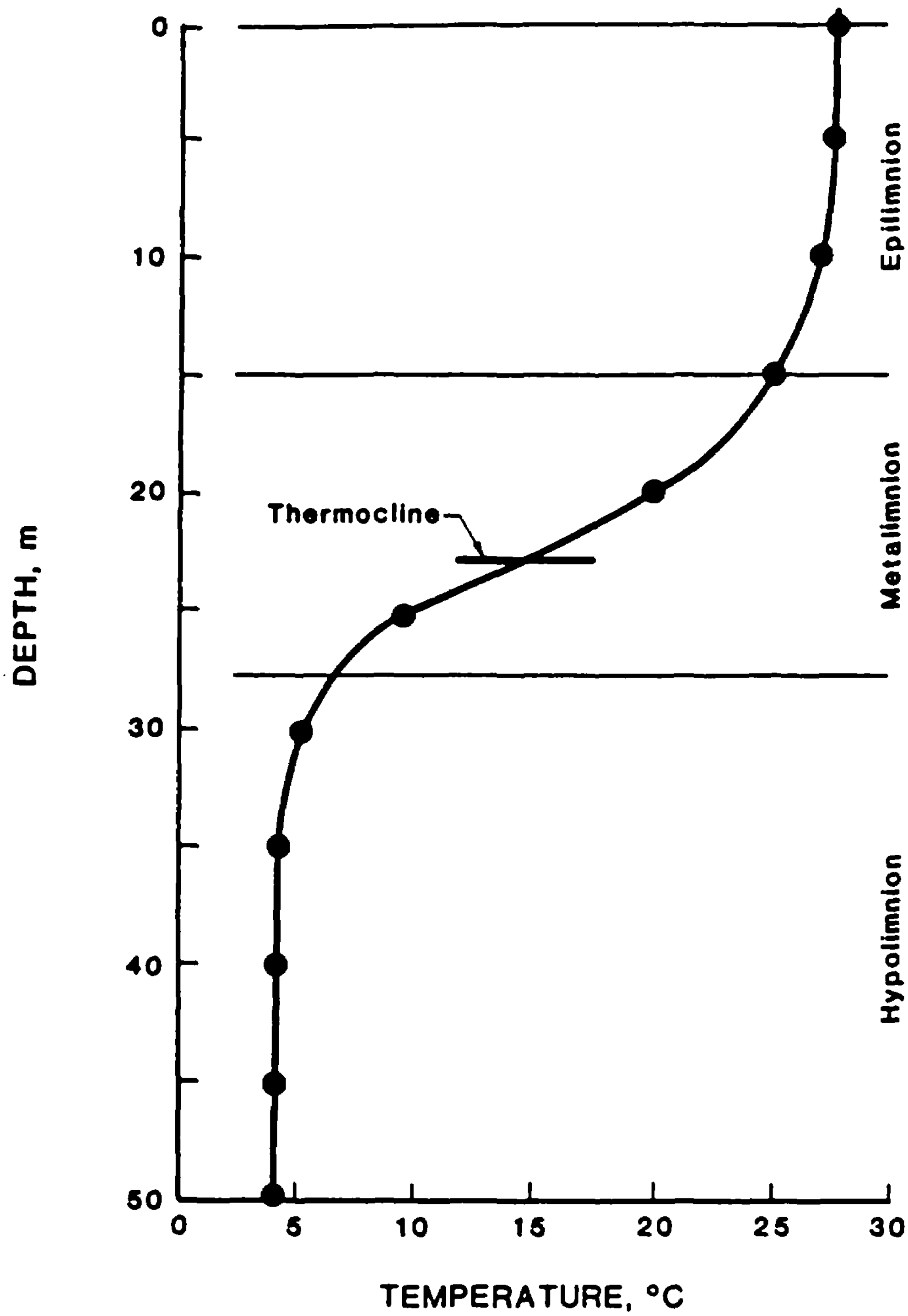

FIGURE II-8. Vertical temperature profile showing direct stratification and the lake regions defined by it (from Cole, 1979).

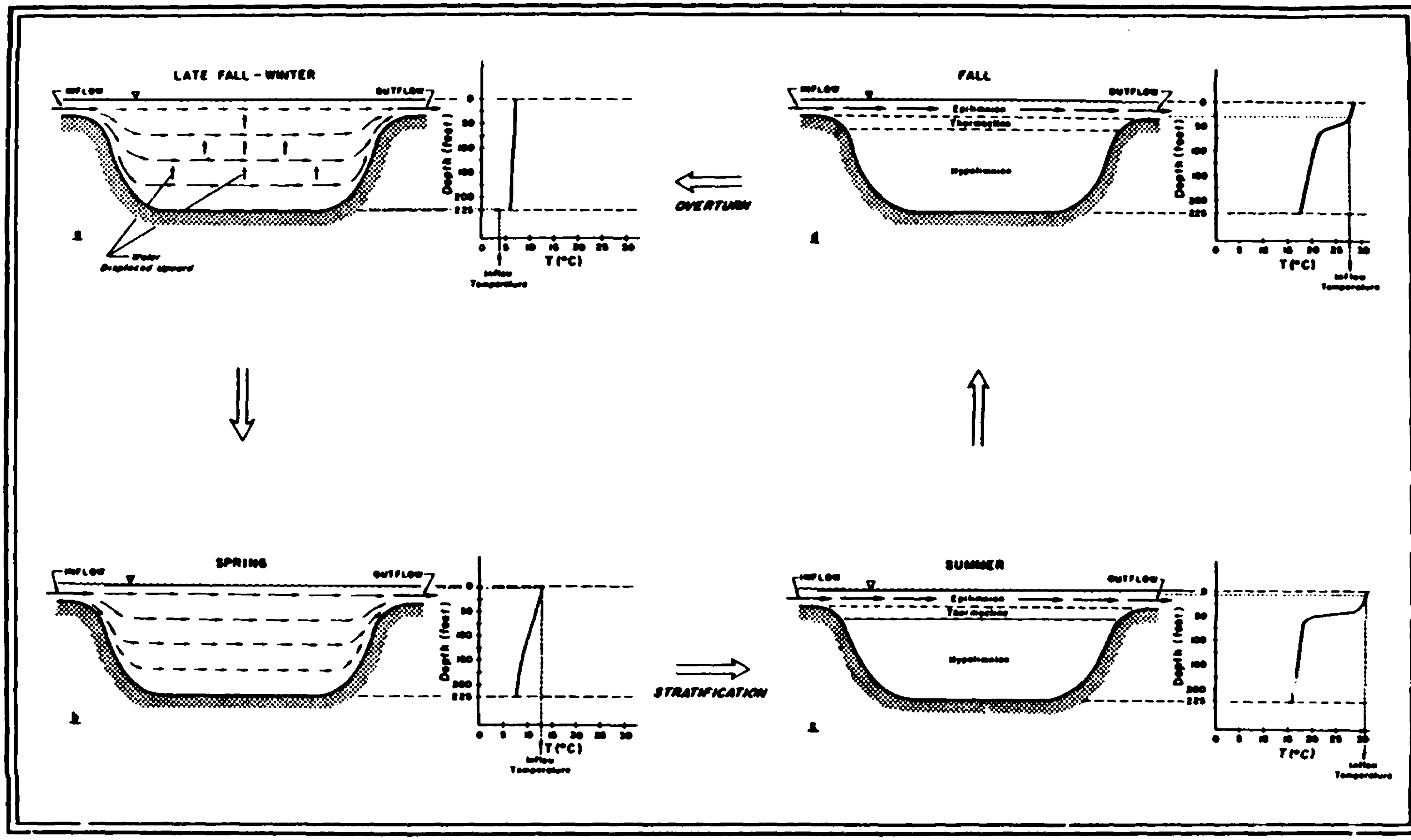

Figure II-9. Annual Cycle of Thermal Stratification and Overturn in an Impoundment (from Zison et al, 1977) dissolved oxygen concentrations in the epilimnion can also occur due to daytime photosynthetic oxygen production superimposed over the continuous oxygen demand from biotic respiration.

Vertical stratification of a lake with respect to nutrients can also occur. In the euphotic zone, dissolved nutrients are converted to particulate organic material through the photosynthetic process. Because the euphotic zone of an ecologically advanced lake does not extend below the thermocline, this assimilation of the dissolved nutrients lowers the ambient nutrient concentrations in the epilimnion. Subsequent sedimentation of the particulate algae and other organic matter then serves to transport the organically bound nutrients to the hypolimnion where they are released by decomposition. In addition, the vertical transport of the released nutrients upward through the thermocline is suppressed by the same mechanisms that inhibit the downward transport of dissolved oxygen. Thus, several processes combine to reduce nutrient concentrations in the epilimnion while simultaneously enriching the hypolimnion.

In addition to the effect of the temperature structure on the movement of water quality constituents, the temperature at any point has a more direct impact on the biology and therefore the water quality structure of an impoundment. All life processes are temperature dependent. In aquatic environments, growth, respiration, reproduction, migration, mortality and decay are strongly influenced by the ambient temperature. According to the van't Hoff rule, within a certain tolerance range, biological reaction rates approximately double with a 10°C increase in temperature.

Annual Circulation Pattern and Lake Classification

Lakes can be classified on the basis of their pattern of annual mixing as described below.

<u>Amixis</u> Amictic lakes never circulate. They are permanently covered with ice, and are mostly restricted to the Antarctic and very high mountains.

<u>Holomixis</u> In holomictic lakes, wind-driven circulation mixes the entire lake from surface to bottom. Several types of holomictic lakes have been described.

<u>Oligomictic lakes</u> are characterized by circulation that is unusual, irregular, and in short duration. These are generally tropical lakes of small to moderate area or lakes of very great depth. They may circulate only at irregular intervals during periods of abnormally cold weather.

<u>Monomictic lakes</u> undergo one regular period of circulation per year. <u>Cold monomictic</u> lakes are frozen in the winter (and therefore stagnant and inversely stratified) and mix throughout the summer. Cold monomictic lakes are ideally defined as lakes whose water temperature never exceeds 4°C. They are generally found in the Arctic or at high altitudes. <u>Warm monomictic</u> lakes circulate in the winter at or above 4°C and stratify directly during the summer. Warm monomixis is common to warm

regions of temperate zones, particularly coastal areas, and to mountainous areas of subtropical latitudes. Warm monomictic lakes are prevalent in coastal regions of North America and northern Europe.

Dimictic lakes circulate freely twice a year in spring and fall, and are directly stratified in summer and inversely stratified in winter. Dimixis is the most common type of annual mixing observed in cool temperate regions of the world. Most lakes of central and eastern North America are dimictic.

Polymictic lakes circulate frequently or continuously. Cold polymictic lakes circulate continually at temperatures near or slightly above 4°C. Warm polymictic lakes circulate frequently at temperatures well above 4°C. These lakes are found in equatorial regions where air temperatures change very little throughout the year.

Meromixis — Meromictic lakes do not circulate throughout the entire water column. The lower water stratum is perennially stagnant and is called the monimolimnion. The overlying stratum, the mixolimnion, circulates periodically, and the two strata are separated by a severe salinity gradient called the chemocline.

Internal Flow and Lake Classification

Experience with prototype lakes (Roesner, 1969) has revealed that with respect to internal flow structure there are basically three distinct classes of lakes. These classes are:

- o The strongly-stratified, deep lake which is characterized by horizontal isotherms.
- o The weakly stratified lake characterized by isotherms which are tilted along the longitudinal axis of the reservoir.
- o The nonstratified, completely mixed lake whose isotherms are essentially vertical.

The single most important parameter determining which of the above classes a lake will fall is the densimetric Froude number, F, which can be written for the lake as:

$$F = (LQ/DV)\,(\rho_0/g\beta)^{1/2} \tag{2}$$

where

L = lake length, m
Q = volumetric discharge through the lake, m^3/s
D = mean lake depth, m
V = lake volume, m^3
ρ_0 = reference density, taken as 1,000 kg/m^3
β = average density gradient in the lake, kg/m^4
g = gravitational constant, 9.81 m/s^2

This number is the ratio of the inertial force of the horizontal flow to the gravitational forces within the stratified impoundment; consequently, it is a measure of the success with which the horizontal flow can alter the internal density (thermal) structure of the lake from that of its gravitational static equilibrium state.

In deep lakes, the fact that the isotherms are horizontal indicates that the inertia of the longitudinal flow is insufficient to disturb the overall gravitational static equilibrium state of the lake except possibly for local disturbances in the vicinity of the lake or reservoir outlets and at points of tributary inflow. Thus, it is expected that F would to be small for such lakes. In completely mixed lakes, on the other hand, the inertia of the flow and its attendant turbulence is sufficient to completely upset the gravitational structure and destratify the reservoir. For lakes of this class, F will be large. Between these two extremes lies the weakly stratified lake in which the longitudinal flow possesses enough inertia to disrupt the reservoir isotherms from their gravitational static equilibrium state configuration, but not enough to completely mix the lake.

For the purpose of classifying lakes by their Froude number, β and ρ_0 in equation (2) may be approximated as 10^{-3} kg/m^4 and 1000 kg/m^3, respectively. Substituting these values and g into equation (2) leads to an expression for F as:

$$F = (320)\ (LQ/DV) \qquad (3)$$

where L and D have units of meters, Q is in m^3/s, and V has units of m^3. It is observed from this equation that the principal lake parameters that determine a lake's classification are its length, depth, and discharge to volume ratio (Q/V).

In developing some familiarity with the magnitude of F for each of the three lake classes, it is helpful to note that theoretical and experimental work in stratified flow indicates that flow separation occurs in a stratified fluid when the Froude number is less than $1/\pi$, i.e., for $F < 1/\pi$, part of the fluid will be in motion longitudinally while the remainder is essentially at rest. Furthermore, as F becomes smaller and smaller, the flowing layer becomes more and more concentrated in the vertical direction. Thus, in the deep lake it is expected that the longitudinal flow is highly concentrated at values of $F \ll 1/\pi$ while in the completely mixed case F must be at least greater than $1/\pi$ since the entire lake is in motion and it may be expected in general that $F \gg 1/\pi$. Values of F for the weakly stratified case would fall between these two limits and might be expected to be on the order of $1/\pi$. As an illustration, five lakes are listed in Table II-2 with their Froude numbers. It is known that Hungry Horse Reservoir and Detroit Reservoir are of the deep reservoir class and can be effectively described with a one-dimensional model along the vertical axis of the lake. Lake Roosevelt, which has been observed to fall into the weakly stratified class is seen to have a Froude number on the order of $1/\pi$, which is considerably larger than F for either Hungry Horse or Detroit Reservoirs. Finally, Priest Rapids and Wells Dams, which are essentially completely mixed along their vertical axes, show Froude numbers much larger than $1/\pi$, as expected.

TABLE II-2

IMPOUNDMENT FROUDE NUMBERS

RESERVOIR	LENGTH (meters)	AVERAGE DEPTH (meters)	DISCHARGE TO VOLUME RATIO (sec^{-1})	F	CLASS
Hungry Horse	4.7×10^4	70	1.2×10^{-8}	0.0026	Deep
Detroit	1.5×10^4	56	3.5×10^{-8}	0.0030	Deep
Lake Roosevelt	2.0×10^5	70	5.0×10^{-7}	0.46	Weakly Stratified
Priest Rapids*	2.9×10^4	18	4.6×10^{-6}	2.4	Completely Mixed
Wells*	4.6×10^4	26	6.7×10^{-6}	3.8	Completely Mixed

*River run dams on the Columbia River below Grand Coulee Dam.

SOURCE: Roesner, 1969.

Sedimentation in Lakes

One physical process that is particularly important to the aquatic community is the deposition of sediment which is carried from the contributing watershed into the body of the lake. Because of the low velocities through a lake, reservoir or impoundment, sediments transported by inflowing waters tend to settle to the bottom before they can be carried through the lake outlets.

Sediment accumulation rates are strongly dependent both on the unique physiographic characteristics of a specific watershed and upon various characteristics of the lake. Although sediment accumulation rates can be transposed from one lake to another, this should be done with a careful consideration of watershed characteristics (Department of Agriculture, 1975, 1979). Apart from the use of predictive computer models, sediment accumulation rates may be determined in one of two basic ways: (1) by periodic sediment surveys on a lake; or (2) by estimates of watershed erosion and bed load. Watershed erosion and bed load may be translated into sediment accumulation rate through use of the trap efficiency, defined as the proportion of the influent pollutant (in this case sediment) load that is retained in the basin. The second method usually employs the development of sediment discharge rate as a function of water discharge. Such a sediment-rating curve is illustrated in Figure II-10. From such relationships, annual sediment transport to the lake is developed and applied to the lake or reservoir trap efficiency functions to develop the sediment accumulation rates. Trap efficiencies have been developed as a function of the lake capacity-inflow ratio, as shown in Figure II-11. Other methods for predicting trap efficiency are described by Novotny and Chesters (1981) and Whipple et al. (1983).

Accumulated sediment in lakes can, over many years, reduce the life of the water body by reducing the water storage capacity. Sediment flow into lakes also reduces light penetration, eliminates bottom habitat for many plants and animals, and carries with it adsorbed chemicals and organic matter which settle to the bottom and can be harmful to the ecology of the lake. Where sediment accumulation is a major problem, proper watershed management including erosion and sediment control must be put into effect.

CHEMICAL CHARACTERISTICS

Overview of Physico-Chemical Phenomena in Lakes

Water chemistry phenomena that are characteristic of freshwater have been discussed in Section III, Technical Support Manual: Water Body Surveys and Assessments for Conducting Use Attainability Analyses (U.S. EPA, 1983b). The material in Section III is applicable to lakes as well as rivers and streams. The reader should refer to this Manual for a discussion of hardness, alkalinity, pH and salinity, and for a discussion of a number of indices of water quality. It would also be helpful to refer to Volume II of this series, Technical Support Manual: Water Body Surveys and Assessments for Conducting Use Attainability Analyses, Volume II: Estuarine Systems, for a discussion of eutrophication and the importance of aquatic vegetation. Even though the flora and fauna of estuaries have adapted to

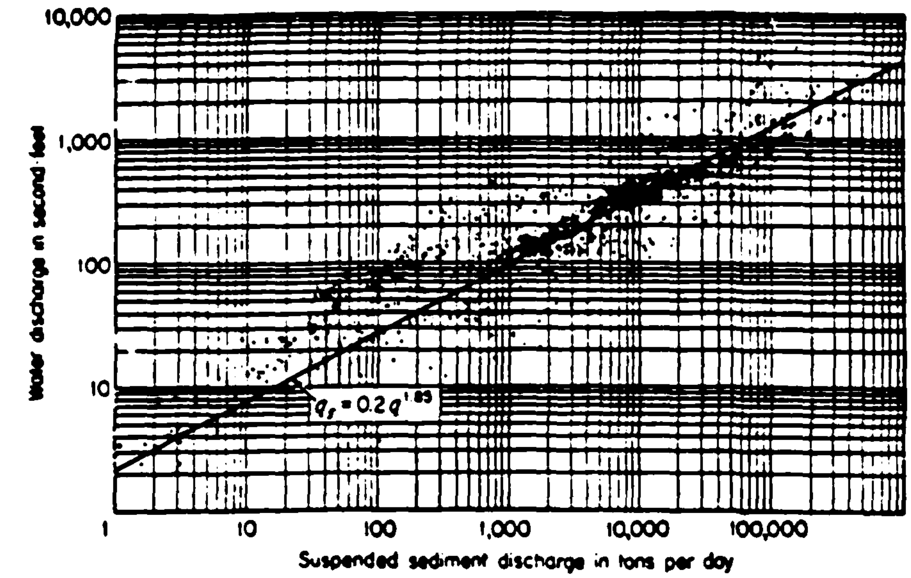

Figure II-10. Sediment-rating curve for the Powder River at Arvada, Wyoming (from Fleming, 1969)

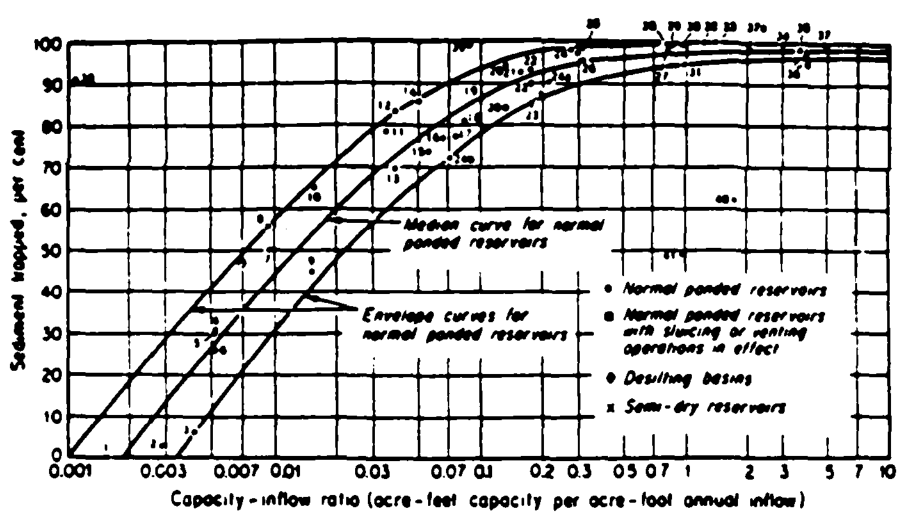

Figure II-11. Reservoir trap efficienty as a function of the capacity-inflow ratio (from Brune, 1953) higher salinities than will be found in the lake, many of the interrelationships of biology and nutrient cycling in the estuary have their counterparts in the lake.

The discussion to follow will be limited to chemical phenomena that are of particular importance to lakes. This will focus on nutrient cycling and eutrophication, but will of necessity also be concerned with the effects of variable pH, dissolved oxygen, and redox potential on lake processes.

Water chemistry in a lake and stages in the annual lake turnover cycle are closely related. Turnover was discussed in greater detail earlier in this chapter in the section on physical processes. For the current discussion on lake water chemistry, we shall refer primarily to the stratified lake that undergoes the classic lake turnover cycle. Since the patterns of lake stratification and turnover vary widely, depending upon such factors as depth, and prevailing climate as characterized by altitude and latitude, the discussion to follow on water chemistry may not be applicable to all lakes.

Once a thermocline has formed, the dissolved oxygen (DO) concentration of the hypolimnion tends to decline. This occurs because the hypolimnion is isolated from surface waters by the thermocline, and there is no mechanism for the aeration of the hypolimnion. In addition, the decay of organic matter in the hypolimnion as well as the oxygen requirements of fish and other organisms in the hypolimnion serve to deplete DO.

With the depletion of DO, reducing conditions prevail and many compounds that have accumulated in the sediment by precipitation are released to the surrounding water. Compounds that are solubilized under such conditions include compounds of nitrogen, phosphorus, iron, manganese and calcium. Phosphorus and nitrogen are of particular concern because of their role in eutrophication processes in lakes.

Nutrients released from bottom sediments under stratified conditions are not available to phytoplankton in the epilimnion. However, during overturn periods, mixing of the hypolimnion and the epilimnion distributes nutrients throughout the water column, making them available to primary producers near the surface. This condition of high nutrient availability is short-lived because the soluble reduced forms are rapidly oxidized to insoluble forms which reprecipitate. Phosphorus and nitrogen are also deposited through sorption to particles that settle to the bottom, and are transported from the epilimnion to the hypolimnion in dead plant material that is added to sediments.

A special case occurs for ice covered lakes, esepcially when a layer of snow effectively stops light penetration into the water. Under these conditions winter algal photosynthesis is curtailed and dissolved oxygen (DO) concentrations may decline as a result. A declining DO may affect both the chemistry and the biology of the system. The curtailment of winter photosynthesis may not pose a problem for a large body of water. For a small lake, however, respiration and decomposition processes may deplete available DO enough to result in fish kills.

The chemical processes that occur during the course of an annual lake cycle are rather complex. They are driven by pH, oxidation-reduction potential, concentration of dissolved oxygen, and by such phenomena as the carbonate buffering system which serves to regulate pH while providing a source of inorganic carbon which may contribute to the many precipitation reactions of the lake. The water chemistry of the lake may be better appreciated through a detailed review of such references as Butler (1964), and Stumm and Morgan (1981).

Of the many raw materials required by aquatic plants (phytoplankton and macrophytes) for growth, carbon, nitrogen and phosphorus are of particular importance. The relative and absolute abundance of nitrogen and phosphorus are important to the extent of growth of aquatic plants that may be seen in a lake. If these nutrients are available in adequate supply, massive algal and macrophyte blooms may occur with severe consequences for the lake.

The concept of the existence of a limiting nutrient is the crux of Liebig's "law of the minimum" which basically states that growth is limited by the essential nutrient that is available in the lowest supply relative to requirements. This applies to the growth of primary producers and to the process of eutrophication in lakes where either phosphorus or nitrogen is usually the limiting nutrient.

Algae require carbon, nitrogen and phosphorus in the approximate atomic ratio of 100:15:1 (Uttormark, 1979), which corresponds to a 39:7:1 ratio on a mass basis. The source of carbon is carbon dioxide which exists in essentially unlimited supply in the water and in the atmosphere. Nitrogen also is abundant in the environment and is not realistically subject to control. Nitrate is introduced to the water body in rainfall, having been produced electrochemically by lightening; in runoff to the water body; and may be produced in the water body itself through the nitrification of ammonia by sediment bacteria (Hergenrader, 1980). In contrast, many sources of phosphorus to a lake are anthropogenic.

There are some lakes that are nitrogen limited, for which nitrogen controls offer a means of controlling eutrophication. This is unusual, however, and phosphorus limiting situations are much more prevalent than nitrogen limiting conditions. As stated above, a N:P mass ratio of 7:1 is commonly assumed to be required for algal growth; a N:P ratio less than 7:1 indicates that nitrogen is limiting, while a N:P ratio greater than 7:1 indicates a phosphorus limiting situation.

The growth of aquatic plants is limited when low phosphorus concentrations prevail in a water body. Adequate control of phosphorus results in nutrient limiting conditions that will hold the growth of aquatic plants in check. Most inputs of phosphorus to a lake are anthropogenic, thus control of this nutrient offers the best means of regulating the trophic condition of the lake. The focus of the discussion to follow will be an overview of the chemistry of phosphorus and its interractions with pH, dissolved oxygen, carbonates and iron in the water body.

A discussion of phosphorus chemistry may be approached through our understanding of the control of phosphorus in wastewater treatment plants by precipitation reactions. As will be seen in Chapter IV, the principles of phosphorus control in wastewater processes may have application to lakes as well. The chemistry of phosphorus is very complex and will not be discussed in great detail in this Manual. The reader who would like further insight into the fine points of phosphorus chemistry should refer to texts such as Butler (1964), and Stumm and Morgan (1981).

Phosphorus Removal by Precipitation

Phosphorus removal is discussed in detail in Process Design Manual for Phosphorus Removal (U.S. EPA, 1976). Chapter 3 of that Manual, "Theory of Phosphorus Removal by Chemical Precipitation," forms the basis of discussion for this section.

Ionic forms of aluminum, iron and calcium have proven most useful for the removal of phosphorus. Calcium in the form of lime is commonly used to precipitate phosphorus. Hydroxyl ions produced when lime is added to water also play a role in phosphorus removal. Because the chemistry of phosphorus reactions with metal ions is complex, it will be assumed for the sake of simplicity that phosphorus reacts in the form of orthophosphate, PO_4^{3-}.

Aluminum

Aluminum and phosphate ions combine to form aluminum phosphate. The principal source of aluminum is alum, or hydrated aluminum sulfate, which reacts with phosphate as follows:

$$Al_2(SO_4)_3 \cdot 14H_2O + 2PO_4^{3-} \longrightarrow 2AlPO_4 + 3SO_4^{2-} + 14H_2O \qquad (4)$$

The solubility of aluminum phosphate varies with pH and reaches a minimum at pH 6. Greater than stoichiometric amounts of alum generally are required for phosphorus removal because of competing reactions, one of which produces aluminum hydroxide and reduces pH as well. Alum addition has often been used as a means of controlling phosphorus problems in lakes. This is discussed in greater detail in Chapter IV in the section on lake restoration techniques.

Lime

Calcium or magnesium and phosphate ions react in the presence of hydroxyl ion to form hydroxyapatite, $Ca_5(OH)(PO_4)_3$. The reaction is pH dependent, but the solubility of the precipitate is so low that even at pH 9 appreciable amounts of phosphorus are removed. Lime addition has occasionally been used to treat phosphorus problems in lakes, but the high pH required to form and maintain hydroxyapatite generally precludes this as a practical method of control.

Iron

Iron, which is a micronutrient required by algae, has been shown to be limiting in some lakes (Wetzel, 1975) and could be an important factor in the eutrophication of lakes. When a lake is well oxygenated, most iron in the system is tied up in organic, suspended and particulate matter, and very little exists in soluble form (Hergenrader, 1980). Under anoxic conditions in the hypolimnion, iron tends to be released from bottom sediments along with phosphorus that has been tied up in the form of iron and manganese precipitates.

Both ferrous (Fe^{2+}) and ferric (Fe^{3+}) ions may be used to precipitate phosphorus. Ferric iron salts are effective for phosphorus removal at pH 4.5 to 5.0 although significant removal of phosphorus may be attained at higher pH levels. Good phosphorus removal with the ferrous ion is accomplished at pH 7 to 8.

Lazoff (1983) examined phosphorus and iron sedimentation rates during and following overturn to evaluate the removal of phosphorus through adsorption and coprecipitation with iron compounds. At overturn, ferrous iron which has been released along with phosphorus from the sediment, precipitates as ferric hydroxides. Iron precipitation at overturn has been observed as the formation of reddish brown floc particles. Phosphorus is removed from the water column by these floc particles, either through adsorption or through coprecipitation and settling. Thus, large amounts of phosphorus may be removed from the water column and, therefore, become unavailable for phytoplankton growth.

The removal of phosphorus by this mechanism may be aided by phytoplankton and other sources of turbidity in the water. To the extent that these limit light penetration into the water, photosynthesis and phosphorus uptake are inhibited, thus permitting effective removal by ferric iron (Lazoff, 1983).

Dissolved Oxygen

Lake turnover, and mechanical aeration of bottom waters, leads to re-oxygenation of the hypolimnion. If the hypolimnion was previously anoxic, oxygenation will cause a reduction in PO_4^{3-} levels through the formation of iron and manganese complexes and precipitates (Pastorok et al., 1981). The limited ability of iron, manganese and also calcium to tie up phosphorus in a lake is regulated by DO levels and by oxidation-reduction (redox) potential. As the DO of the hypolimnion falls, the redox potential decreases and phosphorus is released during the reduction of metal precipitates that formed when the redox potential was higher. This may not be a problem while the lake remains stratified, but once stratification ends and the lake becomes mixed, the soluble phosphorus becomes available to aquatic plants living near the surface. Lime does not reliably remove phosphorus from the aquatic system because effective removal occurs at pH levels greater than those found in natural waters.

Aluminum complexes are much less susceptible to redox changes and, therefore, are effective in permanently removing particulate and soluble phosphorus from the water column. Removal of phosphorus by aluminum occurs by precipitation, by sorption of phosphates to the surface of aluminum hydroxide floc and by the entrapment and sedimentation of phosphorus containing particulates by aluminum hydroxide floc. Once deposited, the floc of aluminum hydroxide appears to consolidate and phosphorus is apparently sorbed from interstitial water as it flows through the floc (Cooke, 1981).

Oxygen depletion leads to low redox potentials in the sediment and a net release of phosphorus into the water column. With aeration, the redox potential increases causing phosphorus to be precipitated and to be sorbed by the sediment. Low pH values in the hypolimnion may be attributed to high carbon dioxide associated with decay processes in the sediment. With oxygenation, CO_2 levels decrease and pH increases (Fast, 1971).

Eutrophication and Nutrient Cycling

Eutrophication

There are two general ways in which the term "eutrophication" is used. In the first, eutrophication is defined as the process of nutrient enrichment in a water body. In the second, "eutrophication" is used to describe the effects of nutrient enrichment, that is, the uncontrolled growth of plants, particularly phytoplankton, in a lake or reservoir. The second use also encompasses changes in the composition of animal communities in the water body. Both of these uses of the term eutrophication are commonly found in the literature, and the distinction, if important, must be discerned from the context of use in a particular article.

Eutrophication is the natural progression, or aging process, undergone by all lentic water bodies. However, eutrophication is often greatly accelerated by anthropogenic nutrient enrichment, which has been termed "cultural eutrophication."

In lakes nutrient enrichment often leads to the increased growth of algae and/or rooted aquatic plants. For many reasons, however, excessive algal growth will not necessarily occur under conditions of nutrient enrichment; thus, the presence of high nutrient levels may not necessarily portend the problems associated with the second use of the term eutrophication. For example, the water body may be nitrogen limited or phosphorus limited, toxics may be present that inhibit the growth of algae, or high turbidity may inhibit algal photosynthesis despite an abundance of nutrients.

The three basic trophic states that may exist in a lake (or a river or estuary) may be described in very general terms as follows:

- Oligotrophic - the water body is low in plant nutrients, and may be well oxygenated
- Eutrophic - the water body is rich in plant nutrients, and the hypolimnion may be deficient in DO
- Mesotrophic - the water body is in a state between oligotrophic and eutrophic.

What specific range of phosphorus or nitrogen concentration to ascribe to each of these trophic levels is a matter of controversy since the degree of response of a water body to enrichment may be controlled by factors other than nutrient concentrations, in effect making the response site specific. As will be seen in Chapter III, in a discussion of various measures of the trophic state of a lake, eutrophication is a complex process and whether or not a water body is eutrophic is not always clear, although the consequences are.

Nutrients are transported to lakes from external sources, but once in the lake may be recycled internally. A consideration of attainable uses in a lake must include an understanding of the sources of nitrogen and phosphorus, the significance of internal cycling, especially of phosphorus, and the changes that might be anticipated if eutrophication could be controlled.

Nutrient Cycling in Lakes

There are many sources of nitrogen in the lake ecosystem. Significant amounts of this nitrogen stem from natural sources and cannot be controlled. Many anthropogenic sources, such as agricultural runoff, also are not readily controlled. This is true in large part because the policy issues surrounding nitrogen (and phosphorus) control through Best Management Practices (BMPs) have not been resolved even though technical implementation of BMPs could appreciably reduce nutrient loadings to a water body. Once in the aquatic system nitrogen may undergo several bacterially mediated transformations such as nitrification to nitrite and nitrate or denitrification of nitrate to nitrogen. Proteins undergo ammonification to ammonia which in turn is oxidized to nitrate. Also, some Cyanophyta (blue-green algae) are capable of using atmospheric nitrogen. Unlike phosphorus, nitrogen is not readily removed from a system by complexation and precipitation reactions.

Whereas nitrogen inputs to a water body are predominantly non-point sources, phosphorus inputs are predominantly point sources that are more readily identified and controlled. There are some parts of the country, as in Florida, where extensive phosphorus deposits are found which could be the source of significant natural inputs to a lake and its feeder streams. Such lakes may be nitrogen limited. With the exception of runoff, the anthropogenic sources (particularly the point sources) of phosphorus can be controlled to a large extent. Control of the external inputs of phosphorus to a lake may not necessarily end problems of eutrophication, however, annual fluctuations in DO, pH and other parameters may result in the recycling of significant amounts of phosphorus within the system.

Uttormark (1979) has noted that most lakes are nutrient traps, on an annual basis, and that the trophic status of a lake can be dependent on the degree of internal nutrient cycling that occurs. There is typically a seasonal release from and deposition of nutrients to the sediment, and the effect of this internal nutrient cycling is dependent upon physical characteristics such as morphology, mixing processes and stratification.

As discussed earlier, phosphorus that has been released from sediments to anoxic bottom waters under stratified conditions may become temporarily available to primary producers during overturn periods. This often causes phytoplankton blooms in spring and fall. During winter and summer, stratification limits vertical cycling of nutrients and nutrient availability may limit phytoplankton growth.

Macrophytes derive phosphorus directly from lake sediment or from the water column. The release of some of this phosphorus to the surrounding water has been reported for some macrophytes (Landers, 1982). In addition, significant amounts of phosphorus and nitrogen are released to the surrounding water by macrophytes as they die and decompose. Landers has estimated that about one-fourth of the phosphorus and one-half of the nitrogen within a decaying plant will remain as a refractory portion, while the rest is released to the surrounding water.

In response to soluble phosphorus released by decomposing macrophytes, the algal biomass (as measured by chlorophyll-a concentration) may show a significant increase. When these algae later die, phosphorus will be returned to the system in soluble form, as precipitates that form with iron, calcium and manganese, or will be tied up in dead cells that settle to the bottom to become part of the sediment.

Significance of Chemical Phenomena to Use Attainability

The most critical water quality indicators for aquatic use attainment in a lake are dissolved oxygen (DO), nutrients, chlorophyll-a and toxicants. Dissolved oxygen is an important water quality indicator for all fisheries uses and, as we have seen above, is an important factor in the internal cycling of nutrients in a lake. In evaluating use attainability, the relative importance of three forms of oxygen demand should be considered: respiratory demand of phytoplankton and macrophytes during non-photosynthetic periods, water column demand, and benthic demand. If use impairment is occurring, assessments of the significance of each oxygen sink can be useful in evaluating the feasibility of achieving sufficient pollution control, or in implementing the best internal nutrient management practices to attain a designated use.

Chlorophyll-a is a good indicator of algal concentrations and of nutrient overenrichment. Excessive phytoplankton concentrations, as indicated by high chlorophyll-a levels, can cause adverse DO impacts such as: (a) wide diurnal variation in surface DO due to daytime photosynthetic oxygen production and nighttime oxygen depletion by respiration and (b) depletion of bottom DO through the decomposition of dead algae and other organic matter. Excessive algal growth may also result in shading which reduces light penetration needed by submerged plants.

The nutrients of concern in a lake are nitrogen and phosphorus. Their sources typically are discharges from industry and from sewage treatment plants, and runoff from urban and agricultural areas. Increased nutrient levels may lead to phytoplankton blooms and a subsequent reduction in DO levels, as discussed above.

Sewage treatment plants are typically the major point source of nutrients. Agricultural land uses and urban land uses are significant non-point sources of nutrients. Wastewater treatment facilities often are the major source of phosphorus loadings while non-point sources tend to be the major contributors of nitrogen. It is important to base control strategies on an understanding of the sources of each type of nutrient, both in the lake and in its feeder streams.

Clearly the levels of both nitrogen and phosphorus can be important determinants of the uses that can be attained in a lake. Because point sources of nutrients are typically more amenable to control than non-point sources, and because phosphorus removal for municipal wastewater discharges is typically less expensive than nitrogen removal, the control of phosphorus discharges is often the method of choice for the prevention or reversal of use impairment in the lake.

Discussion of the impact of toxicants such as pesticides, herbicides and heavy metals is beyond the scope of this volume. Nevertheless, the presence of toxics in sediments or in the water column may prevent the attainment of uses (particularly those related to fish propagation and maintenance in water bodies) which would otherwise be supported by water quality criteria for DO and other parameters.

TECHNIQUES FOR USE ATTAINABILITY EVALUATIONS

Introduction

In the use attainability analysis, it must initially be determined if the present aquatic life use of a lake corresponds to the designated use. The aquatic use of a lake is evaluated in terms of biological measures and indices. If the designated use is not being achieved, then physical, chemical and biological investigations are carried out to determine the causes of impairment. Physical and chemical factors are examined to explain the lack of attainment, and they are used as a guide in determining the highest use level the system can achieve.

Physical parameters and processes must be characterized so that the study lake can be compared with a reference lake. Physical parameters to be considered are average depth, surface area, volume and retention time. The physical processes of concern include degree of stratification and importance of circulation patterns. Once a reference lake has been selected, comparisons can be made with the lake of interest in terms of water quality differences and differences in biological communities.

Empirical (desktop) and simulation (computer-based mathematical) models can be used to improve our understanding of how physical and chemical characteristics affect biological communities. Desktop analyses may be used to obtain an overall picture of lake water quality. These methods are usually based on average annual conditions. For example, they are used to predict trophic state based on annual loading rates of nutrients. They are simple, inexpensive procedures that provide a useful perspective on lake water quality and in many cases will provide sufficient information for the use study. For a more detailed analysis of lake conditions, computer models can be employed to analyze various aspects of a lake. These models can simulate the distribution of water quality constituents spatially (at various locations within the lake) and temporally (at various times of the year).

Desktop calculations and larger simulation models may both be used to enhance our understanding of existing lake conditions. More importantly, they can be used to evaluate the lake's response to different conditions without actually imposing those conditions on the lake. This is of great benefit in determining the cause of impairment where, for example, the model can predict the lake response to the removal of point and nonpoint loads to the lake system. Models can also be used to assess potential uses by simulating the lake's response to various design conditions or restoration activities. A good discussion of model selection and use is provided by the U.S. EPA (1983c).

Empirical Models

In contrast to the complex computer models available for the study of lake processes, there are a number of simple empirical, input/output models that have proven to be widely applicable to lake studies. Most of these models consider phosphorus loadings or chlorophyll-a concentrations in order to estimate the trophic status of a lake.

Vollenweider Model

Vollenweider (1975) proposed an empirical fit to a simplified phosphorus mass balance model, using the factor:

$$\sigma = 10/\bar{z}$$

where

σ = specific sedimentation rate, years^{-1}
$\bar{z}$ = mean lake depth, m

Sedimentation is used by Vollenweider to describe all net internal losses of phosphorus (Uttormark, 1978) and is extremely difficult to determine experimentally. Vollenweider derived his value for σ through an analysis of specific sedimentation rate versus mean depth for actual lake data. Under steady state conditions, the phosphorus concentration may be expressed in terms of phosphorus loadings as:

$$[P] = L/(10 + \bar{z}\rho) \qquad (5)$$

where

$[P]$ = in-lake total phosphorus concentration, ML^{-3}
L = specific areal phosphorus loading, $ML^{-2}T^{-1}$
$\bar{z}$ = mean lake depth, L
ρ = flushing rate, Q/V, T^{-1}
Q = annual water flow rate, L^3T^{-1}
V = lake volume, L^3
M = units of mass
L = units of length
T = units of time

Vollenweider examined the relationship of areal loading rate to mean depth times flushing rate and defined in-lake phosphorus concentrations of 10 mg/m^3 to distinguish oligotrophic from mesotrophic conditions, and 20 mg/m^3 to distinguish mesotrophic from eutrophic conditions. Solving Equation 5 for L and substituting the predefined values of 10 mg/m^3 or 20 mg/m^3 for [P], Vollenweider developed the type of plot shown in Figure II-12a (Zison, et al., 1977) which provides a simple, straightforward means by which to use phosphorus loading to a lake to assess trophic level. Vollenweider's model, and other models that use phosphorus loading to evaluate eutrophication-related water quality, generally are only applicable to water bodies in which algal growth is limited by phosphorus.

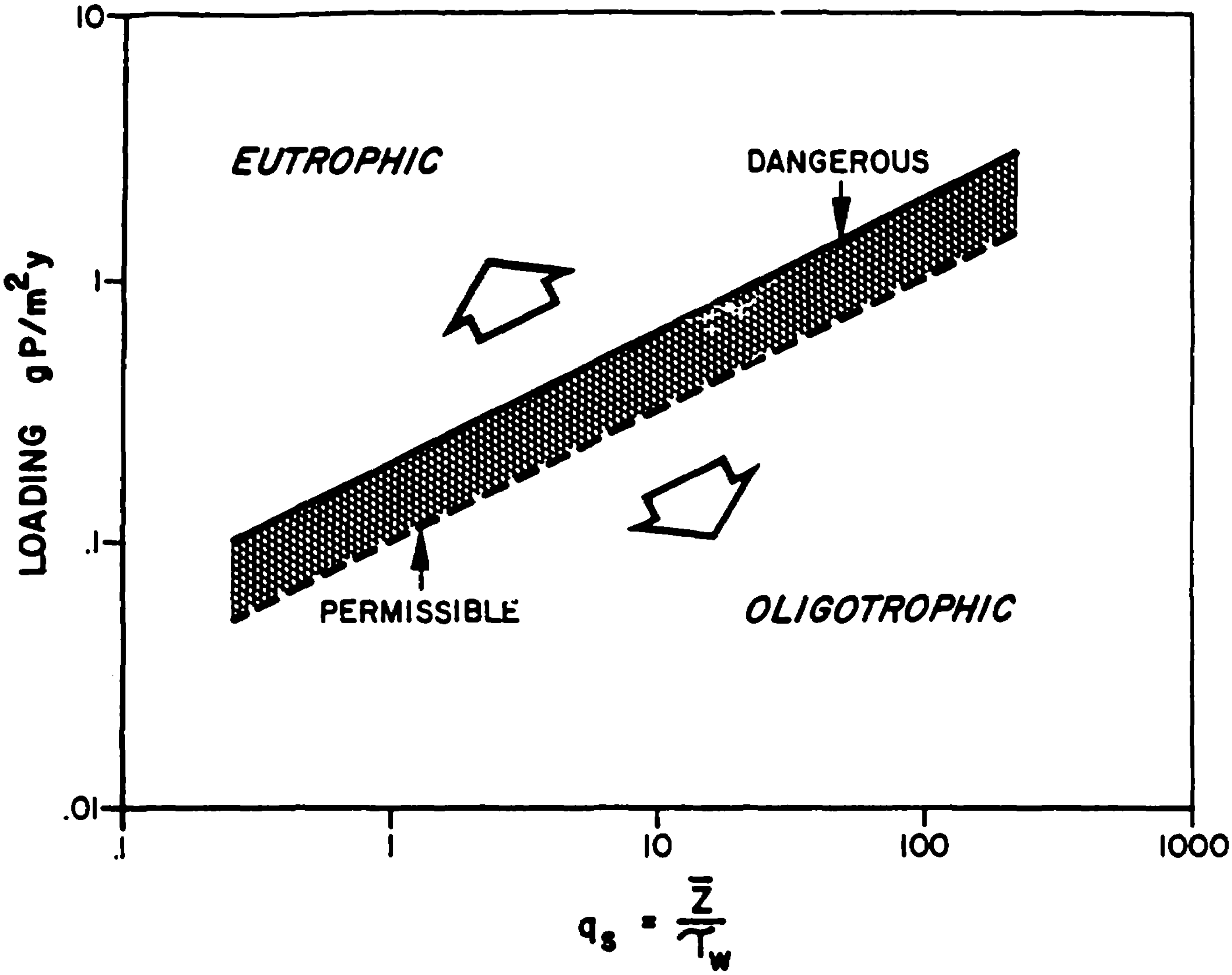

Figure II-12a. The Vollenweider Model (from Zison, et al., 1977).

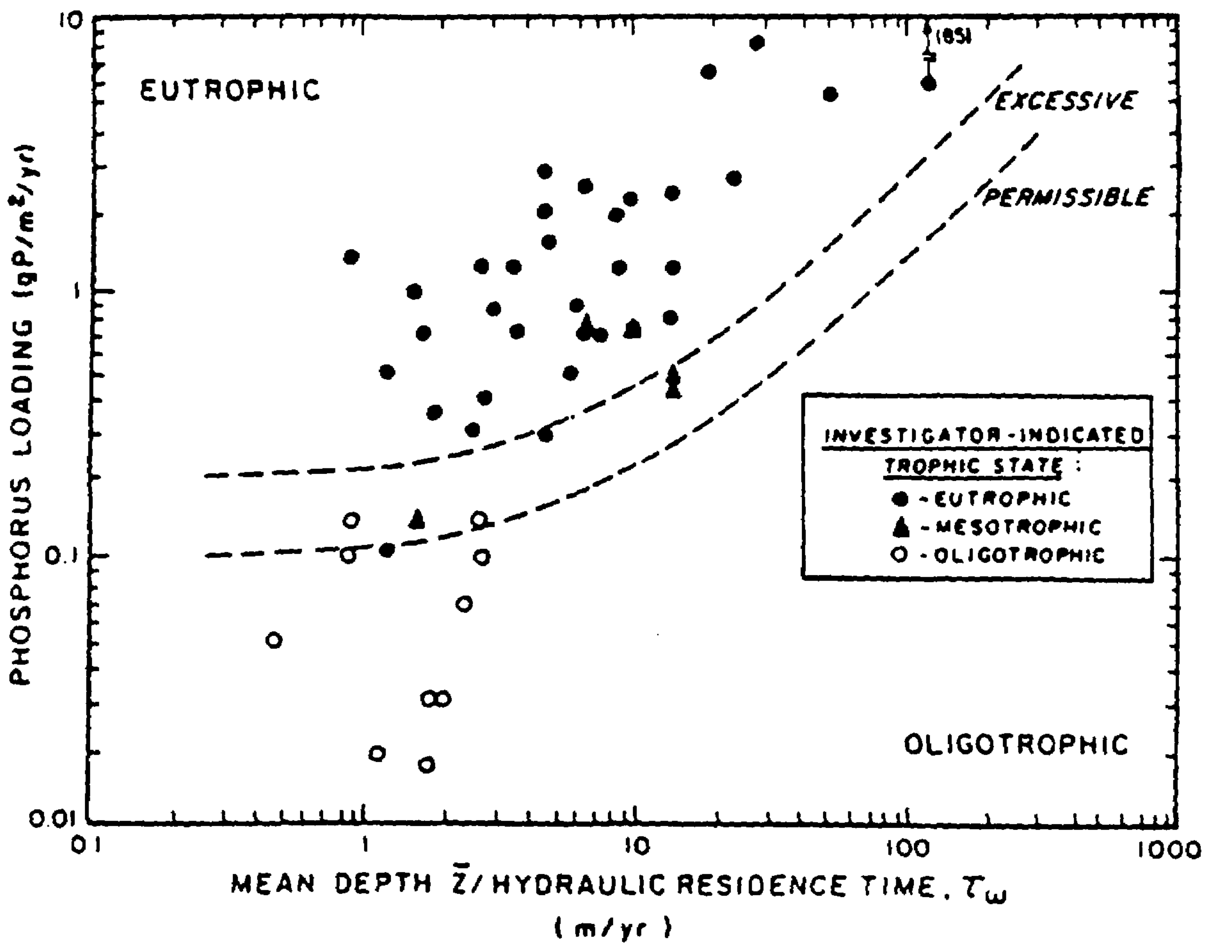

Figure II-12b. The Vollenweider-OECD Model (from Rast and Lee, 1978).

An example application of this type of approach is given by Zison, et al. (1977), where the characteristics of a reservoir are given as:

Bigger Reservoir

Available Data (all values are means):

Length	20 mi = 32.2 km
Width	10 mi = 16.1 km
Depth ($\bar{z}$)	200 ft = 61 m
Inflow (Q)	500 cfs
Total phosphorus concentration in inflow	0.8 ppm
Total nitrogen concentration in inflow	10.6 ppm

First determine whether phosphorus is likely to be growth limiting. Since data are available only for influent water, and since no additional data are available on impoundment water quality, N:P for influent water will be used.

$$N:P = 10.6/0.8 = 13.25$$

Thus, recalling that a N:P mass ratio of 7:1 is required for algal growth, Bigger Reservoir is probably phosphorus limited.

Compute the approximate surface area, volume and the hydraulic residence time.

$$\text{Volume } (V) = (20 \text{ mi})\ (10 \text{ mi})\ (200 \text{ ft})\ (5280 \text{ ft/mi})^2 =$$

$$1.12 \times 10^{12} \text{ft}^3 = 3.16 \times 10^{10} \text{m}^3$$

$$\text{Hydraulic residence time } (\tau_w) = V/Q =$$

$$1.12 \times 10^{12} \text{ft}^3/500 \text{ ft}^3\text{sec}^{-1} = 2.24 \times 10^9 \text{sec} = 71 \text{ yr}$$

$$\text{Surface area } (A) = (20 \text{ mi})\ (10 \text{ mi})\ (5280 \text{ ft/mi})^2 =$$

$$5.57 \times 10^9 \text{ft}^2 = 5.18 \times 10^8 \text{m}^2$$

Next, compute hydraulic loading, q_s

$$q_s = \bar{z}/\tau_w$$

$$q_s = 61 \text{ m}/71 \text{ yr} = 0.86 \text{ m yr}^{-1}$$

Compute annual inflow, Q_y

$$Q_y = (Q)\ (3.25 \times 10^7 \text{sec yr}^{-1})$$

$$Q_y = 1.58 \times 10^{10} \text{ft}^3 \text{ yr}^{-1}$$

Phosphorus concentration in the inflow is 0.8 ppm, or 0.8 mg/l. Loading (L_p) in grams per square meter per year is computed from the phosphorus concentration (C_p), the annual inflow (Q_y), and the surface area (A):

$$L_p = \frac{Q_y C_p}{A}$$

$$L_p = \frac{(1.58 \times 10^{10}\ ft^3/yr)(0.8\ mg\ P/l)}{(5.18 \times 10^8\ m^2)}(28.32\ l/ft^3)(1 \times 10^{-3}\ mg/g)$$

$$L_p = 0.70\ g/m^2\text{-yr}$$

Referring to the plot in Figure II-12a, we would expect that Bigger Reservoir, with L_p = 0.7 and q_s = 0.86, is eutrophic, possibly with severe summer algal blooms.

The Vollenweider type of approach has many useful and varied applications. For example, a phosphorus loading model was used to evaluate three prospective reservoir sites for eutrophication potential (Camp Dresser & McKee, 1983). Since this evaluation was part of a study to select a future dam site, and an impoundment did not exist, there was very little information available with which to work. While such an evaluation was not a use attainability study per se, the application is instructive because in many cases there may be virtually no data available for use in evaluating an existing lake or impoundment for attainable uses. For these cases where few historical data are available, use of a computer model would require simulation predictions without the benefit of a calibrated model, unless considerable resources are available to conduct a sampling program to characterize the water body from season to season in order to generate the data required by such a model. There are few options in this case other than use of an empirical model which, nevertheless, may provide very instructive results.

In the reservoir site study, phosphorus loading was estimated from water quality data for the streams that would feed each of the prospective reservoirs, and from an evaluation of land use practices in the watersheds. Streamflow data and an analysis of rainfall-runoff relationships provided an estimate of flow (Q) to each of the three reservoirs, and topographic maps were used to determine reservoir volume, average depth ($\bar{z}$), and surface area (A).

In the analyses, the quantity $\bar{z}/\tau_w$ may be calculated as:

$$\bar{z}/\tau_w = \bar{z}\rho = (V/A)(Q/V) = Q/A$$

where ρ, the flushing rate, is equal to the reciprocal of τ, the hydraulic residence time.

The quantity Q/A is the hydraulic loading rate--the amount of water added annually per unit area of lake surface. This may be interpreted to imply that lakes with the same hydraulic and phosphorus loadings should have the same in-lake phosphorus concentration regardless of differences in flushing rates (Uttormark and Hutchins, 1978).

The flushing rate is a very important characteristic of a lake, and is an important determinant of trophic state. If the flushing rate is high, as might be the case in a run-of-river impoundment, algal growth problems may be much less for a given phosphorus loading than for the same phosphorus loading to a lake with a low flushing rate. Although hydraulic loading serves as a surrogate for flushing rate in the Vollenweider model, the model still represents an important advancement beyond static loading estimations, such as were presented in Vollenweider in 1968 (Table II-3) where estimates for trophic state are based solely on mass loading.

Vollenweider-OECD Model

The Organization for Economic Cooperation and Development (OECD) Eutrophication Study was conducted in the early 1970's to quantify the relationship between the nutrient (phosphorus) load to a water body (lake, reservoir, or estuary) and the eutrophication-related water quality response of the water body to that load. Rast and Lee (1978) applied the Vollenweider (1975) model to the OECD water bodies in the United States. The results are plotted in Figure II-12b. It is apparent that the eutrophic water bodies are clustered in one area of the plot and the oligotrophic water bodies in another. Between those two zones, the authors delineated rough boundaries of permissible and excessive phosphorus loading with respect to eutrophication-related water quality. This model can be used in the same way as the Vollenweider model discussed previously.

Dillon and Rigler Model

In 1974, Dillon and Rigler (as reported by Uttormark and Hutchins) published an empirical model, similar to that of Vollenweider, in which a phosphorus retention coefficient (R) was proposed to account for phosphorus retention in the lake.

$$R = (P_{in} - P_{out})/P_{in} \tag{6}$$

Incorporation of R into the phosphorus mass balance equation leads to Equation 7 for the Dillon-Rigler model which is analogous to Equation 5 for the Vollenweider model.

$$[P] = L(1-R)/(\bar{z}\rho) \tag{7}$$

Dillon and Rigler used values of 10 and 20 mg-P/m^3 to define acceptable and excessive loading values to derive Figure II-13. Figure II-13 may be used to estimate trophic state by plotting the quantity:

$$L(1-R)/\rho \quad \text{vs.} \quad \bar{z}$$

where

L = annual phosphorus loading, g/m^2-yr
R = retention coefficient, $(P_{in} - P_{out})/P_{in}$
ρ = flushing rate = Q/V, yr^{-1}
$\bar{z}$ = mean depth, m

TABLE II-3

SPECIFIC NUTRIENT LOADING LEVELS FOR LAKES (EXPRESSED AS TOTAL NITROGEN AND TOTAL PHOSPHORUS IN g/m^2-yr)*

Mean Depth Up To:	Permissible Loading Up To:		Dangerous Loading in Excess of:	
	N	P	N	P
5 m	1.0	0.07	2.0	0.13
10 m	1.5	0.10	3.0	0.20
50 m	4.0	0.25	8.0	0.50
100 m	6.0	0.40	12.0	0.80
150 m	7.5	0.50	15.0	1.00
200 m	9.0	0.60	18.0	1.20

*from Vollenweider (1968)

SOURCE: Uttormark and Hutchins, 1978.

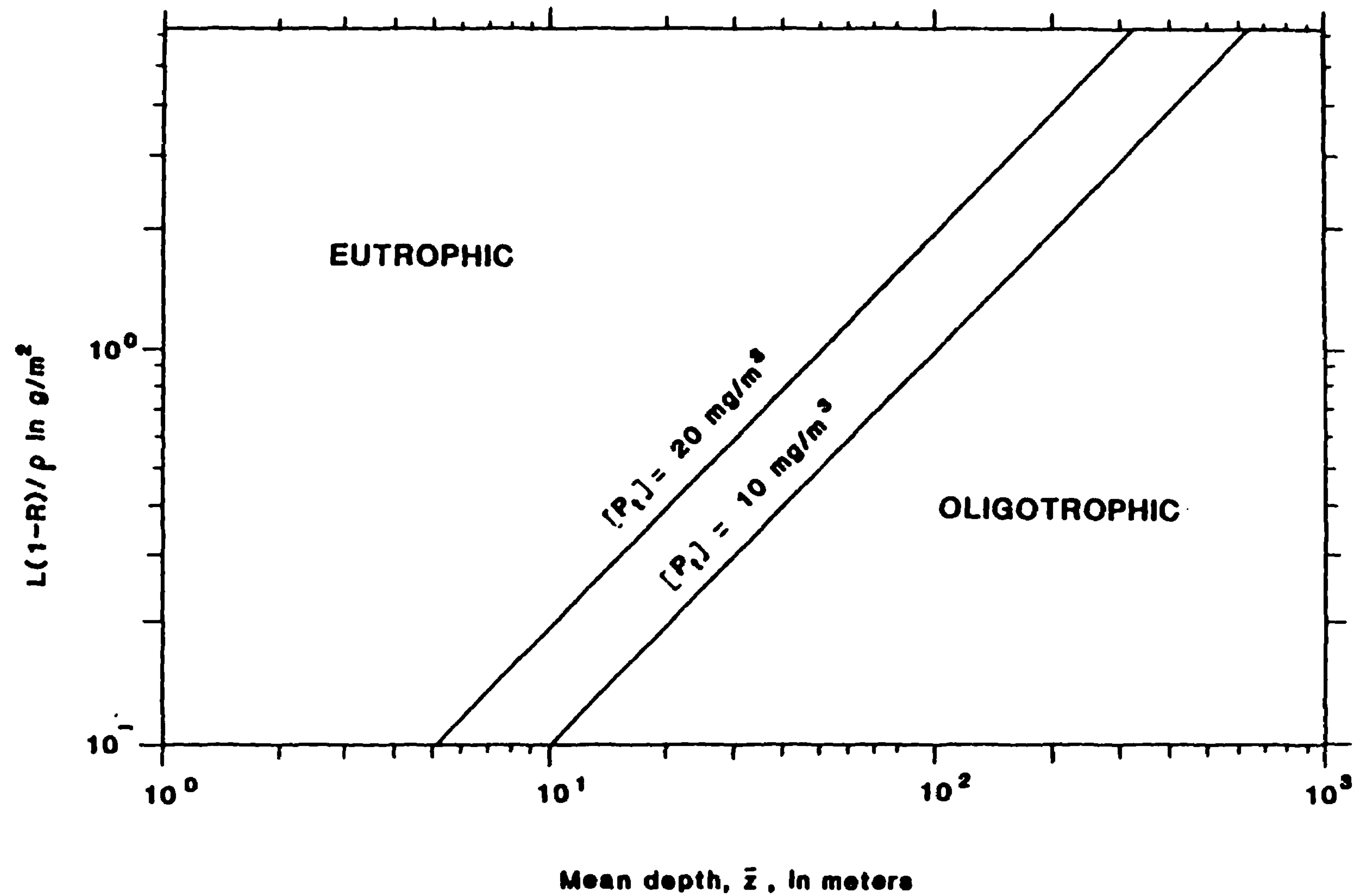

Figure II-13. The Dillon-Rigler Model (from Dillon and Rigler, 1974).

The lines of Figure II-13 represent equal predictive phosphorus concentrations, indicating that the prediction of the trophic state of a lake is based on a measure of the predictive phosphorus concentration in the lake rather than on the phosphorus loading (Tapp, 1978).

Larsen and Mercier Model

Larsen and Mercier (as reported in Tapp, 1978) used the phosphorus mass balance model to describe the relationship between the steady state lake and mean input phosphorus concentrations. Again using values of 10 and 20 mg/m^3 (ug/l), Larsen and Mercier developed the curves of Figure II-14 to distinguish oligotrophic, mesotrophic and eutrophic conditions. To use Figure II-14, one needs to estimate the mean influent lake phosphorus concentration, $\bar{P}$, in g/m^3, and R_{exp}, the fraction of phosphorus retained in the lake. The Larsen and Mercier formula plots mean tributary total phosphorus concentration against a phosphorus retention coefficient, thereby addressing the criticism of other models that no distinction is made between phosphorus increases due to influent flows or concentrations or both (Hern, et al., 1981). In effect, the Larsen and Mercier model predicts the mean tributary phosphorus concentration which would cause eutrophic or mesotrophic conditions.

In a comparative test of these three phosphorus loading models, using data collected under the National Eutrophication Survey on 23 water bodies (most in the northeastern and north central United States), it was found that the Dillon-Rigler and Larsen-Mercier models fit the data much better than the Vollenweider model (Tapp, 1978). This is probably because the Vollenweider model considers only total phosphorus loading without regard to in-lake processes that reduce the effective phosphorus concentration. In a similar comparison on data from southeastern water bodies, however, all three of the models generally fit the data.

Of the empirical models, the Vollenweider is the most conservative because it does not account for phosphorus in the outflow from a lake. This model should be used in a first level of analysis, in the absence of sufficient data to establish a phosphorus retention coefficient. If the retention coefficient can be derived, the Dillon-Rigler or Larsen-Mercier models would be preferable (Tapp, 1978).

Reckhow (1979) cautions that the application of empirical phosphorus lake models may not be appropriate for certain conditions or types of lakes. These include conditions of heavy aquatic weed growth, violation of model assumptions (for example, no outlet from a lake), or because the lake type (such as extremely shallow lakes) was not included in the data sets used to develop each of the models.

Sedimentation rates are apt to differ in a closed lake from sedimentation in a lake with an outlet. Based on a consideration of the phosphorus mass balance equation with the outflow term removed, and upon settling rates discussed by Dillon and Kirchner (1975) and Chapra (1977), Reckhow (1979) proposed the following expression for predicted phosphorus concentration:

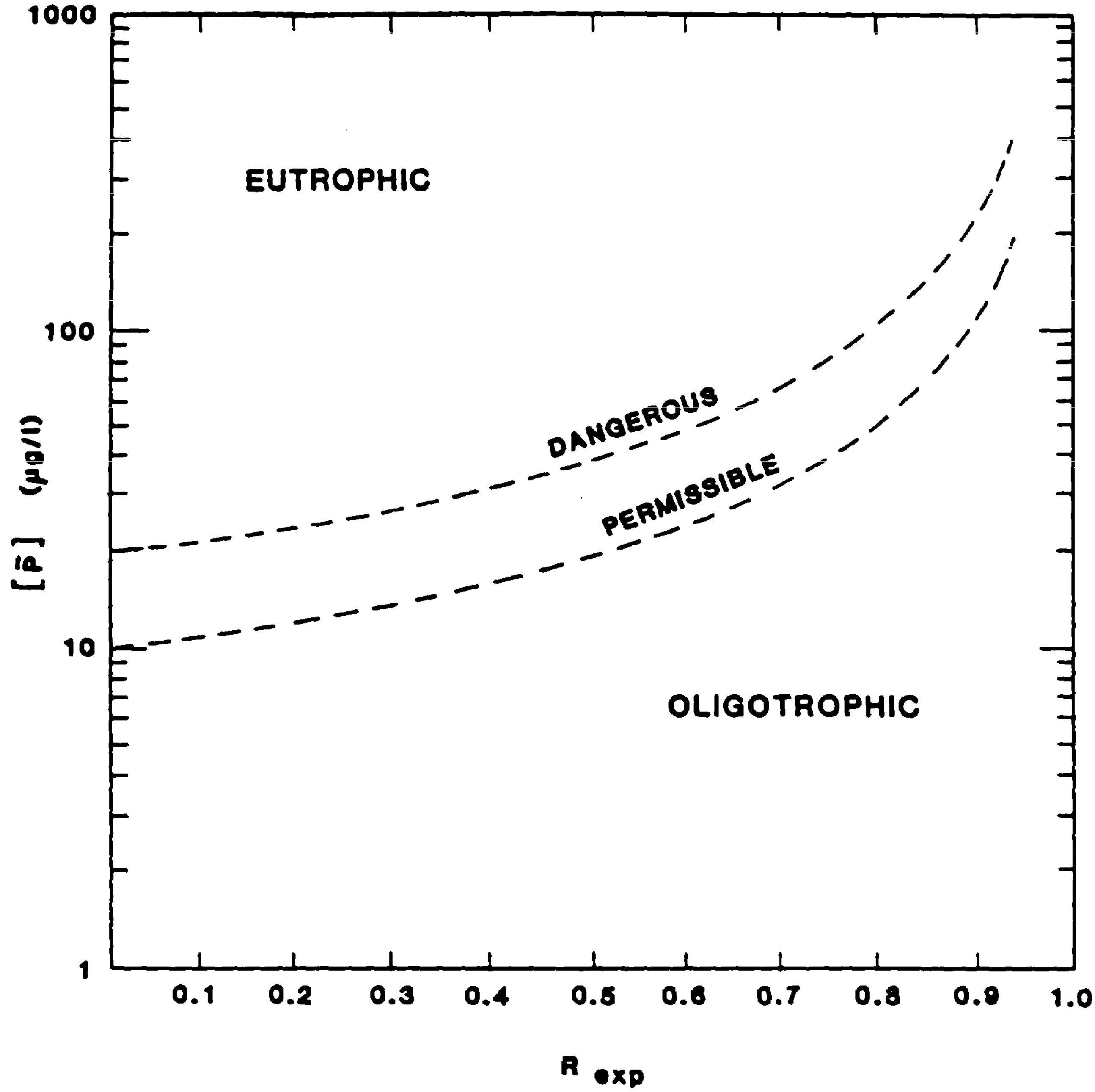

Figure II-14. The Larsen-Mercier Model (from Tapp, 1978).

$$L/(16 + \bar{z}\rho) < P_{true} < L/13.2 \qquad (8)$$

Shallow lakes present a problem because the potential for mixing of the sediments results in phosphorus concentrations that may be more variable than in deeper lakes. On the other hand, these same conditions may prevent the development of anaerobic conditions and serve to reduce concentration variability. Modeling of lakes with heavy weed growth is problematic because thick growths may restrict mixing, while interacting directly with the sediment.

Modified Larsen and Mercier Model

Hern, et al. (1981) note the assumption inherent to each of the phosphorus models discussed above that the relationship of phytoplankton biomass to phosphorus is the same for all lakes, yet point out that the utilization and incorporation of phosphorus into phytoplankton biomass varies significantly from lake to lake, depending on availability of light, supply of other nutrients, bioavailability of the various species of phosphorus, and a number of other factors. They go on to evaluate the factors affecting the relationship of phytoplankton biomass to phosphorus levels and show how the phosphorus models may be modified to base trophic state assessments on chlorophyll-a rather than phosphorus.

In their analysis of sampling data from a number of lakes, Hern et al. determined that the response ratio of chlorophyll-a (CHLA) to high summer phosphorus concentrations decreases as total phosphorus increases, in contrast to the findings of other authors (Vollenweider, Dillon, etc.) whose work is based on data collected in lakes that were free of major interferences. Hern, et al., indicate a belief that the reason most lakes do not reach maximum production of chlorophyll-a is because of interference factors. Factors which may prevent phytoplankton chlorophyll-a from achieving maximum theoretical concentrations based on ambient total phosphorus (TP) levels in a lake include:

1. Availability of light (for example, limitations due to turbidity or plankton self shading);
2. Limitation of growth by nutrients other than total phosphorus, e.g., nitrogen, carbon, silica, etc.;
3. Biological availability of the TP components;
4. Domination of the aquatic flora by vascular plants rather than phytoplankton;
5. Grazing by zooplankton;
6. Temperature;
7. Short hydraulic retention time; and
8. Presence of toxic substances.

The response ratio (RA) is defined as the amount of chlorophyll-a formed per unit of total phosphorus. A strong relationship between CHLA (a measure of phytoplankton biomass) and TP in lakes has been established by a number of authors, as discussed by Hern et al. (1981). A log-log transformation of the response ratio and total phosphorus concentration yields a straight line (Figure II-15) which provides a basis of comparison between the theoretical RA and the actual RA at a given phosphorus level. This relationship was used to modify the Larsen-Mercier model to accompish the following objectives:

1. Change the trophic classification based on an ambient TP level to one based on the biological manifestation of nutrients as measured by chlorophyll-a;

2. Determine the "critical" levels of TP which will result in an unacceptable level of CHLA concentration so that the level of TP can be manipulated to achieve the desired use of a given water body; and

3. Account for the unique characteristics of a lake or reservoir which affect the RA.

The Larsen and Mercier (1976) model predicts the mean tributary TP concentration which would cause eutrophic or mesotrophic conditions as follows:

$$TP_E = \frac{ETP}{1-R} \quad \text{or} \tag{9}$$

$$TP_M = \frac{MTP}{1-R} \tag{10}$$

where

TP_E = the minimum mean tributary TP concentration in ug/l which will cause a lake to be eutrophic at equilibrium,

TP_M = the minimum mean tributary TP concentration in ug/l which will cause a lake to be mesotrophic at equilibrium,

ETP = a constant equal to 20, which is the theoretical minimum ambient ug/l of TP in a lake resulting in eutrophic conditions and is the level which if not equaled or exceeded will result in meso- or oligotrophic conditions,

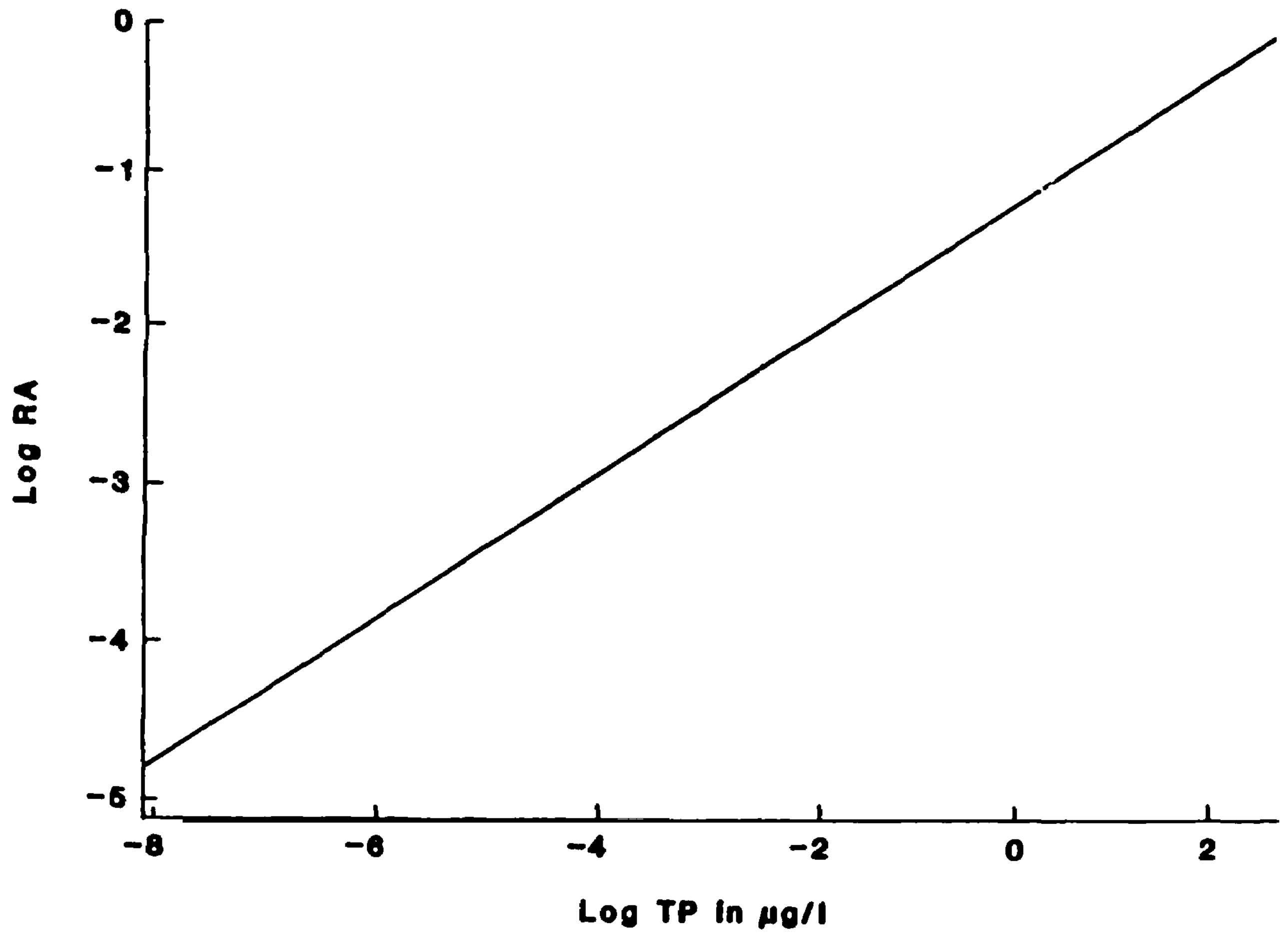

Figure II-15. The relationship between summer log RA and log TP based on Jones and Bachmann's (1976) regression equation (from Hern, et al., 1981).

MTP = a constant equal to 10, which is the theoretical minimum ambient ug/l of TP in a lake resulting in mesotrophic conditions and is the level which if not equaled or exceeded will result in oligotrophic conditions, and

R = fraction of phosphorus retained in the lake.

The Larsen and Mercier equations (i.e., Equations 9 and 10) can be corrected to account for the RA of a specific lake as follows:

$$\overline{TP}_{AE} = \frac{ETP(ERA/AERA)}{1-R} \quad (11)$$

$$\overline{TP}_{AM} = \frac{MTP(MRA/AMRA)}{1-R} \quad (12)$$

where

$\overline{TP}_{AE}$ = the minimum mean tributary TP concentrations in ug/l which will cause a lake to be eutrophic at equilibrium corrected to account for the lake's RA,

$\overline{TP}_{AM}$ = the minimum mean tributary TP concentrations in ug/l which will cause a lake to be mesotrophic at equilibrium corrected to account for the lake's RA,

ERA = a constant equal to 0.32 which is the RA predicted from 20 ug/l of ambient TP utilizing Jones and Bachmann's (1976) regression equation,

MRA = a constant equal to 0.23 which is the RA predicted from 10 ug/l of ambient TP utilizing Jones and Bachmann's (1976) regression equation,

AERA = the mean summer RA for the lake corrected to what it would be at the 20 ug/l level of TP, i.e., the ambient eutrophic level, and

AMRA = the mean summer RA for the lake corrected to what it would be at the 10 ug/l level of TP, i.e., the ambient mesotrophic level.

The ERA constant of 0.32 was determined from utilizing the ETP constant of 20 ug/l of ambient TP in the Jones and Bachmann (1976) regression equation:

$$\log \text{ug/l CHLA} = -1.09 + 1.46 \log \text{ug/l TP} \quad (13)$$

Substituting 20 ug/l for TP, log CHLA is equal to 0.81 and CHLA is equal to 6.4. Therefore, the ERA is equal to 6.4/20 or 0.32. Similarly, the MRA constant of 0.23 was determined utilizing the MTP constant of 10 ug/l of ambient TP.

The AERA is determined from the following equation:

$$\log \text{AERA} = \left[\frac{\log \text{ORA} - \text{A}}{\log \text{OTP} - \text{B}}\right] \left[\log \text{ETP} - \text{B}\right] + \text{A} \tag{14}$$

where

ORA = the observed summer ambient RA in the lake,

OTP = the observed summer ambient TP in the lake,

A = -4.77 which is the log of the RA determined from Equation 13 utilizing a TP concentration at approximately 0 (since log 0 is undefined, an extremely low TP concentration, i.e., 0.00000001 ug/l, was used to approximate 0 on the log scale), and

B = -8 which is the log of the TP (i.e., 0.00000001 ug/l, which is used to approximate 0 in Equation 13).

Substituting into Equation 14:

$$\log \text{AERA} = \left[\frac{\log \text{ORA} + 4.77}{\log \text{OTP} + 8}\right] \left[9.30\right] - 4.77 \tag{15}$$

The AMRA is determined from the following equation:

$$\log \text{AMRA} = \left[\frac{\log \text{ORA} - \text{A}}{\log \text{OTP} - \text{B}}\right] \left[\log \text{MTP} - \text{B}\right] + \text{A} \tag{16}$$

Substituting into Equation 16:

$$\log \text{AMRA} = \left[\frac{\log \text{ORA} + 4.77}{\log \text{OTP} + 8}\right] (9) - 4.77 \tag{17}$$

The constants used in Equations 14 and 16 are used to establish the slope of a line (Figure II-15) which begins at -4.77 (log RA) and -8 (log TP). Using the ORA and the OTP, the RA is adjusted using the relationship shown in Figure II-15, which was determined from the Jones and Bachmann (1976) regression equation (Equation 13) to one which would cause eutrophic (AERA) or mesotrophic conditions in the lake (AMRA).

A comparison of trophic state predictions using the Larsen and Mercier equations (Equations 9 and 10) with the modified equations to account for a lake's RA (Equations 11 and 12) was made using lake field data (Hern, et al., 1981). Those data showed that the lake had:

$$OTP = 36.3 \text{ ug/l},$$

$$\text{observed mean summer CHLA (OCHLA)} = 6.3 \text{ ug/l},$$

$$1\text{-}R = 0.71,$$

$$ORA = 0.17, \text{ and}$$

$$\text{observed mean tributary TP (OTTP)} = 57.3 \text{ ug/l}.$$

Substituting into Equation 9 (the Larsen-Mercier equation that yields the minimum mean tributary TP that will cause a lake to be eutrophic), we find:

$$\overline{TP}_E = \frac{20}{0.71} = 28.2 \text{ ug/l} \tag{9}$$

Since 28.2 ug/l of TP represents the theoretical minimum mean tributary concentration which will cause the lake to be eutrophic under steady state conditions and the OTTP is 57.3 ug/l, the use of Equation 9 would classify the lake as eutrophic. Substituting into Equation 11 which gives the mean tributary TP that will cause a lake to be eutrophic, when this TP is corrected for the lake's response ratio, RA:

$$\overline{TP}_{AE} = \frac{20(0.32/0.13)}{0.71} = 69.3 \text{ ug/l} \tag{11}$$

Since 69.3 ug/l is greater than 57.3 ug/l, we find if we use the modified equation which accounts for the lake's RA, the lake could be classified as mesotrophic and could possibly be oligotrophic. To determine whether it is mesotrophic or oligotrophic, we substitute into Equation 12 to determine the mean tributary TP, corrected for the lake's RA, that will support mesotrophic conditions.

$$\overline{TP}_{AM} = \frac{10(0.23/0.10)}{0.71} = 32.4 \text{ ug/l} \tag{12}$$

Since 32.4 ug/l is less than 57.3 ug/l, we would classify the lake as mesotrophic.

Computer Models

For many lakes, desktop evaluations and the analysis of field data may not be sufficient for an analysis of attainable uses. When a more sophisticated analysis is indicated, computer-based mathematical models can be used to simulate physical and water quality parameters, as well as various life forms and their interrelationships. The model predictions can be used to determine whether physical and water quality conditions are adequate for use attainment. For example, using the information on biological requi- ments presented later in this manual in conjunction with predicted wate quality conditions, judgments can be made regarding what type of aquatic life community a lake is likely to be capable of supporting. Computer models have the great advantage that they can predict the lake's ecological system rapidly under various design conditions and in addition, many computer models can simulate dynamic processes in the water body. In contrast, the phosphorus loading empirical models are suited only to steady state assumptions about the lake.

Which computer model to select will depend on the level of sophistication required in the analysis to be conducted. The selection will also depend highly on the size of the lake and its particular physical characteristics. For example, a long, narrow lake which is fully mixed horizontally and vertically can be modeled by a one-dimensional model. Two-dimensional models may be required where lake currents in a very large, shallow lake are the dominant factor affecting lake processes. In deep lakes where the vertical variations in lake conditions are most important, one-dimensional models in the vertical direction are appropriate.

In many cases lake water quality and ecological models have been developed to high degrees of sophistication, but these models do not provide the same degree of sophistication for the mechanisms that describe transport phenomena in the lake. On the other hand, models developed to simulate the hydrodynamics of a lake did not include the simulation of an extensive array of chemical and biological conditions. One of the major weaknesses in current water quality models as perceived by Shanahan and Harleman (1982) is the linkage of hydrodynamic and biochemical models.

Hydrodynamic Modeling

Shanahan and Harleman (1982) have described various types of models for lake circulation studies. They included two major groups: simplified models and true circulation models.

The simplified models included zero-dimensional models in which a lake is represented by a fully-mixed tank or continuous-flow stirred tank reactor. For a larger lake, representation with the zero-dimensional model is accomplished by treating different areas of the lake as separate fully mixed tanks. Simplified models also include longitudinal and vertical one-dimensional models. These models consider a series of vertical layers or horizontal segments.

True circulation models are those which employ two- and three-dimensional analysis. Two-dimensional models have been developed with a single or with multiple layers where it is assumed that the lake is vertically homogeneous within a layer. While lake circulation is modeled in each layer, the interactions between layers must be considered separately. The fully three-dimensional model, which also handles vertical transport between layers, is the most complex, and most expensive to set up and run. Although there are some examples of this type of model in use, Shanahan and Harleman believe that these models have not reached a point of practical application.

ce circulation models have been investigated in detail by of the Case Western Reserve University. In a report for the iental Protection Agency, Lick (1976b) describes his work on onal models. The three-dimensional models developed by Lick a steady-state, constant-density model; (2) a time-dependent, ity model; and (3) a time-dependent, variable-density model. veraged models are also presented which average the three-dimensional equations over the depth, thus reducing the model to a two-dimensional model.

Lake Water Quality Modeling

Many one-, two- or three-dimensional lake water quality models have been developed for various applications. As part of an EPA technical guidance manual for performing wasteload allocations (U.S. EPA, 1983c), available water quality models were reviewed. Information concerning model capability, model developers, and technical support were presented. Descriptions of lake models from Book IV - Lakes and Impoundments, Chapter 2 - Eutrophication (U.S. EPA, 1983c) are provided in Tables II-4 through II-8 to present an overview of some of the models that have been developed for lake studies.

Lake water quality models such as those described in Tables II-4 through II-8 generally are stand-alone models, however, some lake quality models have been linked to sophisticated hydrodynamic models. For example, in one special study for Lake Ontario, Chen and Smith (1979) developed a three-dimensional ecological-hydrodynamic model. The hydrodynamic model calculated currents and the temperature regime throughout the lake using a horizontal grid with eight layers of thickness. The water quality model included a coarser horizontal grid with seven layers. The hydrodynamic information was transferred through an interface program to the water quality model.

Much of the focus in water quality models developed for deep lakes and reservoirs has centered around the prediction of the thermal energy distribution, and has led to the development of one-dimensional ecological models such as LAKECO and WQRRS as described in Tables II-7 and II-8, respectively. This type of model is described in more detail in the following section.

One-Dimensional Lake Modeling

Development of LAKECO, WQRRS and other variations of these ecological models such as EPAECO (Gaume and Duke, 1975) began in the late sixties with studies on the prediction of thermal energy distribution (Water Resources Engineers, 1968, 1969). From some of their earlier work, Chen and Orlob (1972) developed a model of Ecological Simulations for Aquatic Environments which was used as the basis for many of the subsequent lake and reservoir models.

One-dimensional lake models assume that mass and energy transfers only occur along the vertical axis of a lake. To facilitate application of the necessary mass and energy balance equations, the lake is represented as a one-dimensional system of horizontal elements with uniform thickness, as

TABLE II-4

DESCRIPTION OF WATER ANALYSIS SIMULATION PROGRAM

Name of Model:	Water Analysis Simulation Program (WASP)* - LAKE1A, ERIE01, and LAKE3
Respondent:	William L. Richardson U.S. Environmental Protection Agency Large Lakes Research Station (LLRS) 9311 Groh Road Grosse Isle, Michigan 48138 (313) 226-7811
Developers:	Robert V. Thomann, Dominic DiToro, Manhattan College, N.Y.
Year Developed:	1975 (LAKE1) 1979 (LAKE3)
Capabilities:	Model is one (LAKE1) or three (LAKE3) dimensional and computes concentration of state variable in each completely mixed segment given input data for nutrient loadings, sunlight, temperature, boundary concentration, and transport coefficients. The kinetic structure includes linear and non-linear interactions between the following eight variables: phytoplankton chlorophyll, herbivorous zooplankton, carniverous zooplankton, non-living organic nitrogen (particulate plus dissolved), ammonia nitrogen, nitrate nitrogen, non-living organic phosphorus (particulate plus dissolved), and available phosphorus (usually orthophosphate). Also, a refined biochemical kinetic structure which incorporates two groups of phytoplankton, silica and revised recycle processes is available.
Availability:	Models are in the public domain and are available from Large Lakes Research Station.
Applicability:	The model is general, however, coefficients are site specific reflecting past studies.
Support:	User's Manual A user's manual titled "Water Analysis Simulation Program" (WASP) is available from Large Lakes Research Station. Technical Assistance Technical assistance would be provided if requested in writing through an EPA Program Office or Regional Office.

*The Advanced Ecosystem Model Program (AESOP) described next is a modified version of WASP.

SOURCE: U.S. EPA, 1983c.

TABLE II-5

DESCRIPTION OF WATER ANALYSIS SIMULATION PROGRAM
AND ADVANCED ECOSYSTEM MODELING PROGRAM

Name of Model:	Water Analysis Simulation Program (WASP) Advanced Ecosystem Modeling Program (AESOP)
Respondent:	John P. St. John HydroQual, Inc. 1 Lethbridge Plaza Mahwah, N.J. 07430 (201) 529-5151
Developers:	WASP Dominic M. DiToro, James J. Fitzpatrick, John L. Mancini, Donald J. O'Conner, Robert V. Thomann (Hydroscience, Inc.) (1970) AESOP Dominic DiToro, James J. Fitzpatrick, Robert V. Thomann (Hydroscience, Inc.) (1975)
Capabilities:	The Water Quality Analysis Simulation Program, WASP, may be applied to one-, two-, and three-dimensional water bodies, and models may be structured to include linear and non-linear kinetics. Depending upon the modeling framework the user formulates, the user may choose, via input options, to input constant or time variable transport and kinetic processes, as well as point and non-point waste discharges. The Model Verification Program, MVP, may be used as an indicator of "goodness of fit" or adequacy of the model as a representation of the real world. AESOP, a modified version of WASP, includes a steady state option and an improved transport component.
Verification:	To date WASP has been applied to over twenty water resource management problems. These applications have included one-, two-, and three-dimensional water bodies and a number of different physical, chemical and biological modeling frameworks, such as BOD-DO, eutrophication, and toxic substances. Applications include several of the Great Lakes, Potomac Estuary, Western Delta-Suisun Bay Area of San Francisco Bay, Upper Mississippi, and New York Harbor.
Availability:	WASP is in public domain and code is available from USEPA (Grosse Isle Laboratory and Athens Research Laboratory). AESOP is proprietary.
Applicability:	Models are general and may be applied to different types of water bodies and to a variety of water quality problems.

TABLE II-5

DESCRIPTION OF WATER ANALYSIS SIMULATION PROGRAM
AND ADVANCED ECOSYSTEM MODELING PROGRAM (Concluded)

Support:	User's Manual WASP and MVP documentation is available from USEPA (Grosse Isle Laboratory). AESOP documentation is available from HydroQual. Technical Assistance Technical asssistance of general nature from advisory to implementation (model set-up, running, calibration/ vertification, and analysis) available on contractural basis.

SOURCE: U.S. EPA, 1983c.

TABLE II-6

DESCRIPTION OF CLEAN PROGRAMS

Name of Model:	CLEAN, CLEANER, MS. CLEANER, MINI. CLEANER
Respondent:	Richard A. Park Center for Ecological Modeling Rensselaer Polytechnic Institute MRC-202, Troy, N.Y. 12181 (518) 270-6494
Developers:	Park, O'Neill, Bloomfield, Shugart, et al. Eastern Deciduous Forest Biome International Biological Program (RPI, ORNL, and University of Wisconsin)
Supporting Agency:	Thomas O. Barnwell, Jr. Technology Development and Application Branch Environmental Research Laboratory Environmental Protection Agency Athens, Georgia 30605
Year Developed:	1973 (CLEAN) 1977 (CLEANER) 1980 (MS. CLEANER) 1981 - estimated completion date for MINI. CLEANER
Capabilities:	The MINI. CLEANER package represents a complete restructuring of the Multi-Segment Comprehensive Lake Ecosystem Analyzer for Environmental Resources (MS. CLEANER) in order for it to run in a memory space of 22K bytes. The package includes a series of simulations to represent a variety of distinct environments, such as well mixed hypereutrophic lakes, stratified reservoirs, fish ponds and alpine lakes. MINI. CLEANER has been designed for optimal user application--a turnkey system that can be used by the most inexperienced environmental technician, yet can provide the full range of interactive editing and output manipulation desired by the experienced professional. Up to 32 state variables can be represented in as many as 12 ecosystem segments simultaneously. State variables include 4 phytoplankton groups, with or without surplus intracellular nitrogen and phosphorus; 5 zooplankton groups; and 2 oxygen, and dissolved carbon dioxide. The model has a full set of readily understood commands and a machine-independent, free-format editor for efficient usage. Perturbation and sensitivity analysis can be performed easily. The model has been calibrated and is being validated. Typical output is provided for

TABLE II-6

DESCRIPTION OF CLEAN PROGRAMS (Concluded)

	a set of test data. File and overlay structures are described for implementation on virtually any computer with at least 22K bytes of available memory.
Verification:	The MINI. CLEANER model is being verified with data from DeGray Lake, Arkansas; Coralville Reservoir, Iowa; Slapy Reservoir, Czechoslovakia; Ovre Heimdalsvatn, Norway; Vorderer Finstertak See, Austria; Lake Balaton, Hungary; and Lago Mergozzo, Italy. The phytoplankton/zooplankton submodels were validated for Vorderer Finstertaler See.
Availability:	Models are in public domain and code is available from Richard A. Park (RPI) and Thomas O. Barnwell (EPA/Athens).
Applicability:	Model is general.
Support:	User's Manual A user's manual for MS. CLEANER is available from Thomas O. Barnwell, Jr. A user's manual for MINI. CLEANER is in preparation. Technical Assistance Assistance may be available from the Athens Laboratory; code and initial support is available for a nominal service charge from RPI; additional assistance is negotiable.

SOURCE: U.S. EPA, 1983c.

TABLE II-7

DESCRIPTION OF LAKECO AND ONTARIO MODELS

Name of Model:	LAKECO*, ONTARIO
Respondent:	Carl W. Chen
Developers:	Carl W. Chen Tetra Tech Inc. 3746 Mount Diablo Blvd., Suite 300 Lafayette, California 94596 (415) 283-3771 (Original version developed when Dr. Chen was with Water Resources Engineers)
User Developed:	1970 (original version)
Capabilities:	LAKECO Model is one-dimensional (assumes lake is horizontally homogeneous) and calculates temperature, dissolved oxygen, and nutrient profiles with daily time step for several years. Four algal species, four zooplankton species, and three fish types are represented. The model evaluates the consequences of wasteload reduction, sediment removal, and reaeration as remedial measures. ONTARIO Same as above but in three-dimensions for application to Great Lakes.
Verification:	The models have been applied to more than 15 lakes by Dr. Chen and to numerous other lakes by other investigators.
Availability:	The model is in the public domain and the code is available from the Corps of Engineers (Hydrologic Engineering Center), EPA and NOAA.
Applicability:	General
Support:	User's Manual User's manuals are available from Tetra Tech, Corps of Engineers, EPA and NOAA. Technical Assistance Technical assistance is available and would be negotiated on a case-by-case basis.

*A version of LAKECO, contained in a model referred to as Water Quality for River Reservoir Systems (WQRSS) and supported by the Corps of Engineers (Hydrologic Engineering Center), is described separately.

SOURCE: U.S. EPA, 1983c.

TABLE II-8

DESCRIPTION OF WATER QUALITY FOR RIVER RESERVOIR SYSTEMS

Name of Model:	Water Quality for River Reservoir Systems (WQRRS)
Respondent:	Mr. R.G. Willey Corps of Engineers 609 Second Street Davis, California 95616 (916) 440-3292
Developers:	Carl W. Chen, G.T. Orlob, W. Norton, D. Smith Water Resources Engineers, Inc.
History:	1970 (original version of lake eutrophication model) 1978 (initial version of WQRRS package) 1980 (updated version of WQRRS)
Capabilities:	See description of LAKECO in Table II-7 (model also can consider river flow and water quality).
Verification:	Chattahoochee River (Chattahoochee River Water Quality Analysis, April 1978, Hydrologic Engineering Center Project Report)
Availability:	Model is in public domain and code is available from Corps.
Applicability:	Model is general.
Support:	User's Manual A user's manual is available from Corps. Technical Assistance Advisory assistance is available to all users. Actual execution assistance is available to federal agencies through an inter-agency funding agreement.

SOURCE: U.S. EPA, 1983c.

shown in Figure II-16. Each hydraulic element is treated as a continuous-flow stirred tank reactor (CFSTR) with completely uniform properties.

The implicit assumption of this geometric structuring of the problem is that mass concentration and thermal gradients in the horizontal plane are insignificant in determining the ecological responses and thermal behavior of the impoundment along the vertical axis. Therefore, simulated results are interpreted as being average conditions across the lake at a particular elevation.

These models solve a set of equations representing the water quality of a lake and the interactions of the lake biota with water quality. In reality, an aquatic ecosystem exhibits a delicate balance of a multiplicity of different aquatic organisms and water quality constituents. Of necessity, lake ecological models account only for the more significant interactions in this balance.

An aquatic ecosystem is comprised of water, its chemical impurities, and various life forms: bacteria, algae, zooplankton, benthos and fish, among others. The biota responds to nutrients and to other environmental conditions that affect growth, respiration, recruitment, decay, mortality and predation. Abiotic substances derived from air, soil, tributary waters and the activities of man, are inputs to the system that exert an influence on the biotic structure of the lake. Figure II-17 provides a conceptual representation of an aquatic ecosystem.

The fundamental building blocks (nutrients) for all living organisms are the same: carbon, nitrogen and phosphorous. With solar radiation as the energy source, these inorganic nutrients are transformed into complex organic materials by photosynthetic organisms. The organic products of photosynthesis serve as food sources for aquatic animals. It is evident that a natural succession up the food chain occurs whereby inorganic nutrients are transformed to biomass.

Biological activities generate wastes which include dead cell material and excreta which initially are suspended but may settle to the bottom to become part of the sediment. The organic fraction of the bottom sediment decays with an attendant release of the original abiotic substances. These transformations are integral parts of the carbon, nitrogen and phosphorous cycles and result in a natural "recycling" of nutrients within an aquatic ecosystem.

The water quality and biological productivity of a lake vary in both time and space. Temporal variations are associated with a wide variety of external influences on a lake. Examples of these influences are atmospheric energy exchanges, tributary contributions and lake outflows.

Spatial variations occur both in the horizontal plane and with depth. Variations in the horizontal plane are normally due to local conditions, such as distance from shoreline, depth of water and circulation patterns. Many times these variations do not affect the overall ecological balance of a lake and are not modeled by the one-dimensional lake model.

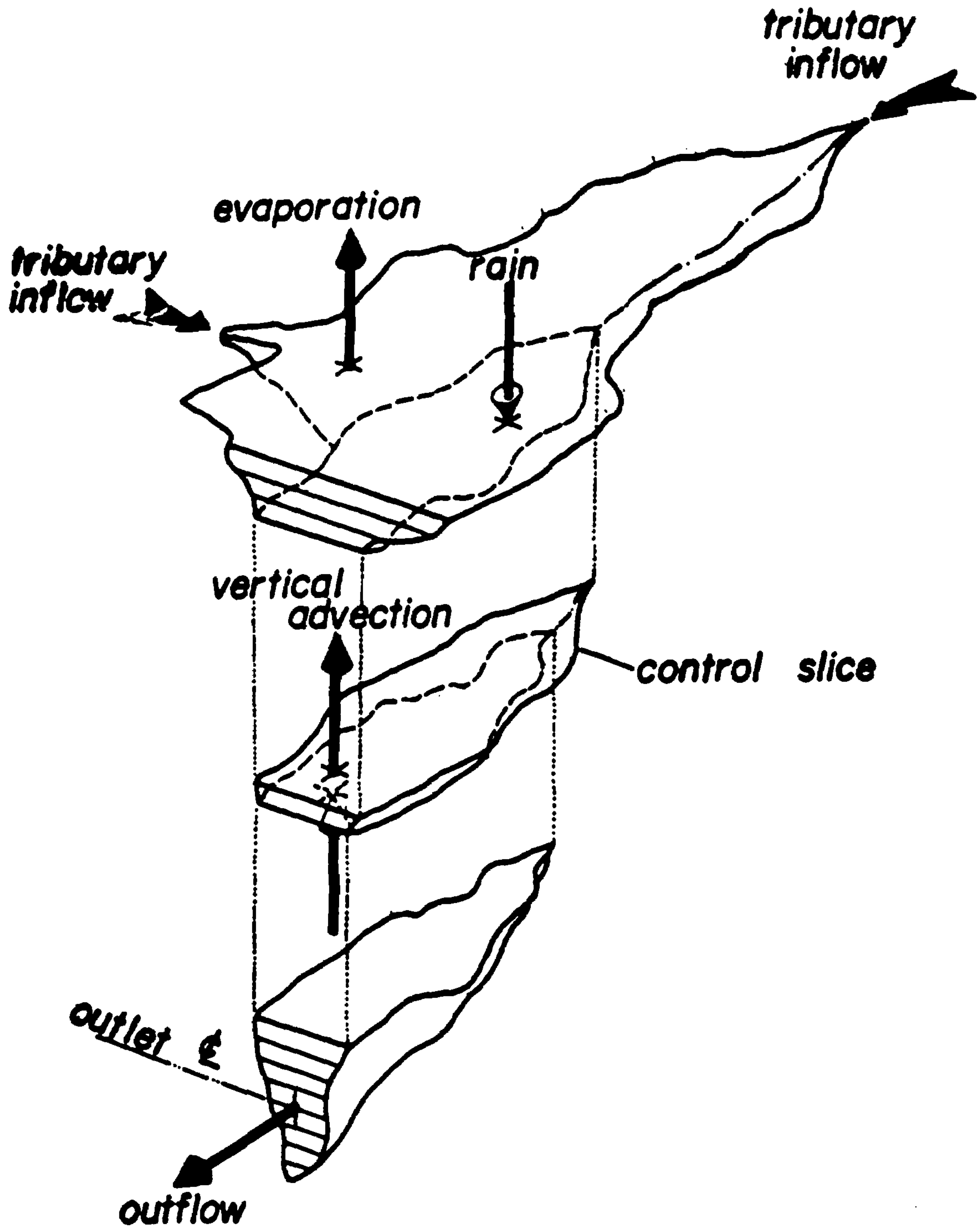

Figure II-16. Geometric Representation of a Stratified Lake
(from Gaume and Duke, 1975).

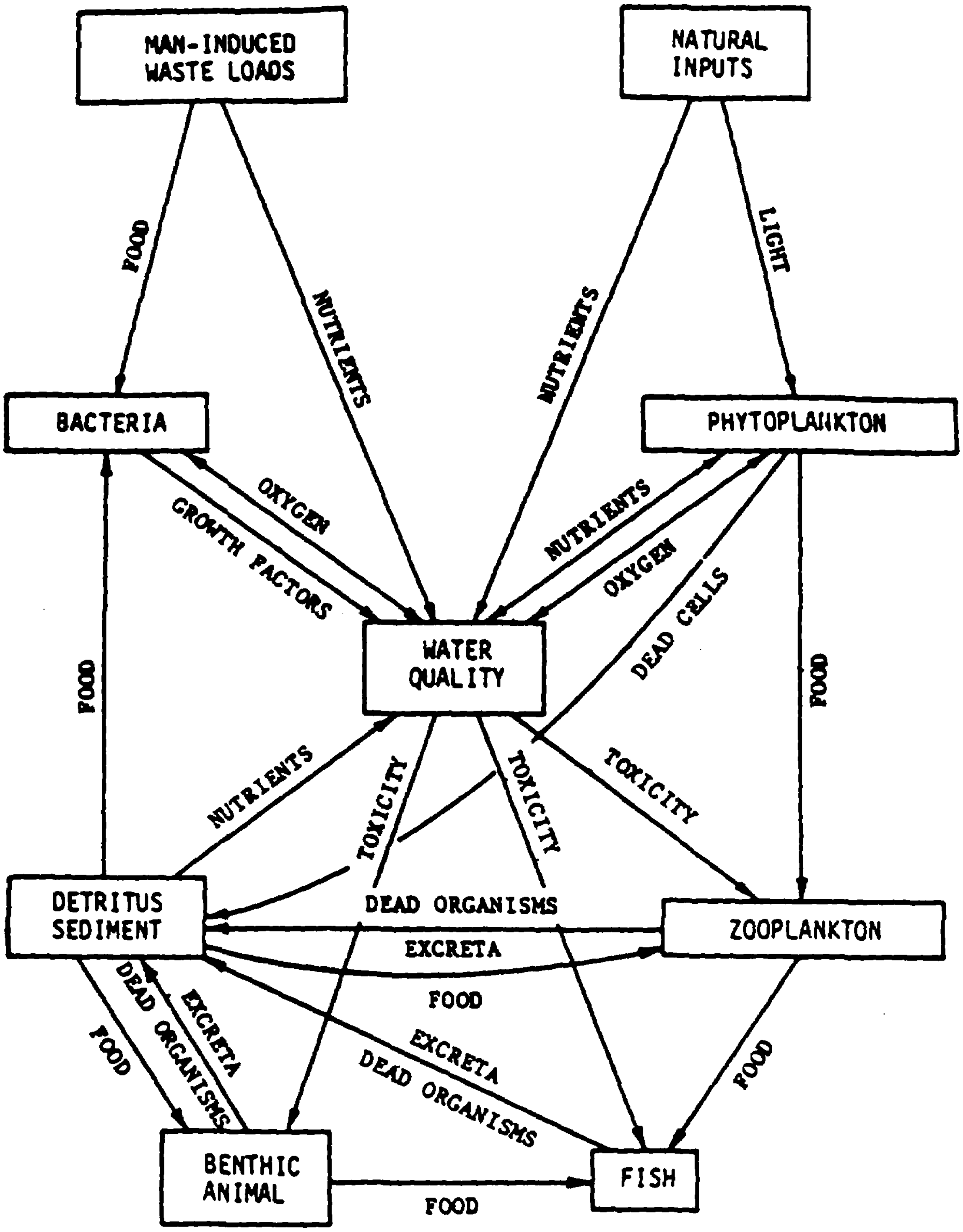

Figure II-17. Conceptual Model of an Aquatic Ecosystem (from Chen and Orlob, 1972).

Variations of water quality along the vertical axis of a lake have a more general effect. The hydrodynamic behavior of a well-stratified lake is density-dependent and, therefore, is related closely to the vertical temperature structure of the impoundment. The vertical temperature structure, in turn, is governed by the same external environmental factors as the temporal variations, i.e., atmospheric energy exchanges, tributary contributions and lake outflows.

EPA Center for Water Quality Modeling

The Center for Water Quality Modeling, located at the Environmental Research Laboratory in Athens, Georgia, has long been involved in the development and application of mathematical models that predict the transport and fate of water contaminants. The Center provides a central file and distribution point for computer programs and documentation for selected water quality and pollutant loading models. In addition, the Center sponsors workshops and seminars that provide both generalized training in the use of models and specific instruction in the application of individual simulation techniques.

The water quality model supported by U.S. EPA for well-mixed lakes is the Stream Water Quality Model QUAL-II (Roesner, et al., 1981). The model assumes that the major transport mechanisms--advection and dispersion--are significant only along the main direction of flow (longitudinal axis of the lake). It allows for multiple waste discharges, withdrawals, tributary flows, and incremental inflow. Hydraulically, QUAL-II is limited to the simulation of time periods during which the flows through the lake are essentially constant. Input waste loads must also be held constant over time. QUAL-II can be operated as a steady-state model or a dynamic model. Dynamic operation makes it possible to study water quality (primarily dissolved oxygen and temperature) as it is affected by diurnal variations in meteorological data.

The Army Corps of Engineers have developed a numerical one-dimensional model (CE-QUAL-R1), of reservoir water quality (U.S. Army Corps of Engineers, 1982). The reservoir model is a direct descendant of the reservoir portion of a model called "Water Quality for River-Reservoir Systems" (WQRRS) which was assembled for the Hydrologic Engineering Center of the Corps of Engineers by Water Resources Engineers, Inc. (Camp Dresser & McKee). The definitive origin of WQRRS was the work of Chen and Orlob (1972).

The aquatic ecosystem and geometric representation of this model are similar to those discussed in the previous section on one-dimensional lake modeling. A summary of the model capabilities of CE-QUAL-R1 is given in Table II-9.

Example Application of Mathematical Modeling

Mathematical modeling of natural phenomena allows planners, engineers, biologists, and the general public to see the effects on the lake system of changes in the environment which are planned or predicted to occur in the future. This insight allows a state to assess the environmental responses

TABLE II-9

CE-QUAL-R1 MODEL CAPABILITIES

Factors considered by CE-QUAL-R1 include the following:

a. Physical Factors

(1) Shortwave and longwave solar radiation at the water surface.

(2) Net heat transfer across the air-water interface.

(3) Convective and radiative heat transfer within the water body.

(4) Convective mixing due to density instabilities.

(5) Placement of inflowing waters at depths with comparable density.

(6) Withdrawal of outflowing waters from depths influenced by the outlet structure and density stratification.

(7) Conservative substance routing.

(8) Suspended solids routing and settling.

b. Chemical and Biological Factors

(1) Accumulation, dispersion, and depletion of dissolved oxygen through aeration, photosynthesis, respiration, and organic demand.

(2) Uptake-excretion kinetics and regeneration of nitrogen and phosphorus and nitrification processes under aerobic conditions.

(3) Carbon cycling and dynamics and alkalinity-pH-CO_2 interactions.

(4) Phytoplankton dynamics and trophic relationships.

(5) Transfers through higher trophic levels of the food chain.

(6) Accumulation, dispersion, and decomposition of detritus and sediment.

(7) Coliform bacteria die-off.

(8) Accumulation, dispersion, and reoxidation of manganese, iron, and sulfide when anaerobic conditions prevail.

SOURCE: U.S. Army Corps of Engineers, 1982.

of the lake and help it to analyze alternative plans for protecting the present use or determining what uses could be attained.

External factors, such as increased nutrients which accelerate the growth of algae, may destroy the delicate balance of nature, and cause considerable harm to the lake and its biology. Therefore, it is important to be able to predict what the lake response will be to external factors without actually imposing those conditions on it. The mathematical portrayal of the lake ecosystem by the computer model helps us toward that end.

As an example, the lake ecological model EPAECO (Gaume and Duke, 1975) provided a tool to mathematically represent the aquatic ecological system in the Fort Loudoun Lake, Tennessee. This study was conducted as part of the 208 plan for the Knoxville/Knox County Metropolitan Planning Commission (Hall, et al., 1976). The 208 study area map is shown in Figure II-18. In general, the model EPAECO is designed to simulate the vertical distribution of the following constituents over an annual cycle:

1. Temperature
2. Total Dissolved Solids
3. Alkalinity
4. Coliforms
5. Carbonaceous Biochemical Oxygen Demand (CBOD)
6. Ammonia Nitrogen
7. Nitrite Nitrogen
8. Nitrate Nitrogen
9. Phosphorus
10. Total Inorganic Carbon
11. Carbon Dioxide
12. Hydrogen Ion (pH)
13. Dissolved Oxygen
14. Algae (two classes)
15. Zooplankton
16. Fish (three classes)
17. Benthic Animals
18. Organic Sediment, and
19. Suspended Detritus.

The general approach to use of the mathematical model EPAECO is to obtain data which describe the geometric properties of the lake and its past history of water quality and hydrodynamics. Data on water quantity and quality of tributary inputs to the lake (streams and/or waste loads) and meteorological data are also necessary. Initially, the lake must be described as a mathematical system of depths, areas, volumes, tributary inputs and releases. A site-specific model must be developed which properly describes the environmental community and its interactions for Fort Loudoun Lake. This is done by a procedure called calibration. A calibrated model gives the user greater confidence that the simulation model will react as would the lake itself to changes in external factors such as increased tributary nutrient concentrations.

Examples of calibration results are shown in Figures II-19 through II-21. Figure II-19 presents the observed and simulated reservoir elevations for the year 1971; Figure II-20 shows the vertical temperature profiles, observed and simulated, for the months of April, May and July, 1971; and Figure II-21 gives the observed and simulated profiles for several water quality constituents for a single day in September 1971.

One of the main considerations in the study of Fort Loudoun Lake was an evaluation of present and future trophic states. Lakes which become enriched with excessive nutrients may be defined as eutrophic. Eutrophication produces large algal communities which affect the taste and odor of the lake's waters. Bacteria which degrade the large amounts of dead

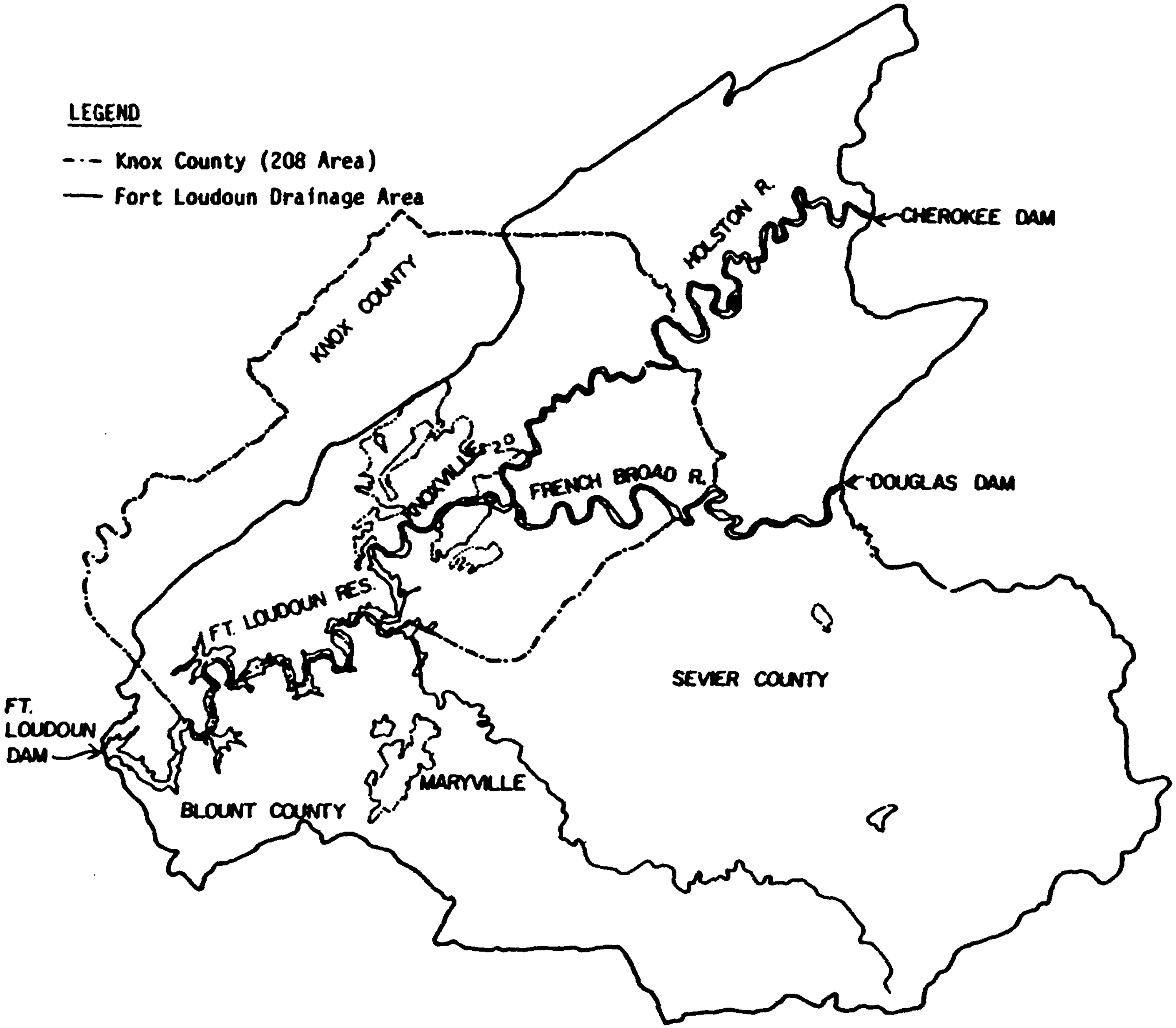

Figure II-18. 208 Study A[illegible] from Hall et al, 1976)

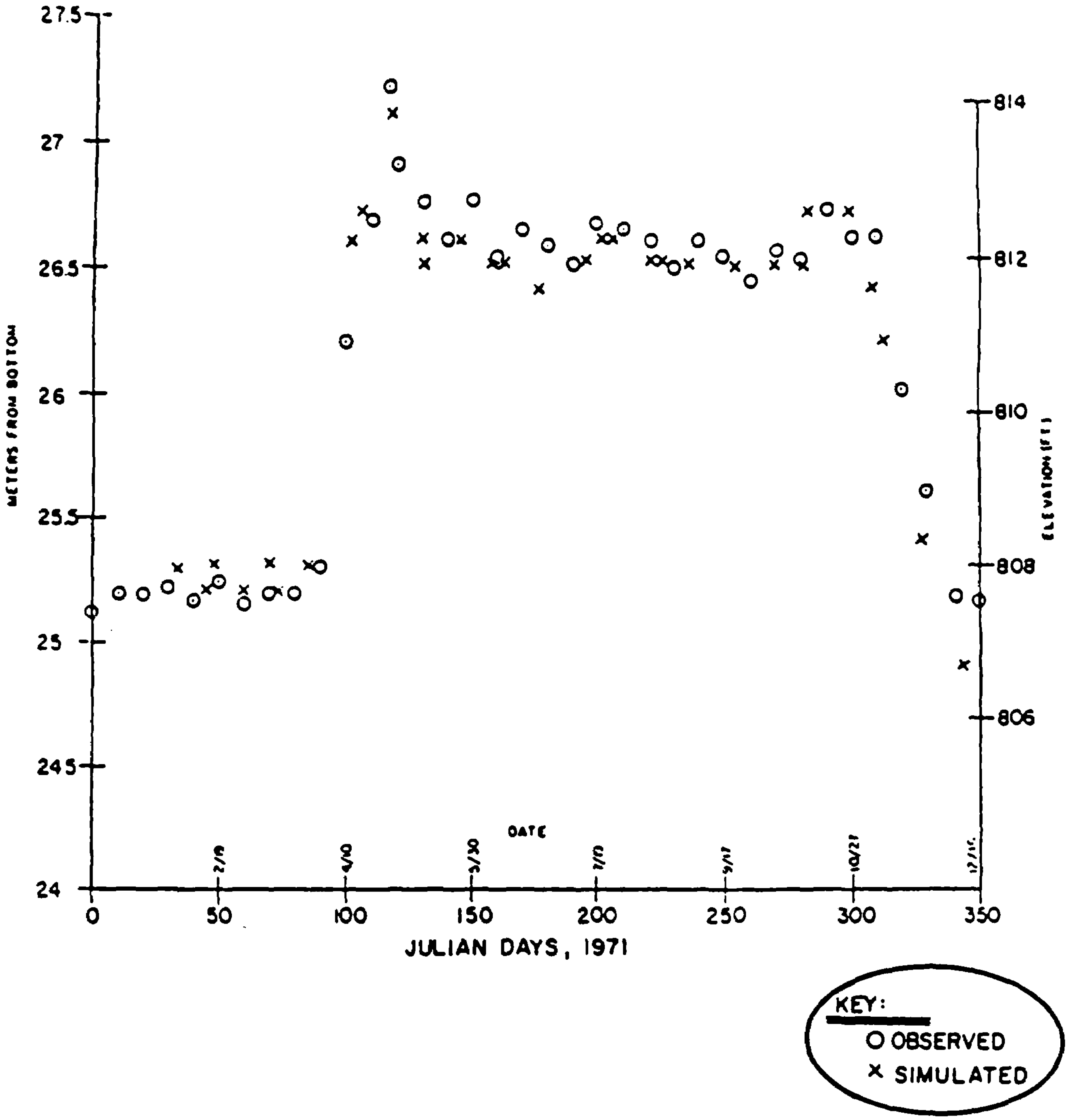

Figure II-19. Fort Loudoun Reservoir Elevations 1971 Observed vs. Simulated (from Hall et al, 1976)

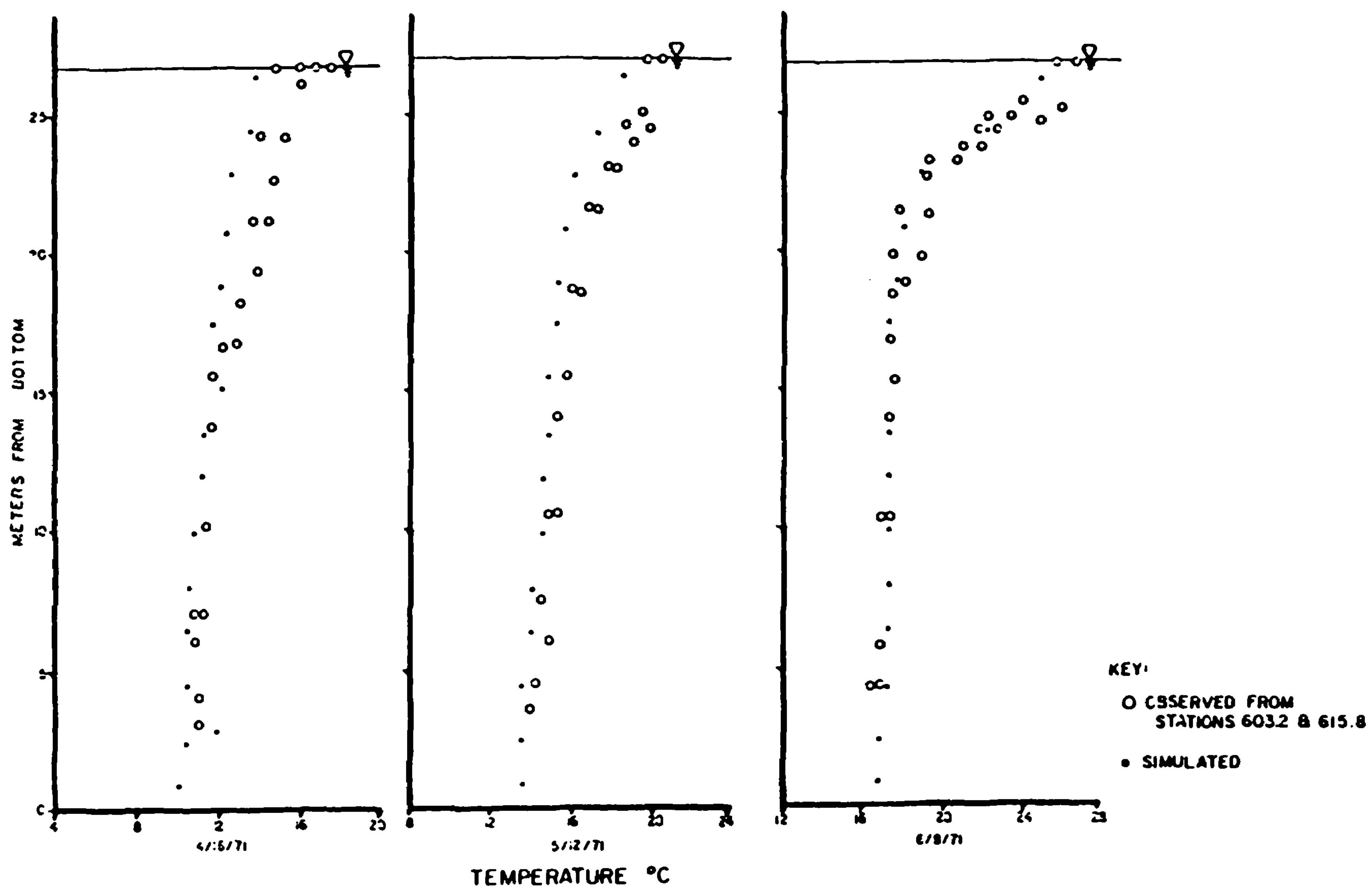

Figure II- 20 Temperature Profile for ~t Loudoun (from Hall et al, 1976)

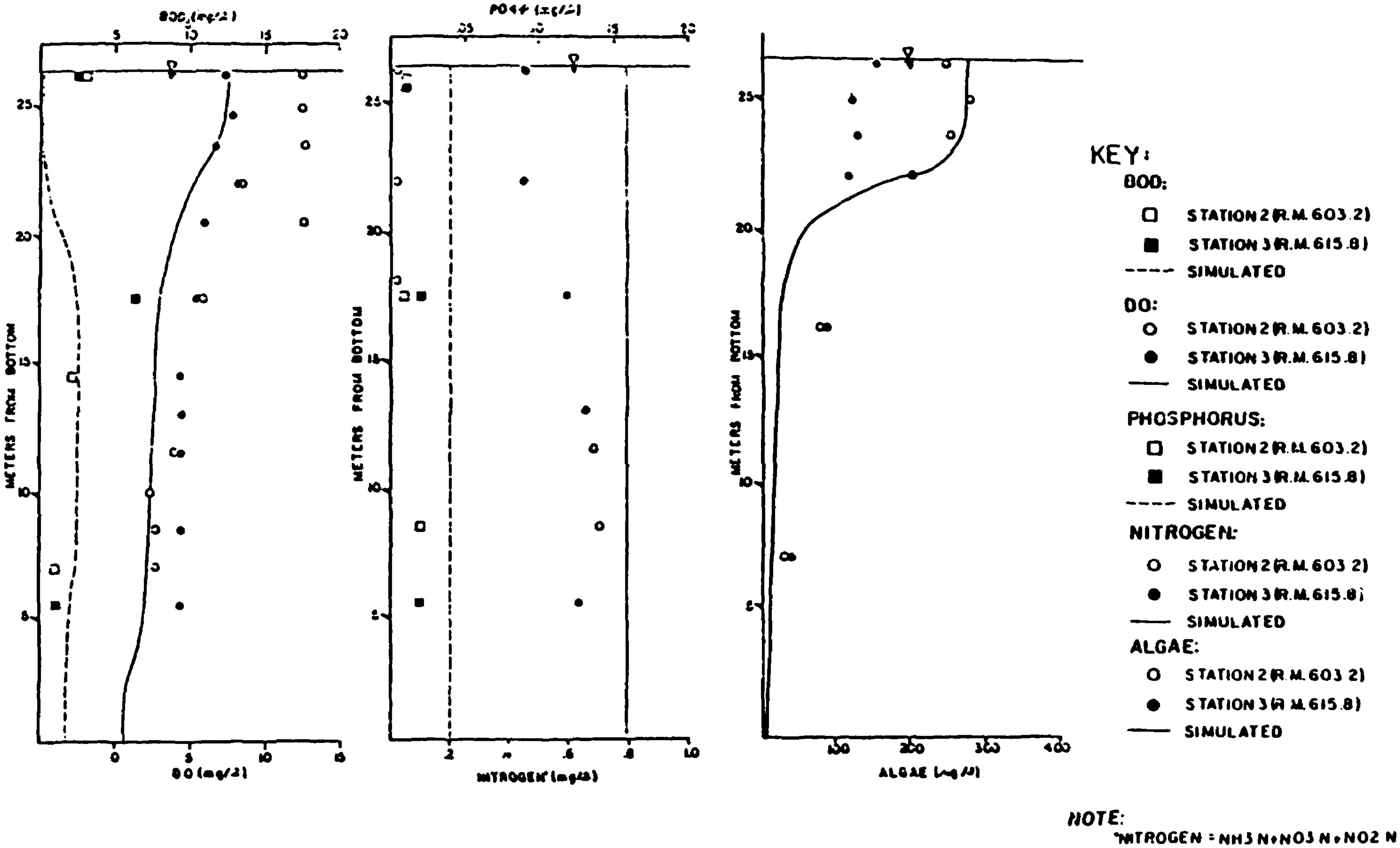

Figure II-21. DO and BOD_5, Inorganic Phosphorus and Nitrogen, and Algae September 10, 1971 Fort Loudoun (from Hall et al, 1976) organic matter in the lake deplete the oxygen supply, which in turn results in a loss of some types of fish. Excessive aquatic weed growth is also detrimental to swimming, boating and fishing.

The model EPAECO was used to assess algal growth as a result of various nutrient loads (high, medium and low) to the lake during the period of May through September. This type of model application not only quantified the degree of expected algal growth as a function of the availability of nutrients but also predicted the algal population and total lake ecology for future nutrient loads to the lake.

Since phosphorus was the limiting nutrient for algal growth in this lake study, the total available phosphorus was compared to the maximum seasonal algal concentrations simulated for the sensitivity study. Figure II-22 shows this comparison. The curve is derived from the maximum algal concentrations resulting from the following sensitivity conditions: high P, medium P, and low P. This curve represents the maximum algal concentrations reached by a constant inflow concentration of phosphorus during the algal growing season.

A limited amount of phosphorus is required in the inflows to the stratified portion of the reservoir to support a desirable algal community without producing excess growth and thus undesirable conditions. As shown on the graph in Figure II-22, Fort Loudoun Lake phosphorus concentrations in the range of 0.013-0.037 mg/l produced algal concentrations which were suitable for a well-balanced ecosystem with good water quality as observed in 1971 by the Tennessee Valley Authority.

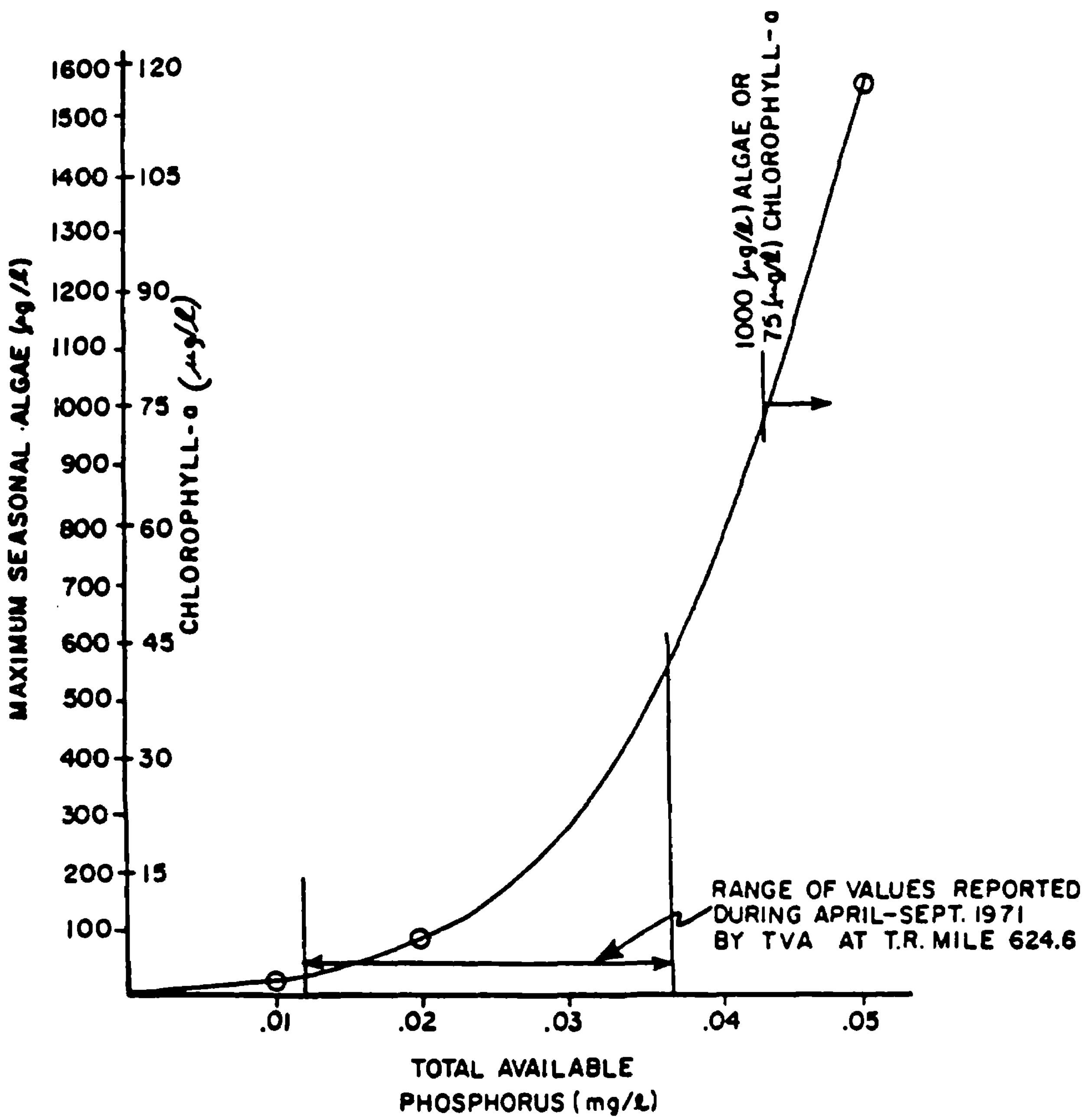

Figure II- 22 Maximum Seasonal Algae vs. Total Available Phosphorus
Lake Model Sensitivity Study - Fort Loudoun
(from Hall et al, 1976)

CHAPTER III

BIOLOGICAL CHARACTERISTICS

INTRODUCTION

This chapter contains information about the characteristic plants and animals found in lakes and provides an overview of the water quality and the types of habitat that they require. The chapter is divided into major sections: Plankton, Aquatic Macrophytes, Benthos, and Fish.

Particular emphasis is placed on changes in species composition as lakes progress from oligotrophy to eutrophy. The biota of lakes is often studied to assess the trophic state or biological health of the water body. Thus, indicator organisms are also discussed in this chapter, along with qualitative and quantitative methods of assessing the biological health of a lake. The reader is referred to the Technical Support Manual: Water Body Surveys and Use Attainability Analyses (U.S. EPA, 1983b) where an extensive discussion on species diversity and other measures of community health will be found.

PLANKTON

Planktonic plants and animals are important members of the lacustrine food web. Phytoplankton, which comprise pigmented flagellates, green and blue-green algae, and diatoms, are lowest on the food chain and serve as a primary food source for higher organisms. Zooplankton may be grazers (consuming phytoplankton) or predators (feeding on species smaller than themselves). The zooplankton, in turn, serve as the primary food source for the young of many fish species. The findings of various authors who have studied the effects of organic pollution and nutrient enrichment on the lacustrine plankton are summarized below.

Phytoplankton

The growth of phytoplankton is normally limited by the amount of nitrogen and/or phosphorus available. When increased quantities of nutrients enter the lake in runoff or effluents, eutrophication with its attendant uncontrolled algal growth and its consequences may begin. For example, the production of toxic substances by some algae may cause human gastrointestinal, skin and respiratory disorders, while blooms of *Microcystis* and *Nostoc rivulare* may poison wild and domestic animals, causing unconsciousness, convulsions and sometimes death (Mackenthun, 1969).

Algal blooms affect the dissolved oxygen (DO) content of the water. Diurnal fluctuations of DO and pH become more pronounced with large algal populations. In addition, the dissolved oxygen in the hypolimnion is depleted through algal death and decay, leading to anoxic conditions. Fish may die because of anaerobic conditions or the production of toxic substances. Water quality problems caused by algae, such as taste and odor, are especially troublesome if the water body is used as a source of drinking water. Finally, scums and mats of the algae destroy the aesthetic value of the lake.

Since some species are able to compete better than others, increased nutrients cause changes in phytoplankton community composition. Thus, specific algal associations may be indicative of eutrophic conditions. Indices of trophic state based on phytoplankton taxon are also related to the degree of eutrophy. The use of phytoplankton as indicators of eutrophication is discussed below.

Qualitative Response to Environmental Change

The identification of phytoplankton that are commonly found in eutrophic and oligotrophic lake waters has resulted in lists of pollution tolerant/intolerant genera and species. Palmer (1969) developed several lists of pollution tolerant algal genera and species by compiling information in 269 reports by 165 authors. The eight most tolerant genera were *Euglena*, *Oscillatoria*, *Chlamydomonas*, *Scenedesmus*, *Chlorella*, *Nitzchia*, *Navicula*, and *Stigeoclonium*. The five most tolerant species were *Euglena viridis*, *Nitzchia palea*, *Oscillatoria limosa*, *Scenedesmus quadricauda*, and *Oscillatoria tenuis*. Palmer used the following method to combine the works of the various authors: A score of 1 or 2 points was given for each algae reported by an author as tolerating organic enrichment, the larger figure being reserved for the algae that an author emphasized as being typical of waters with high organic pollution. The compilation by Palmer is presented in Appendix A, pollution-tolerant genera and pollution-tolerant species.

Palmer's listings have been criticized because the information used to compile them came from a broad range of sources and geographical areas. In addition, the compilation is restricted to algae tolerating high organic pollution. Thus, the listing may not be valid for other types of pollutants. Nevertheless, it does provide an indication of relative tolerance to organic pollution.

Taylor, et al. (1979) studied the environmental conditions associated with phytoplankton genera. The occurrence of 57 genera was related to total phosphorus levels, total Kjeldahl nitrogen levels, chlorophyll-*a* levels, and N/P ratio values. Most genera were found to occur over extremely wide ranges or conditions. The seven genera associated with levels of phosphorus greater than 200 ug/l were found to also represent seven of the eight highest chlorophyll-*a* values. Taylor designated this group containing *Actinastrum*, *Anabaenopsis*, *Schroederia*, *Raphidiopsis*, *Chlorogonium*, *Golenkinia*, and *Lagerheimia* as the "nutrient rich genera". All seven genera were summer and fall forms, while *Actinastrum* and *Lagerheimia* also occur in spring.

The "nutrient-poor" group, containing five genera, were associated with total phosphorus levels less than 70 ug/l. *Asterionella*, *Dinobryon*, *Tabellaria*, *Peridinium*, and *Ceratium* make up this group. *Asterionella* is the only genus occurring solely in spring. The other genera occur in summer and fall; *Dinobryon* and *Tabellaria* also occur equally in spring, summer and fall.

Taylor, et al. (1979) also noted which genera achieved numerical dominance most frequently in the lakes studied. *Melosira* was the most dominant genus, followed by *Oscillatoria* and *Lyngbya*. *Asterionella* was considered spring dominant, while *Stephanodiscus*, *Synedra* and *Tabellaria* were categorized as spring and summer dominant. Fragilaria occurred equally throughout the seasons as a dominant, and the remaining genera were summer and fall dominant. Additional information about the environmental conditions associated with the presence of the 20 phytoplankton genera most frequently recorded as dominants is available in Taylor, et al. (1979).

The study by Taylor, et al. (1979) concluded the following: (1) Phytoplankton genera survive over such a broad range of environmental conditions that they cannot be used as indicator organisms; (2) No phytoplankton genera emerged as dependable indicators of any one or combination of the environmental parameters measured; (3) Preliminary analyses suggest that phytoplankton community composition shows promise for use in water quality assessment; (4) Some taxa, e.g., Pediastrum and Euglena, were very frequent components of phytoplankton communities, but rarely achieved high relative numerical importance within those communities; (5) Flagellates and diatoms were the most common springtime plankton genera, while the blue-green and coccoid green genera were most common in the summer and fall; and (6) Blue-green algal forms, including several not known to fix elemental nitrogen, contributed 9 of the 10 genera which attained numerical dominance in water with a mean inorganic nitrogen/total phosphorus ratio (N/P) of less than 10 (generally suggestive of nitrogen-limitation).

Similarly, Bush and Welch (1972) concluded that phosphorus availability was most critical to the biomass formation of blue-green algae. They found that Aphanizomenon and Microcystis formed mats on the water surface during warm summer days, and were typical of shallow, hypereutrophic lakes such as Clear Lake (California), Klamath Lake (Oregon) and Moses Lake (Washington). Their study showed that the biomass of blue-green algae was related to inorganic phosphate even when nitrate was low and invariable.

Harris and Vollenweider (1982) noted some diatoms that are characteristic of oligotrophic lakes. Species of Tabellaria, Fragilaria, and Asterionella indicated oligotrophic conditions. In sediment cores of Lake Erie, species of Melosira showed the transition from oligotrophic to eutrophic conditions. The succession of species was as follows: Melosira distans and M. italica were present prior to 1850 and are considered indicative of oligotrophy; after 1850, M. distans and M. italica populations dwindled, and M. islandica (moderate enrichment) and M. granulata (eutrophication indicator) appeared in the core; in the next phase, around 1960, M. distans disappeared and was replaced by M. binderana.

Quantitative Response to Environmental Change

Because phytoplankton exhibit such a broad range of tolerance to environmental conditions, the presence or absence of a single species is not necessarily indicative of trophic state. In contrast, indices based on dominant genera, community composition, cell count, or chlorophyll-a provide a useful assessment of lake trophic levels and are better suited to the classification of lakes than single species evaluations.

Chlorophyll-a. Chlorophyll-a is a widely accepted index of algal biomass. In lakes and reservoirs with retention times greater than 14 days, it is highly correlated with phosphorus. The correlation does not hold for systems with less than 14-day retention times (U.S. EPA, 1979a). Estimates of chlorophyll-a values indicative of trophic state are shown in Table III-1.

Carlson's Trophic State Indices. Carlson (1977) developed three indices of trophic state, based upon Secchi depth, total phosphorus and chlorophyll-a. The three indices are defined below:

$$\text{Carlson's Secchi Depth Index, TSI(SD)} = 10\left(6 - \frac{\ln SD}{\ln 2}\right) \quad (1)$$

$$\text{Carlson's Chlorophyll-a Index, TSI(CHL)} = 10\left(6 - \frac{2.04 - 0.68 \ln CHL}{\ln 2}\right) \quad (2)$$

$$\text{Carlson's Total Phosphorus Index, TSI(TP)} = 10\left(6 - \frac{\ln 48/TP}{\ln 2}\right) \quad (3)$$

where

SD = Secchi disc depth, m

CHL = Concentration of chlorophyll-a, ug/l

TP = Concentration of total phosphorus, ug/l.

The scale of values for Carlson's Secchi Depth Index ranges from zero to greater than 100. A Secchi depth transparency of 64 m, which is greater than the highest value reported for any lake in the world, yields a value of zero. A Secchi depth of 32 m corresponds to an Index value of 10. An Index value of 100 represents a transparency of 0.062 m. Using empirically determined relationships between total phosphorus and transparency, and chlorophyll-a and transparency, Carlson developed equations (1), (2) and (3). These equations arrive at the same trophic state index value, regardless of whether Secchi depth, total phosphorus, or chlorophyll-a is the parameter used. However, it is desirable to evaluate all three indices because of non-nutrient related factors (temperature, inorganic turbidity, toxics) which may affect productivity and cause disagreement among the indices.

Based on observations of several lakes, most oligotrophic lakes had TSI below 40, mesotrophic lakes had TSI between 35 and 45, and most eutrophic lakes had TSI greater than 45. Hypereutrophic lakes may have values above 60 (Novotny and Chesters, 1981; Uttormark and Hutchins, 1978).

Nygaard's Trophic State Indices. Nygaard (cited by Sullivan and Carpenter, 1982) developed five phytoplankton indices (myxophycean, chlorophycean, diatom, euglenophyte, and compound) based on the assumption that certain algal groups are indicative of various levels of nutrient enrichment. He assumed that Cyanophyta, Euglenophyta, centric diatoms, and members of Chlorococcales are typical of eutrophic waters, while desmids and many pennate diatoms are generally found in oligotrophic waters. Nygaard's indices are listed in Table III-2. In applying these indices, the number of taxa in each major group is determined from the species list for each sample (U.S. EPA 1979a).

TABLE III-1

TROPHIC STATE VS. CHLOROPHYLL-a

Chlorophyll-a (ug/l)

Trophic Condition	Sakamoto, 1966	National Academy of Sciences, 1972	Dobson, et al., 1974	U.S. EPA, 1974
Oligotrophic	0.3-2.5	0-4	0-4.3	<7
Mesotrophic	1-15	4-10	4.3-8.8	7-12
Eutrophic	5-140	>10	>8.8	>12

SOURCE: U.S. EPA, 1979a.

TABLE III-2

NYGAARD'S TROPHIC STATE INDICES

Index	Calculation	Oligotrophic	Eutrophic
Myxophycean	$\frac{\text{Myxophyceae}}{\text{Desmideae}}$	0.0-0.4	0.1-3.0
Chlorophycean	$\frac{\text{Chlorococcales}}{\text{Desmideae}}$	0.0-0.7	0.2-9.0
Diatom	$\frac{\text{Centric Diatoms}}{\text{Pennate Diatoms}}$	0.0-0.3	0.0-1.75
Euglenophyte	$\frac{\text{Euglenophyta}}{\text{(Myxophyceae + Chlorococcales)}}$	0.0-0.2	0.0-1.0
Compound	$\frac{\text{(Myxophyceae + Chlorococcales + Centric Diatoms + Euglenophyta)}}{\text{Desmideae}}$	0.0-1.0	1.2-25

SOURCE: U.S. EPA, 1979a.

Nygaard's ranges show considerable overlap between trophic states. Sullivan and Carpenter (1982) sampled 27 lakes and reservoirs and found that Nygaard's indices did not differentiate between trophic states. In addition, an index value is undefined whenever the denominator is zero.

Palmer's Organic Pollution Indices. Palmer (1969) developed two algal pollution indices (genus and species) for rating water samples with high organic pollution. After reviewing reports of 165 authors, Palmer prepared two lists of organic pollution-tolerant forms, one containing 20 genera (Table III-3), and the other, 20 species (Table III-4).

In analyzing a water sample, any of the 20 genera or species present in concentrations of 50/ml or more are recorded. The pollution index numbers of the algae present are then totaled, giving a genus score (Palmer's Genus Index) and a species score (Palmer's Species Index). A score of 20 or more is taken as evidence of high organic pollution, while a score of 15 to 19 is taken as probable evidence of high organic pollution. Lower figures indicate that the organic pollution of the sample is not high, or that some substance or factor interfering with algal persistence is present or active (Palmer, 1969).

Use of Palmer's indices in a study of Indiana lakes and reservoirs showed that the Genus Index was more sensitive to differences among samples than the Species Index. The Genus Index was correlated with the degree of eutrophication, reflecting the abundance of eutrophic indicator genera. Another advantage of the Genus Index is that genera are easier to identify than species. However, a study of 250 lakes in the eastern and southeastern states showed that Palmer's indices were poorly correlated with summer mean phosphorus and chlorophyll-a levels, although the Genus Index ranked higher (Spearman's rank correlation coefficient) than the Species Index (U.S. EPA, 1979a).

U.S. EPA Proposed Phytoplankton Indices of Trophic State. Using a test set of 44 lakes in the eastern and southeastern states, EPA compared the abilities of several indices to measure trophic state (U.S. EPA, 1979a). The same report introduced 10 additional indices that used a combination of data including total phosphorus, chlorophyll-a, Kjeldahl nitrogen, phytoplankton genera counts and cell counts/ml.

Each genus was assigned "trophic values" based on mean parameter values associated with the dominant occurrence of that genus. The data used to assign trophic values was taken from studies of 250 lakes that were sampled during spring, summer and fall of 1973. Trophic values used in the general formulas of the new indices (Table III-5) are presented in Appendix B, along with sample problems using the indices.

When the newly developed indices were compared to Nygaard's and Palmer's indices, they showed a consistently stronger correlation with summer mean phosphorus levels and chlorophyll-a levels. When applied to the dominant phytoplankton community components, the indices generally had higher correlations than the analogous indices applied to all phytoplankton community components, although the differences were small (U.S. EPA 1979a).

TABLE III-3

VALUES USED IN ALGAL GENUS POLLUTION INDEX

Genus	Pollution Index
Anacystis	1
Ankistrodesmus	2
Chlamydomonas	4
Chlorella	3
Closterium	1
Cyclotella	1
Euglena	5
Gomphonema	1
Lepocinclis	1
Melosira	1
Micractinium	1
Navicula	3
Nitzschia	3
Oscillatoria	5
Pandorina	1
Phacus	2
Phormidium	1
Scenedesmus	4
Stigeoclonium	2
Synedra	2

SOURCE: Palmer, 1969.

TABLE III-4

VALUES USED IN ALGAL SPECIES POLLUTION INDEX

Species	Pollution Index
Ankistrodesmus falcatus	3
Arthrospira jenneri	2
Chlorella vulgaris	2
Cyclotella meneghiniana	2
Euglena gracilis	1
Euglena viridis	6
Gomphonema parvulum	1
Melosira varians	2
Navicula cryptocephala	1
Nitzschia acicularis	1
Nitzschia palea	5
Oscillatoria chlorina	2
Oscillatoria limosa	4
Oscillatoria princeps	1
Oscillatoria putrida	1
Oscillatoria tenuis	4
Pandorina morum	3
Scenedesmus quadricauda	4
Stigeoclonium tenue	3
Synedra ulna	3

SOURCE: Palmer, 1969.

TABLE III-5

EPA PROPOSED PHYTOPLANKTON INDICES TO TROPHIC STATE

Phytoplankton Trophic State Index (TSI) Calculations Without Cell Counts:

$$TSI = \sum_{i=1}^{n} V_i/n$$

n = number of dominant genera in the sample (Concentration $\geq$ 10 percent of the total sample concentration).

V_i* = the trophic value for each dominant genus in the sample; TOTALP (PD), CHLA (PD), KJEL (PD), MV (PD); MV = Log TOTALP + Log CHLA + Log KJEL - Log SECCHI

Phytoplankton Trophic State Index (TSI) Calculations with Cell Counts:

$$TSI = \sum_{i=1}^{n} V_i C_i$$

Total Community:

n = the number of genera in the sample (entire phytoplankton community)

C = the concentration of the genus in the sample (units/ml)

V = the trophic value for each genus;
TOTALP/CONC(P), CHLA/CONC(P), KJEL/CONC(P)

Dominant Community:

n = the total number of dominant genera in the sample

C = the concentration of the genus in the sample (units/ml)

V = the trophic value for each genus;
TOTALP/CONC (P), CHLA/CONC (PD), KJEL/CONC (PD)

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*The parameters TOTALP, CHLA, etc. are defined in Appendix B.

SOURCE: U.S. EPA, 1979a.

Zooplankton

As lakes become enriched, phytoplankton and (to a large degree) herbivorous zooplankton populations increase. Changes in species composition also occur, although it is difficult to classify the trophic state of a water body on the basis of a list of zooplankton species living in it. Generally, larger species of zooplankton dominate in oligotrophic waters. This is probably largely due to predation pressure. In eutrophic waters, where the fish stock is heavy, the larger zooplankton are eaten first. Thus, the number of zooplankters that attain a large size is limited.

Species of Bosmina have been commonly accepted as indicators of enrichment. Hutchinson (1967) observed that Bosmina coregoni longispina appeared to be characteristic of larger and less productive lakes, and B. longirostris of smaller and more productive lakes. Studies on the sediments of Linsley Pond, Connecticut (Deevy, 1940), indicated that the disappearance of B. coregoni longispina was concurrent with the appearance of B. longirostris as the lake became enriched. However, the collection of B. longirostris from the epilimnion, and B. coregoni from the hypolimnion of another lake shows the uncertainty of using Bosmina spp. as indicators.

Studies of zooplankton in the Great Lakes showed the following:

1. A decreased significance of calanoids and an increased predominance of cyclopoids and cladocerans were seen as a general trend from oligotrophic Lake Superior to eutrophic Lake Erie (Patalas, 1972; Watson, 1974).

2. Larger zooplankton were observed in Lakes Superior and Huron, although Lake Erie had an increased biomass of zooplankton (Patalas, 1972; Watson, 1974).

3. In Lake Michigan, Bosmina coregoni has been replaced by B. longirostris, Diaptomus oregonensis has become an important copepod species, Eurytemora affinis appeared (Beeton, 1969).

4. Diaptomus siciloides, usually found in eutrophic waters has become a dominant zooplankton in Lake Erie (Beeton, 1969).

Some rotifers have been considered indicators of eutrophied waters. However, these organisms (in particular, Brachionus and Keratella quadrata) have also been collected from oligotrophic lakes. Other zooplankton are difficult to identify and thus are not practical to use as indicators of water quality. For example, Cyclops scutifer is principally an oligotrophic form while Cyclops scutifer wigrensis lives in meso- and eutrophic lakes (Ravera, 1980).

Sprules (1977) developed a technique for predicting the limnological characteristics of a lake which is based on its midsummer limnetic crustacean zooplankton community. The results indicated that northwestern Ontario lakes characterized by Cyclops bicuspidatus thomasi, and Diaptomus minutus are generally large and clear, whereas Tropocyclops prasinus mexicanus and Diaptomus minutus are typical of smaller lakes with lower water clarity. Acidic, small and clear lakes of the Killarney region, Ontario, are dominated by _Diaptomus minutus_, while _Diaphanosoma leuchtenbergianum_, _Bosmina longirostris_ and _Mesocyclops edax_ dominate in lakes that are less clear, larger and have a higher pH. Finally, in the Haliburton region of Ontario, small and productive lakes are characterized by _Diaptomus oregonensis_, _M. edax_, and _Ceriodaphnia lacustris_. Those lakes with _D. minutus_, _D. sicilis_, _B. longirostris_ and _Daphnia duba_ are larger and less productive.

Thus, the direct effects of nutrient enrichment on the zooplankton are unclear. Although a few qualitative changes have been mentioned, the only quantitative information refers obliquely to diversity indices. The diversity of the zooplankton community generally decreases with increasing enrichment, as do the other organism communities. Diversity Indices are discussed in the _Technical Support Manual: Water Body Surveys and Assessments for Conducting Use Attainability Analyses_ (1983b).

AQUATIC MACROPHYTES

Aquatic plants play several roles in the lake ecosystem. They produce oxygen through photosynthesis, shade and cool sediments, diminish water currents and provide habitat for benthic organisms and fish (Boyd, 1971). Carignan and Kalff (1982) found that water milfoil (_Myriophyllum spicatum_ L.) was important as physical support for microbial communities. Submersed macrophytes serve as food and nest sites for aquatic insects and fish, and provide protection from predation. The plants also play a role in nutrient cycling, especially in the mobilization of phosphorus from sediments. Barko and Smart (1980) investigated the uptake of phosphorus from five different sediments by _Egeria densa_, _Hydrilla verticillata_, and _Myriophyllum spicatum_. The amount of sediment phosphorus mobilization differed among species and sediments, but it was demonstrated that the plants were able to obtain their phosphorus nutrition exclusively from the sediments. Release of phosphorus from the macrophytes occurred primarily through death and decay rather than through excretion. Landers (1982) showed that decomposing _Myriophyllum spicatum_ supplied significant amounts of nitrogen and phosphorus to surrounding waters. Nitrogen inputs accounted for less than 2.2 percent of annual allochthonous inputs, but phosphorus recycling from decaying plants equaled up to 18 percent of the total annual phosphorus loading for the reservoir studied.

Response of Macrophytes to Environmental Change

Major environmental changes in lakes generally occur in response to nutrient increases (which accelerate eutrophication), suspended sediment, and sediment deposition. Suspended sediment attenuates light penetration, resulting in reduced photosynthesis by submerged aquatic macrophytes, and a possible decrease in the coverage by plants. Reed, et al. (1983) noted that the growth of _Chara_ in a test pond was restricted during years when the turbidity was high, but luxurious stands developed when the water was clearer. Sediment deposition smothers some plants. For example, _Isoetes lacustris_ is not present in areas with rapid silting, but _Nitella_ and _Juncus_ often occur instead (Farnworth, 1979). _Potamogeton perfoliatus_ may also replace _Isoetes_ where silting occurs. The composition of the substrate is important in the growth of macrophytes. _Potamogeton perfoliatus_, _Elodea canadensis_, and _Myriophyllum spicatum_ reportedly grew more rapidly in natural sediment than in sand. Lobelia dortmanna grew only in sand containing organic matter (Farnworth, 1979).

Although aquatic macrophytes are vital to the ecosystem, eutrophication and the subsequent overgrowth of plants may be detrimental to the water body. Diurnal DO fluctuations driven by photosynthesis and respiration may be so extreme that oxygen deficits occur. Oxygen depletion in the hypolimnion may also be caused by decaying macrophytes. Low DO may cause fish kills and eliminate sensitive species (Boyd, 1971).

Although eutrophication is often considered the cause of changes in macrophyte composition, management techniques may also be responsible. Nicholson (1981) argued that techniques such as herbicidal poisoning and mechanized cutting were primary reasons for the replacement of native Potamogeton species in Chautagua Lake, New York, by Potamogeton crispus and Myriophyllum spicatum.

Preferred Conditions

Certain aquatic plants are able to "out-compete" others and in large populations become established under eutrophic conditions. Such excessive growth is usually undesirable, and the plants are considered aquatic weeds. Aquatic plants that cause difficulty in the United States include Myriophyllum spicatum var. exalbescens (water milfoil), Potamogeton crispus (curly-leaved pondweed), Eichornia crassipes (water hyacinth), Pistia stratioles (water lettuce), Alternanthera philoxeroides (alligator weed), Heteranthera dubia (water stargrass), Myriophyllum brasiliense (parrot feather), M. spicatum var. spicatum (eurasian water milfoil), Najas guadalupensis (southern naiad), Potamogeton pectinatus (sago pondweed), Elodea canadensis (elodea), and Phragmites communis (common weed).

Seddon (1972) investigated the environmental tolerances of certain aquatic macrophytes found in lakes. He grouped the species into the following:

1. Tolerant species that occur over a wide range of solute concentrations - Potamogeton natans, Nuphar lutea, Nymphaea alba, Glyceria fluitans, Littorella uniflora;

2. Highly eutrophic species - Potamogeton pectinatus, Myriophyllum spicatum;

3. Moderately eutrophic species - Potamogeton crispus, Lemna trisulca;

4. Species tolerant of mesotrophic as well as eutrophic conditions - Ranunculus circinatus, Lemna minor, Polygonum amphibium, Ceratophyllum demersum, Potamogeton obtusifolius;

5. Species of oligotrophic tolerance - Potamogeton perfoliatus, Ranunculus aquatilis, Apium inundatum, Elodea canadensis, Potamogeton berchtoldii.

Plants occurring only in eutrophic conditions were considered restricted to such areas by physiological demands. It should be noted that the last group, although classified as of oligotrophic tolerance, may also be found in eutrophic waters. Oligotrophic species, while shown to have a wide tolerance, are thought to be excluded by competition rather than by physiological limitation from sites with higher trophic status. The last group in effect includes those species that can adapt to the relatively nutrient free conditions of oligotrophic water.

BENTHOS

Benthic macroinvertebrates are often used as indicators of water quality. Because they are present year-round, are abundant, and are not very motile, they are well-suited to reflect average conditions at the sampling point. Many species are sensitive to pollution and die if at any time during their life cycle they are exposed to environmental conditions outside their tolerance limits.

There are also disadvantages to basing the evaluation of the biotic integrity of a water body solely on macroinvertebrates. Identification to the species level is time-consuming and requires taxonomic expertise. Furthermore, the results may be difficult to interpret because life history information is lacking for many species and groups, and because a history of pollution episodes in the receiving water may not be available to provide perspective for the interpretation of results.

Certain organisms and associations of organisms point to various stages of eutrophy. Decay of organic material often decreases the DO (dissolved oxygen) content of the hypolimnion below the tolerance of the invertebrates. Attempts to translate the results of studies into meaningful values have yielded lists (presented later in this section) of tolerant and intolerant groups of macroinvertebrates. In addition, mathematical formulas have been developed which assign numerical values to various trophic states depending upon the benthos present. However, factors other than organic pollution (e.g., substrate, temperature, depth) may also influence the species composition of benthic populations. Parameters such as these which govern species distribution are discussed in Merritt and Cummins (1978).

Composition of Benthic Communities

The composition of the benthos in littoral and profundal areas of a lake is mostly dependent upon substrate, but is also influenced by depth, temperature, light penetration and turbidity. The littoral regions of lakes usually support larger and more diverse populations of benthic invertebrates than profundal areas (Moore, 1981). Benthic communities in the littoral regions consist of a rich fauna with high oxygen demands.

The vegetation and substrate heterogeneity of the littoral zone provide an abundance of microhabitats occupied by a varied fauna. By contrast, the profundal zone is more homogeneous, becoming more so as lakes become more eutrophic (Wetzel, 1975). One of the best illustrations of the differences of littoral and profundal benthos is seen in studies of Lake Esrom, a dimictic lake in Denmark (Jonasson, 1970). The bottom fauna found on subsurface weeds (depth about 2m) comprises thirty-three groups and species, totaling 10,810 individuals per square meter. In contrast, only five species are found in the profundal zone of Lake Esrom, although the density is high (20,441 per square meter). The animals in this region burrow into the bottom instead of living on or near the surface.

The factors mentioned above should be considered in the design of a study of lake benthos. Because substrates of deep waters generally have finer sediment particles than substrates of shallow waters, depth should be considered in quantitative calculations to help compensate for substrate differences. Adjustments for depth will be discussed in greater detail in the section on quantitative measures of the effects of pollution on benthos.

General Response to Environmental Change

The benthos of freshwater is composed largely of larvae and nymphs of aquatic insects (Arthropoda: Insecta). The benthos also comprises freshwater sponges (Porifera: Spongillidae), flatworms (Platyhelminthes: Tricladida), leeches (Annelida: Hirudinea), aquatic earthworms (Annelida: Oligochaeta), snails (Mollusca: Gastropoda), clams and mussels (Mollusca: Bivalvia). Particular groups of insects are most abundant in specific kinds of freshwater habitat. Damselflies and dragonflies (Odonata) are generally found in shallow lakes, but some species occur in running water. Stoneflies (Plecoptera) and mayflies (Ephemeroptera) are predominantly running water forms, although certain Ephemeroptera dwell in lakes and ponds. Caddisflies (Tricoptera) abound in lakes and streams where the water is well-aerated. The other groups also occur in both streams and lakes (Edmondson, 1959).

Aquatic insects can be identified by using various keys (Pennak, 1978; Edmondson, 1959; Needham and Needham, 1962; Merritt and Cummins, 1978). Merritt and Cummins (1978) also provide lists of the species and habitats (lentic or lotic) where they are most often found.

The species composition and number of individuals of the benthic community change in response to increased organic and inorganic loading. Organic pollution generally causes a decrease in the number of species of organisms, but an increase in the number of individuals. Inorganic pollution, such as sediment, causes a decrease in the number of individuals, as well as a decrease in species. The following sections focus on qualitative and quantitative changes in freshwater benthic populations that are indicative of types of pollution and of trophic state in lakes and reservoirs.

Qualitative Response to Environmental Change

The most sensitive macroinvertebrate species are usually eliminated by organic pollution. Because decay of organics often depletes oxygen, the surviving species are those that are more tolerant of low dissolved oxygen content. The predominant bottom conditions can be inferred by observing which species are present at a specific site.

Suspended sediment and silt deposition may influence macroinvertebrates by causing:

(a) Avoidance of adverse conditions by migration and drift;

(b) Increased mortality due to physiological effects, burial, and physical destruction;

(c) Reduced reproduction rates because of physiological effects, substrate changes, loss of early life stages;

(d) Modified growth rates because of habitat modification and changes in food type and availability (Farnworth, et al., 1979).

Indicator Organisms

The macroinvertebrate classes that are most often used as indicator organisms are the Insecta and Annelida. These organisms are illustrated in Figure III-1. Stonefly nymphs, mayfly naiads, and hellgrammites are generally considered to be relatively sensitive to environmental changes. The intermediately tolerant macroinvertebrates include scuds, sowbugs, blackfly larvae, dragonfly nymphs, damselfly nymphs, and leeches. Bloodworms (midge larvae) and sludgeworms make up the group of very tolerant organisms.

Anaerobic environments are tolerated by sewage fly larvae and rat-tailed maggots. Table III-6 lists those aquatic insects that have been found at dissolved oxygen concentrations of less than 4 ppm. The greatest number of tolerant species are members of the order Diptera.

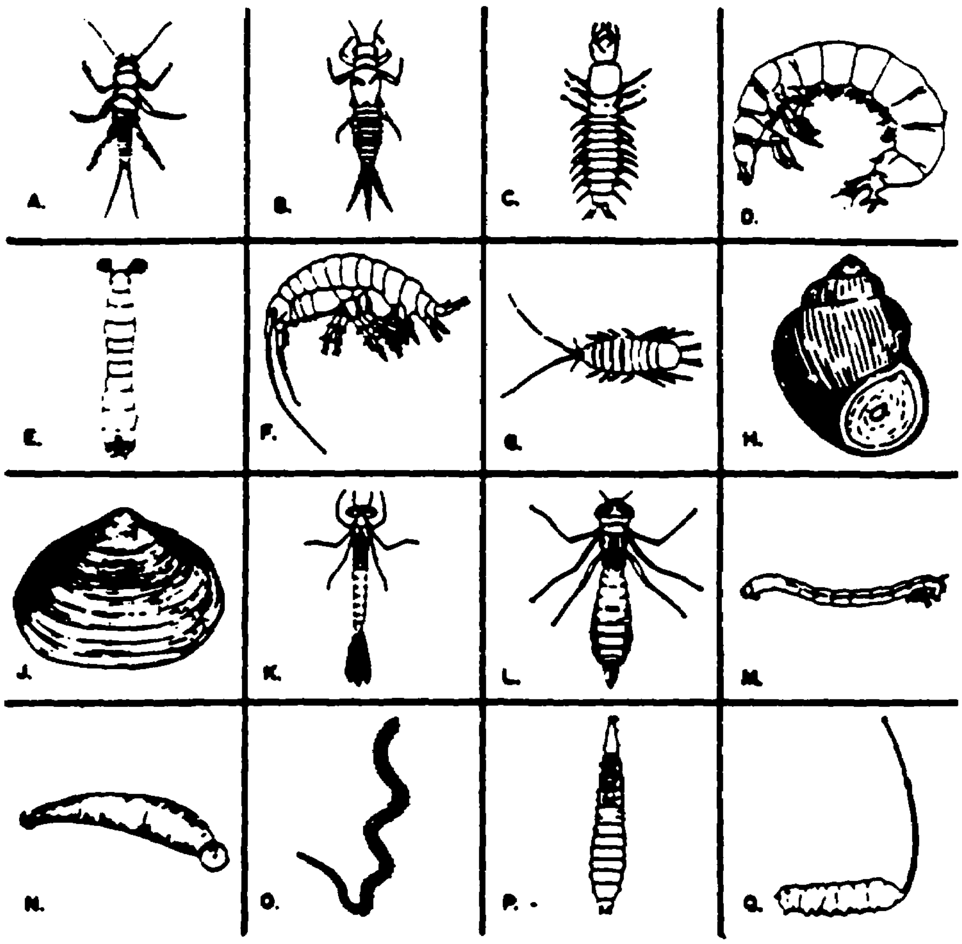
Sponges are affected by pollution although they are not usually considered indicator organisms. Of the freshwater sponges, Ephydatia fluviatilis, E. muelleri, Heteromeyenia tubisperma, and Eunaius fragilis may be found in eutrophic waters. Also, Ephydatia robusta can survive very low dissolved oxygen levels and has been collected at DO tensions of 1.00 ppm (Harrison, 1974). Of the Mollusca, Unionid clams (Bivalvia) are considered sensitive to environmental changes. Snails (Gastropoda) commonly occur in moderately polluted environments. The most resistant species are Physa heterotropha, P. integra, P. gyrina, Gyranulus parvus, Helisoma anceps, and H. trivolvis, but almost every common species has been found in polluted areas (Harman, 1974).

Weber (1973) compiled a list of tolerances of freshwater macroinvertebrate taxa to organic pollution (Appendix C). Organisms that occur in streams and lakes are included. The tolerances of the organisms listed in the appendix are based upon classification by various authors.

Trends in macroinvertebrate populations have been shown in studies of eutrophic lakes. A collection of studies report the following responses of macrofauna to increasing eutrophication:

- Oligochaetes, chironomids, gastropods and sphaerids increase and Hexagenia (mayfly nymph) decreases (Carr and Hiltunen, 1965);

- Numbers of oligochaetes relative to chironomids increase as organic enrichment increases (Peterka, 1972);



A. Stonefly nymph (Plecoptera)
B. Mayfly naiad (Ephemeroptera)
C. Hellgrammite or Dobsonfly larvae (Corydalidae)
D. Caddisfly larvae (Trichoptera)
E. Blackfly larvae (Simuliidae)
F. Scud (Amphipoda)
G. Aquatic sow bug (Isopoda)
H. Snail (Gastropoda)
J. Fingernail clam (Sphaeriidae)
K. Damselfly nymph (Zygoptera)
L. Dragonfly nymph (Anisoptera)
M. Bloodworm or midge fly larvae (Tendipedidae)
N. Leech (Hirundinea)
O. Sludgeworm (Tubificidae)
P. Sewage fly larvae (Psychoda)
Q. Rat-tailed maggot (Tubifera-Eristalis)

Figure III-1. Representative bottom fauna (from Keup, et al., 1966).

TABLE III-6

SPECIES FOUND AT DISSOLVED OXYGEN LESS THAN 4 PPM

Odonata - dragonflies and damselflies
  Ischnura posita (Hagen)
  Pachydiplax longipennis (Burm.)
Ephemeroptera - mayflies
  Paraleptophlebia sp.
  Caenis sp.
Hemiptera - true bugs
  Notonecta irrorata Uhl.
  Plea striola Fieb.
  Ranatra australis Hung.
  Ranatra kirkaldyi Bueno
  Pelocoris femoratus P. de B.
  Belostoma fluminea Say
  Trepobates sp.
  Rhagovelia obesa Uhl.
Megaloptera - alderflies, dobsonflies, and fishflies
  Chauliodes sp.
Coleoptera - beetles
  Haliplus spp.
  Peltodytes spp.
  Coelambus spp.
  Laccophilus spp.
  Hydroporus spp.
  Dineutes spp.
  Gyrinus spp.
  Tropisternus spp.
  Machronychus glabratus Say
  Stenelmis grossa Sand.
Lepidoptera - butterflies and moths
  Parapoynx sp.
Trichoptera - caddisflies
  Polycentropus remotus (Banks)
  Oecetis eddlestoni Ross
Diptera - true flies
  Procladius bellus (Loew)
  Clinotanypus pinguis (Loew)
  Ablabesmyia monilis (L.)
  Trichocladius sp. Roback
  Chironomus attenuatus (Walk.)
  Chironomus riparius (Meig.)
  Cryptochironomus nr. *fulvus* (Joh.)
  Dicrotendipes nervosus (Staeger)
  Harnischia nr. *abortiva* (Mall.)
  Microtendipes pedellus DeGeer
  Tribelos jucundus (Walk.)
  Rheotanytarsus exiguus (Joh.)
  Calopsectra nr. *guerla* Roback
  Palpomyia gp. spp.
  Tubifera tenax (L.)

SOURCE: Roback, 1974.

- o The smallest insect larvae are characteristic of oligotrophic waters, and due to a shift in species composition, larval size increases with increasing eutrophication (Jonasson, 1969);

- o Tanytarsini are replaced by Chironomini in positions of dominance with increasing eutrophication (Paterson and Fernando, 1970).

The study of four reservoirs (Salt Valley Reservoirs) in eastern Nebraska revealed several trends in macrobenthic communities as eutrophication progressed. Contrary to the observation frequently reported that oligochaete populations increase as eutrophication progresses, Hergenrader and Lessig (1980b) observed a decrease in Tubifex. They noted, however, that the deep hypolimnetic waters of the Salt Valley reservoirs do not become anaerobic, as is the case in lakes where oligochaetes have increased. The Tanytarsini (family Chironomidae) present in the less eutrophic reservoirs disappeared in the most eutrophic. Finally, Sphaerium (order Mollusca) increased during the early stages of eutrophication but declined as eutrophy progressed.

Chironomid Communities as Indicators

Instead of using a single organism to indicate water quality, Saether (1979, 1980) suggests studying chironomid communities. By looking at profundal, littoral and sublittoral chironomid communities, Saether was able to delineate 15 characteristic communities found in environments ranging from oligotrophic to eutrophic. The communities, 6 in each of the oligotrophic and eutrophic and 3 in the mesotrophic range, are lettered from alpha to omikron. The Greek letters emphasize that the 15 subdivisions are not trophic level divisions, but are recognizable chironomid communities. The species found in a lake or part of a lake can be used to determine the associations and hence the extent of eutrophy. The key to chironomid associations and the species list noted by Saether are presented in Appendix D. By using this system, Saether found significant correlations between chironomid associations and the ratios of chlorophyll-a to mean depth (Figure III-2) and total phosphorus to mean depth (Figure III-3).

Sediment Effects

The distribution of macroinvertebrates will be much less affected by currents and drift in a lake than in a river. However, at those points where rivers enter a lake, or where a river forms at the outlet from a lake, one might expect to find macroinvertebrate populations that are similar to the population of the connecting river. The distribution of macroinvertebrates found in the littoral zone will be less affected by drift (since rooted plants in the littoral tend to slow currents and thereby inhibit drift) and more by the physical effects of suspended solids and sedimentation. As concentrations of suspended and settleable solids increase, invertebrates tend to release hold of the substrate to be transported by currents or to migrate elsewhere. Migration from those areas affected by sediment changes the structure of the benthic community. The effects of suspended solids on benthic macroinvertebrates are summarized in Table III-7.

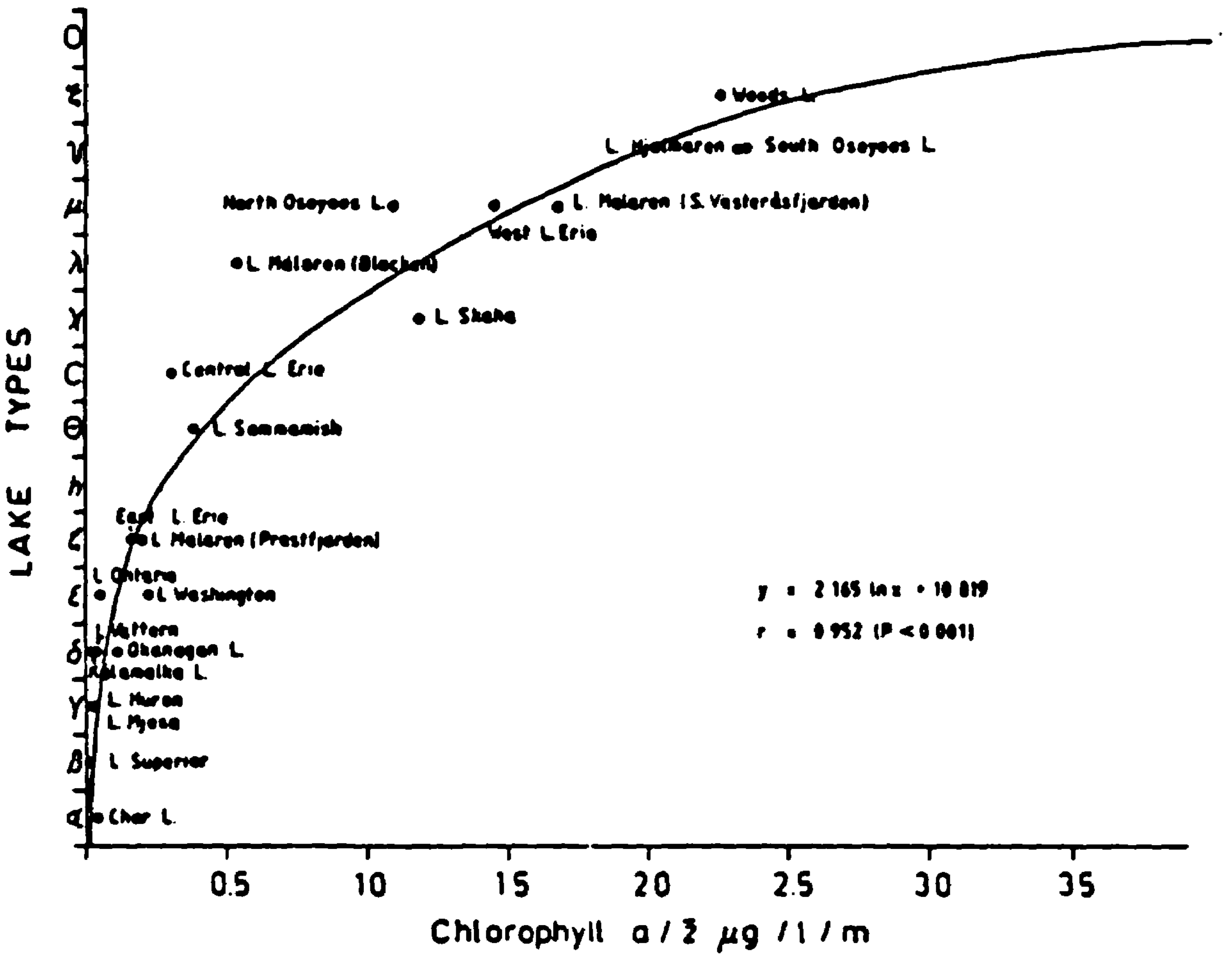


Figure III-2. Chlorophyll-a/ Mean Lake Depth in relation to 15 lake types based on Chironomid Communities (From Saether, 1979).

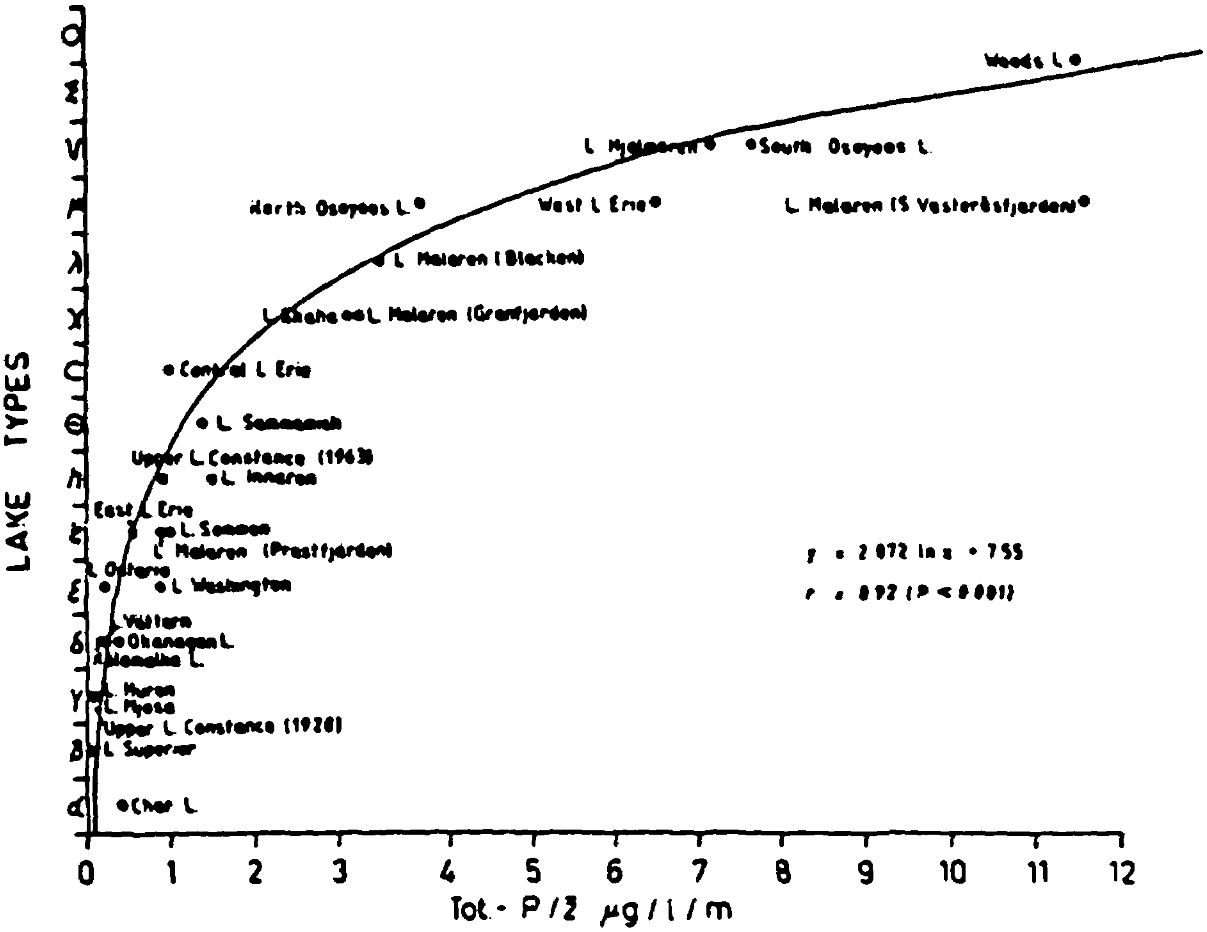


Figure III-3. Total Phosphorus/Mean Lake Depth in relation to 15 lake types based on Chironomid communities (From Saether, 1979).

TABLE III-7
SUMMARY OF SUSPENDED SOLIDS EFFECTS ON AQUATIC MACROINVERTEBRATES

Organism(s)	*Effect*	*Suspended Solid Concentration*	*Source of Suspended Solids*	*Comment*
Mixed Populations	Lower summer populations		Mining area	
Mixed Populations	Reduced populations to 25%	261-390 ppm (Turbidity)	Log dragging	
Mixed Populations	Densities 11% of normal	1000-6000 ppm		Normal populations at 60 ppm
Mixed Populations	No organisms in the zone of settling	>5000 ppm	Glass manufacturing	Effect noted 13 miles downstream
Chrionomus & Tubificidae	Normal fauna replaced by (Species Selection)		Colliery	Reduction in light reduced submerged plants
Cheumatopsyche (Net spinners)	Number reduced	(High concentrations)	Limestone Quarry	Suspended solids as high as 250 mg/l
Tricorythoides	Number increased		Limestone Quarry	Due to preference for mud or silt
Mixed Populations	90% increase in drift	80 mg/l	Limestone Quarry	
Mixed Populations	Reduction in numbers	40-200 JTU	Manganese Strip mine	Also caused changes in density and diversity
Chironomidae	Increased drift with suspended sediment		Experimental sediment addition	
Ephemoptera, Simuliidae, Hydracarina	Inconsistent drift response to added sediment		Experimental sediment addition	

SOURCE: Sorenson, et al., 1977.

Deposition of sediment in the profundal zone may provide a stable substrate. In contrast deltas where streams enter the lake or reservoir may be subject to continuing deposition and erosion. Such areas will support fewer species and fewer numbers of organisms than the more stable profundal zone.

Sediment deposition modifies macroinvertebrate habitat and alters the type, distribution and availability of food. Substrate preference of macroinvertebrates is related to a variety of factors. In addition to particle size, the colonization of an area is dependent on the amount and type of detritus, the presence of vegetation, the degree of compaction and the amount of periphyton (Farnworth et al., 1979). Sediment preferences may change with an organism's life history stage, thus compounding the problem of categorizing associated substrate. Nonetheless, certain groups such as Chironomidae and *Tricorythodes*, are recognized as preferring fine sediment.

Quantitative Response to Environmental Change

Quantitative techniques that are used to assess the biological integrity of lakes include a number of mathematical indices, or focus on the abundance of certain benthic organisms. These methods are summarized in the following sections. Other measures of community health, such as diversity indices, are discussed in the *Technical Support Manual: Water body Surveys and Assessments for Conducting Use Attainability Analyses* (U.S. EPA, 1983b), and in a review by Washington (1984).

Oligochaete Populations

Oligochaetes, particularly members of the family Tubificidae, are present in large numbers in polluted areas. Aston (1973) found that *Limnodrilus hoffmeisteri* and *Tubifex tubifex* predominate in areas receiving heavy sewage pollution. In a review of the relationship between tubificids and water quality, Aston (1973) noted several investigations that have used the population density of tubificids as an index of pollution. Surber (cited by Aston, 1973), studied a number of lakes in Michigan and concluded that areas with an oligochaete density of more than 1,100 per square meter were truly polluted. Carr and Hiltunen (1965) used the following numbers of oligochaetes per square meter to indicate pollution in western Lake Erie: light pollution, 100 to 999; moderate pollution, 1,000 to 5,000; and heavy pollution, more than 5,000. This means of classification fails to consider seasonal variation in population density and the organic content and particle size of the bottom substrate. Since the population density is likely to vary, this method has limited utility (Aston, 1973).

Wiederholm (1980) noted that a simple depth adjustment could make oligochaete abundance more applicable. By dividing the number of oligochaetes per square meter by the sampling depth, he found that the correlation with chlorophyll was increased. This adjustment may account for factors that are affected by depth such as food supply, predation pressure (which declines as depth increases), and possible oxygen deficits.

The relative abundance of oligochaetes may be a better indication of organic pollution than the population density. In a stream study, Goodnight and Whitley (1961) suggested that a population of 80 percent or more of oligochaetes in the total macroinvertebrate population indicates a high degree of organic enrichment. They hypothesized that percentages from 60 to 80 indicate doubtful conditions and below 60 percent, the area is in good condition. Howmiller and Beeton (1971) used this index in a study of Green Bay, Lake Michigan, and concluded that in 1967 the lower bay was in a highly polluted state, and the middle bay had "doubtful conditions."

Brinkhurst (1967) suggested that the relative abundance of the tubificid Limnodrilus hoffmeisteri (as a percentage of all oligochaetes) may be a useful measure of organic pollution. Increased percentages of L. hoffmeisteri are often indicative of organic pollution. Lower Green Bay (73% L. hoffmeisteri) was identified as being more polluted than middle Green Bay (50% and 42% L. hoffmeisteri) by reference to the relative abundance of this oligochaete (Howmiller and Scott, 1977).

Oligochaete/Chironomid Ratio

Another proposed indicator uses the ratio of oligochaetes to chironomids. Generally, the ratio increases as the lake becomes more eutrophic. Wiederholm (1980) advocates including a depth adjustment (ratio divided by sampling depth) when using the oligochaete/chironomid ratio since oligochaetes tend to increase in dominance at greater depths. Studies of Swedish lakes showed a high correlation between depth-adjusted oligochaete/chironomid ratios and trophic state, but very little correlation of the non-adjusted ratio with trophic state. Table III-8 shows that the depth-adjusted oligochaete/chironomid ratio had low values (from 0-1.5) in oligotrophic lakes, and progressively higher values for mesotrophic (1.5-3.0), eutrophic (3.0-7.4) and hypereutrophic (>18) lakes. Wiederholm suggests that the oligochaete/chironomid ratio may be used directly when comparing data from a single site over time or different lakes over time, but a general application needs some adjustment for depth.

Mathematical Indices

A survey of the literature reveals at least four mathematical indices in addition to diversity indices that may be applicable in freshwater lake studies. These indices are described in Table III-9.

Based on their studies of rivers and streams receiving sewage, Kolkwitz and Marsson (1908, 1909) proposed their sapropic system of zones of organic enrichment. They suggested that a river receiving a load of organic matter would purify itself and that it could be divided into saprobic zones downstream from the outfall, each zone having characteristic biota. Kolkwitz and Marsson published long lists of the species of plants and animals that one could expect to be associated with each zone. The zones were defined as follows:

- Polysaprobic: gross pollution with organic matter of high molecular weight, very little or no dissolved oxygen and the formation of sulphides. Bacteria are abundant, and few species of organisms are present.

TABLE III-8

BENTHIC COMMUNITY MEASURE
WITH AND WITHOUT ADJUSTMENT FOR DEPTH

Lake	Approximate Trophic State[a]	Chlorophyll-a (ug/l)[b]	Oligochaete/ Chironomid Ratio (%) without depth adj.	Oligochaete/ Chironomid Ratio (%) with depth adj.[c]
Vattern, 20-40m	O	1.1	38.9	1.3
Vattern, 90-110m	O	1.1	90.1	0.9
Vanern, 40-80 m	O	1.7	86.0	1.5
Skaren, 10-26m	O	2-2.5	25.9	1.5
Innaren, 14-19m	M	2.5-3	19.8	1.2
Sommen, 16-49m	M	3-4	44.3	1.9
Malaren, area C, 30m	M	5.5	85.5	2.9
Malaren, area C, 45-50m	M	5.5	96.4	2.0
Malaren, area B, 15m	E	17.5	69.0	4.6
Hjalmaren, area C, 6-18m	E	9.4	71.9	7.4
S. Bergundasjon, 3-5m	HE	25-75	69.0	18.5
Vaxjosjon, 3-5m	HE	50-100	87.4	21.6
Hjalmaren, area B, 2-3m	HE	102	66.8	34.4

a. O = oligotrophic, M = mesotrophic, E = eutrophic, HE = hypereutrophic

b. May-October, 1m

c. Oligochaete/Chironomid ratio divided by sampling depth

SOURCE: Wiederholm, 1980.

TABLE III-9

MATHEMATICAL INDICES

Index Name and Description	Reference
Saprobic Index	Saether, 1979
Benthic Quality Index	Wiederholm, 1976 Wiederholm, 1980
Trophic Condition Index	Howmiller and Scott, 1977 Saether, 1979

Saprobic Index — Saether, 1979

$$S = \frac{\Sigma s \cdot h}{\Sigma h}$$

s = 1-4, Oligo - to polysaprobic
h = occurrence value; 1, occasional; 3, common; 5, mass occurrence.

Benthic Quality Index — Wiederholm, 1976; Wiederholm, 1980

$$BQI = \sum_{i=0}^{5} \frac{N_i \cdot k_i}{N}$$

k_i = based on indicator species of chironomids, see text
n_i = number of individuals of the various groups
N = the total number of indicator species

Saether, 1979

$$BQI = \sum_{i=0}^{5} \frac{N_i(k_i - 1 + C_i)}{N}$$

C_i = the constancy of the respective groups within a sample

Trophic Condition Index — Howmiller and Scott, 1977; Saether, 1979

$$TCI = \frac{\Sigma N_1 + 2\Sigma N_2}{\Sigma N_0 + \Sigma N_1 + \Sigma N_2}$$

ΣN_0 = total number of oligochaete worms intolerant of eutrophic conditions (see Table C)
ΣN_1 = total number of organisms characteristics of mesotrophic areas
ΣN_2 = total number belonging to species tolerant of extreme eutrophy

- o <u>Mesosaprobic:</u> simpler organic molecules and increased DO content. Upper zone (alpha-mesosaprobic) has many bacteria and often fungi, with more types of animals and lower algae. Lower zone (beta-mesosaprobic) has conditions suitable for many algae, tolerant animals and some rooted plants.

- o <u>Oligosaprobic:</u> oxygen content is back to normal and a wide range of plants and animals occur.

As stated, the saprobic system was designed for rivers and streams. Nevertheless, the concept could be applied to riverine impoundments that have a predominant longitudinal flow. More importantly, however, is the impetus generated by the saprobic system for the development of subsequent biological indices.

Pantle and Buck (1955, cited by Saether, 1979) applied the ideas of Kolkwitz and Marsson in the Saprobic Index (Table III-9), which was proposed for use in stream studies. Further extensions of the saprobic system were made by Sladecek (1965) and these modifications are summarized in Nemerow (1974).

Wiederholm proposed the Benthic Quality Index (BQI) in 1976 for studies of Swedish Lakes (cited by Saether, 1979). The value of k_i (Table III-9) represents the empirical position of each species in the range from oligotrophic to eutrophic conditions. The indicator species used by Wiederholm were given the following values for k_i: 5, <u>Heterotrissocladius subpilosus</u> (Kieff.); 4, <u>Micropsectra</u> spp. and <u>Paracladopelma</u> spp., specifically <u>P. nigritula</u> (Goetgh.); 3, <u>Phaenospectra coracina</u> (Zett.) and <u>Stictochironomus rosenschoeldi</u> (Zett.); 2, <u>Chironomus anthracinus</u> (Zett.); 1, <u>Chironomus plumosus</u> L.; 0, absence of these indicator species. The BQI was related to total phosphorus/mean lake depth as shown in Figure III-4. The value of the index approaches 0 as the lakes become more eutrophic, and is nearly 5 in oligotrophic lakes. With the indicator species used here, the BQI applies to Palearctic lakes (e.g., Europe, Asia north of the Himalayas, Northern Arabia, Africa north of the Sahara). However, the species used as indicators may be redefined for Nearctic lake studies (e.g., lakes in Greenland, arctic America, northern and mountainous parts of North America) by using the species lists given in Appendix D.

The Trophic Condition Index (TCI) is the only commonly used index that was developed in North America specifically for lake studies. This index (Table III-9) was designed by Brinkhurst (1967) for use on Great Lakes waters. It is based on oligochaetes which are classified according to the degree of enrichment of the environments where they are typically found (Table III-10). The TCI ranges from 0 to 2, with the higher values associated with more eutrophic conditions.

In a study of Green Bay, Howmiller and Scott (1977) compared the TCI with four other indices. Only the Trophic Condition Index showed a significant difference between the three areas of Green Bay shown in Figure III-5. The other indices used were Species Diversity, Oligochaete worms per square meter, Oligochaete worms (%) and <u>L. hoffmesiteri</u> (%). As shown in Table III-11, these indices show no statistical difference between Areas II and III, and sometimes no significant difference from values for Area I.

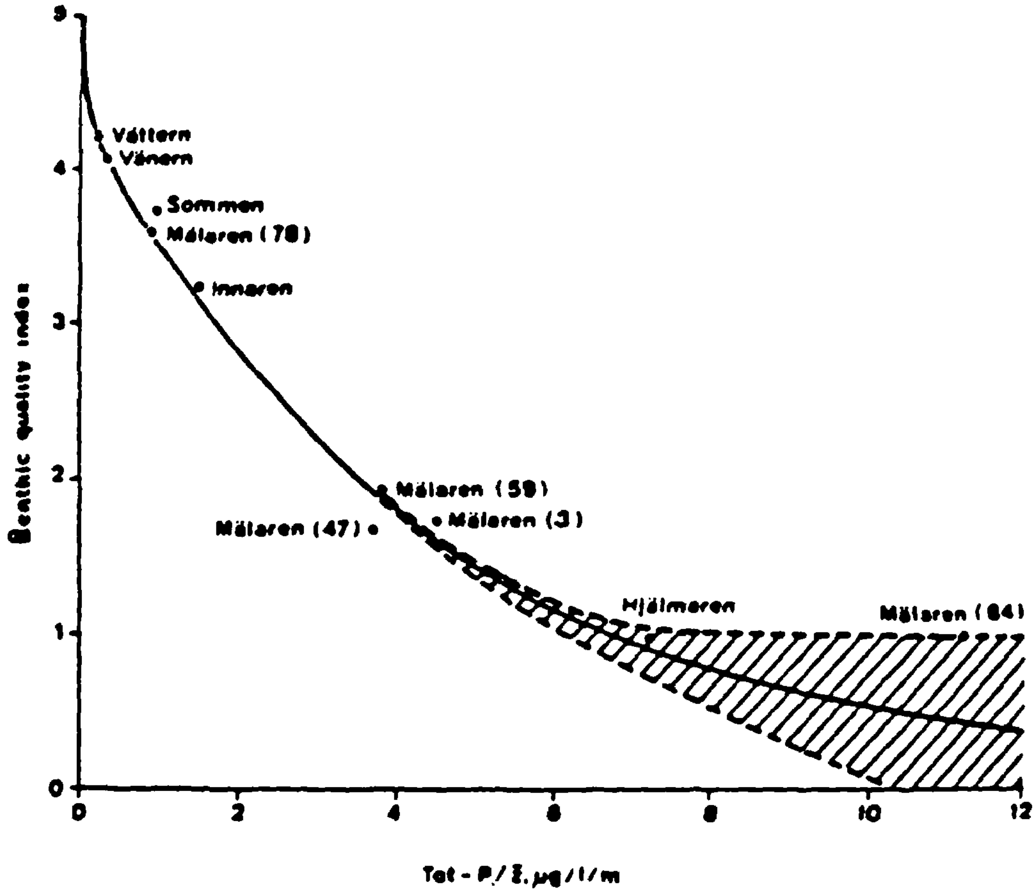


Figure III-4. Total phosphorus/mean lake depth in relation to a benthic quality index (BQI) based on indicator species of chironomids (From Wiederholm, 1980).

TABLE III-10

A CLASSIFICATION OF OLIGOCHAETE SPECIES
ACCORDING TO THE DEGREE OF ENRICHMENT OF THE ENVIRONMENTS
IN WHICH THEY ARE CHARACTERISTICALLY FOUND

Group 0

Species largely restricted to oligotrophic situations:

Stylodrilus heringianus
Peloscolex variegatus
P. superiorensis
Limnodrilus profundicola
Tubifex kessleri
Rhyacodrilus coccineus
R. montana

Group 1

Species characteristic of areas which are mestrophic or only slightly enriched:

Peloscolex ferox
P. freyi
Ilyodrilus templetoni
Potamothrix moldaviensis
P. vejdovskyi
Aulodrilus spp.
Arcteonais lomondi
Dero digitata
Nais elinguis
Slavina appendiculata
Uncinais uncinata

Group 2

Species tolerating extreme enrichment or organic pollution:

Limnodrilus anguistipenis
L. cervix
L. claparedeianus
L. hoffmeisteri
L. maumeensis
L. udekemianus
Peloscolex multisetosus
Tubifex tubifex

SOURCE: Howmiller and Scott, 1977.

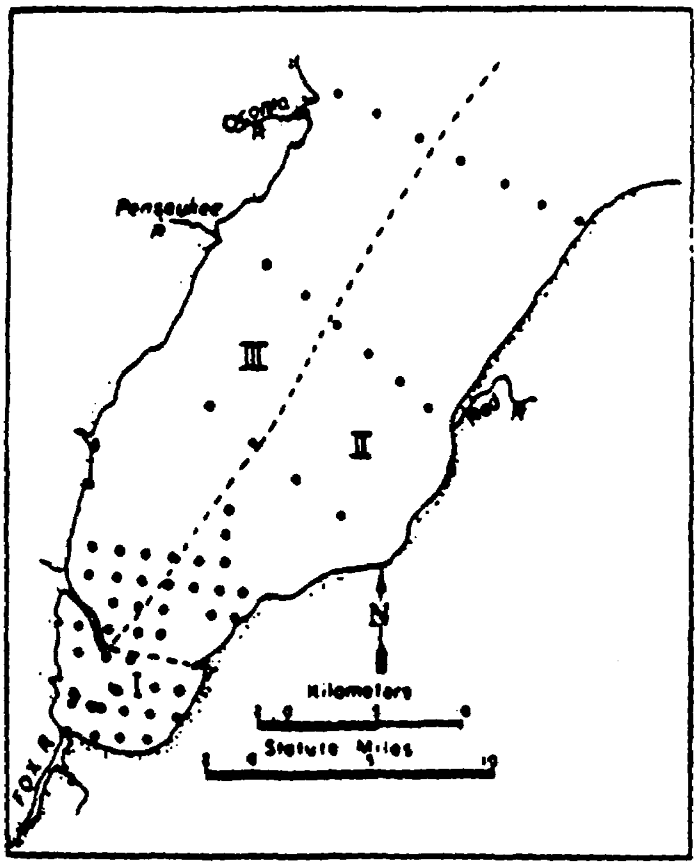


Figure III-5. Map of Lower and Middle Green Bay showing location of benthos sampling stations and areas designated I, II, and III (from Howmiller and Scott, 1977).

TABLE III-11

AVERAGE VALUES OF FIVE INDICES OF POLLUTION
COMPARED FOR THREE AREAS OF GREEN BAY

	Area		
	I	II	III
Species Diversity	1.00	1.62	1.66
Oligochaete worms/m^2	1085	1672	1152
Oligochaete worms, %	63	53	53
L. hoffmeisteri, %	73	50	42
Trophic index	1.92	1.84	1.53

NOTE: Values underscored with a common line are not
significantly different from each other.

SOURCE: Howmiller and Scott, 1977.

FISH

Although fish species in many instances show no preference for either lacustrine or riverine habitat, certain environmental components (e.g., velocity, substrate, dissolved oxygen and temperature) render one habitat more suitable than another. The following paragraphs highlight the habitat requirements of certain fish species that are predominantly lacustrine.

Trophic State Effects

Oligotrophic and eutrophic lakes have characteristic fish populations because of their contrasting habitats. Briefly, oligotrophic lakes are generally deep and often large in size, and are located in regions where the substratum is rocky. These lakes usually stratify in summer, but the cool profundal zone contains sufficient oxygen year-round for fish survival. Oligotrophic lakes support less than 20 pounds of fish per surface acre, and characteristic fish are salmons, trouts, chars, ciscoes, and graylings (Bennett, 1971).

Eutrophic lakes support fish populations of largemouth bass, white bass, white and black crappies, bluegill and other sunfish, buffalo, channel catfish, bullheads, carp, and suckers (Bennett, 1971). Such lakes have shallow to intermediate depths, may have large or small surface areas, and are located in regions with more fertile soil than oligotrophic lakes. Hypolimnetic waters of eutrophic lakes frequently exhibit reduced oxygen levels during summer stratification.

Nutrient enrichment which causes increased production in lakes accelerates the natural progression of trophic state from oligotrophy to eutrophy. Initially, eutrophication and the subsequent abundance of food organisms may cause increased growth of fish. However, undesirable conditions of temperature and dissolved oxygen in later stages force some fish to leave the affected area or perish. Fish commonly respond to changes associated with eutrophication by shifting their horizontal and vertical distribution. In Lake Erie, whitefish and ciscoes became restricted to the eastern basin as the environment became more unsuitable (Beeton, 1969). Perch and whitefish may move from the littoral zone into the pelagic zone, where they are not usually found (Larkin and Northcote, 1969). The restriction of coldwater fishes to a thin layer between the oxygen deficient hypolimnion and the warm epilimnion may lead to mortalities. This may have contributed to the disappearance of ciscoes from Lake Mendota, Wisconsin.

As eutrophication proceeds, there is a general pattern of change in fish populations from coregonines to coarse fish. One of the best examples of population changes is in the Great Lakes. Although factors other than eutrophication may have contributed to the loss of some species, enrichment is recognized as being an important cause. Commercial fisheries provide information on the species composition of catches. In Lake Erie, the major species in the 1899 catch were lake herring (cisco), blue pike, carp, yellow perch, sauger, whitefish and walleye. By 1940, the lake herring and sauger fisheries had collapsed, and the catch was dominated by blue pike, whitefish, yellow perch, walleye, sheepshead, carp, and suckers. Blue pike and whitefish populations have since declined, and the catch has become more concentrated on the warmwater species such as freshwater drum, carp, yellow perch and smelt (Beeton, 1969; Larkin and Northcote, 1969).

Temperature Effects

Temperature as well as trophic state plays a role in determining the fish species inhabiting a lake. Trout are generally considered representative of coldwater species. Rainbow trout and brook trout thrive in water with a maximum summer temperature of about 70°F. Rainbow trout are more tolerant of higher temperatures than brook trout. Prolonged exposure to temperatures of 77.5°F is lethal to brook trout (Bennett, 1971).

Fish typical of warmer waters include largemouth bass, bluegill, black and white crappie, and black and yellow bullhead. These species are fairly tolerant of high, naturally occurring, water temperatures, and generally suffer mortality only when additional adverse factors (e.g., anoxic conditions, toxics, thermal plumes) prevail. Species such as smallmouth bass, rock bass, walleye, northern pike, and muskellunge are more sensitive to increased temperatures than the more typical warmwater fish, but are not as sensitive as trout.

Warmwater fish and coldwater fish may live in the same lake. For example, a two-tier fishery may exist in a stratified lake, wherein warmwater fish live in the epilimnion and the metalimnion, while coldwater fish survive in the cooler waters of the hypolimnion.

Specific Habitat Requirements

Specific habitat requirements for some lake species are published in a series of documents (Habitat Suitability Index Models) prepared by the Fish and Wildlife Service and available through the National Technical Information Service. These publications summarize habitat suitability information for many lake species including: rainbow trout, longnose sucker, smallmouth buffalo, bigmouth buffalo, black bullhead, largemouth bass, yellow perch, green sunfish, and common carp. The following information on the habitat requirements of these species is contained within the Fish and Wildlife Service reports.

Rainbow Trout

Rainbow trout prefer cold, deep lakes that are usually oligotrophic. The size and chemical quality of the lakes may vary. Rainbow trout require streams with gravel substrate in riffle areas for reproduction. Spawning takes place in an inlet or outlet stream, and those lakes with no tributary streams generally do not support reproducing populations of rainbow trout. The optimal water velocity for rainbow trout redds is between 30 and 70 cm/sec. Juvenile lake rainbow trout migrate from natal streams to a freshwater lake rearing area.

Adult lake rainbow trout prefer temperatures less than 18°C, and generally remain at depths below the 18°C isotherm. They require dissolved oxygen levels greater than 3 mg/l (Raleigh, et al., 1984).

Longnose Sucker

This species is most abundant in cold, oligotrophic lakes that are 34-40 m deep. These lakes generally have very little littoral area. They are also capable of inhabiting swift-flowing streams, but longnose suckers in lake environments enter streams and rivers only to spawn or to overwinter. The longnose sucker spawns in riffle areas (velocity 0.3-1.0 m/sec), where the adhesive eggs are broadcast over clean gravel and rocks (Edwards, 1983a).

Smallmouth Buffalo

Although smallmouth buffalo typically inhabit large rivers, preferring deep, clear, warm waters with a current, they can do well in large reservoirs or lakes. Lake or reservoir populations spawn in embayments or along recently flooded shorelines. Although smallmouth buffalo will spawn over all bottom types, they prefer to spawn over vegetation and submerged objects. Juveniles frequent warm, shallow, vegetated areas with velocities less than 20 cm/sec. Adults are found in areas with velocities up to 100 cm/sec (Edwards and Twomey, 1982a).

Bigmouth Buffalo

Bigmouth buffalo prefer low velocity areas (0-70 cm/sec), and inhabit large rivers, lowland lakes and oxbows, and reservoirs. Populations in reservoirs reside in warm, shallow, protected embayments during the summer, and move into deeper water in the fall and winter. Fluctuations of reservoir water levels reduce buffalo populations due to siltation, erosion and loss of vegetation (Edwards, 1983b).

Black Bullhead

Bullheads live in both riverine and lacustrine environments. Optimal lacustrine habitat has an extensive littoral area (more than 25 percent of the surface area), with moderate to abundant (more than 20 percent) cover within this area. Bullhead nests are located in weedy areas at depths of 0.5-1.5 m. Black bullheads are most common in areas of low velocity (less than 4 cm/sec). They prefer intermediate levels of turbidity (25-100 ppm), and can withstand low dissolved oxygen levels (as low as 0.2-0.3 mg/l in winter, 3.0 mg/l in summer) (Stuber, 1982).

Largemouth Bass

Largemouth bass prefer lacustrine environments. Optimal habitats are lakes with extensive shallow areas (more than 25 percent of the surface area less than 6 m depth) for growth of submergent vegetation, but deep enough (3-15 m) to successfully overwinter bass. Current velocities below 6 cm/sec are optimal, and velocities above 20 cm/sec are unsuitable. Temperatures from 24-30°C are optimal for growth of adult bass. Largemouth bass will nest on a variety of substrates, including vegetation, roots, sand, mud, and cobble, but they prefer to spawn on a gravel substrate. Adult bass are considered intolerant of suspended solids; growth and survival of bass is greatest in low turbidity waters (less than 25 ppm suspended solids). Bass show signs of stress at oxygen levels of 5 mg/l, and DO concentrations less than 1.0 mg are lethal (Stuber, et al., 1982a).

Yellow Perch

Yellow perch prefer areas with sluggish currents or slack water. They frequent littoral areas in lakes and reservoirs, where there are moderate amounts of vegetation present. Riverine habitat resembles lacustrine areas, with pools and slack-water. Perch spawn in depths of 1.0 m to 3.7 m, and in waters of low (less than 5 cm/sec) current velocity. Littoral areas of lakes and reservoirs provide both spawning habitat and cover (Krieger, et al., 1983).

Green Sunfish

Green sunfish thrive in both riverine and lacustrine environments. Optimal lacustrine environments are fertile lakes, ponds, and reservoirs with extensive littoral areas (more than 25 percent of the surface area). Preferred environmental parameters are: velocities less than 10 cm/sec, moderate turbidities (25-100 JTU) and DO levels of more than 5 mg/l (lethal levels of 1.5 mg/l) (Stuber, et al., 1982b).

Common Carp

This species prefers areas of slow current. In both riverine and lacustrine environments, carp prefer enriched, relatively shallow, warm, sluggish and well-vegetated waters with a mud or silty substrate. Adults are generally found in association with abundant vegetation. The common carp is extremely tolerant of turbidity and its own feeding and spawning activities over silty bottoms increase turbidity. Adults are also tolerant of low dissolved oxygen levels, and can gulp surface air when the dissolved oxygen is less than 0.5 mg/l (Edwards and Twomey, 1982b).

Stocking

The most common fish management technique used is stocking. The purpose of stocking is to improve the fish population, and certain fish are used more often than others. The following description is based on information in Bennett (1971).

Bass and bluegills have often been stocked in the same pond or lake. The theory behind stocking these species in combination is that both largemouth bass and bluegills would be available for sport-fishing. The role of the bluegills is to convert invertebrates into bluegill flesh. The bass then feed on small bluegills and thereby control the population. Problems may be caused from an overpopulation of one species, especially since the bluegills overpopulate more often than the bass. Stocking ratios (numbers of bass : numbers of bluegills) as discussed by Bennett (1971), influence the outcome of such stocking endeavors.

Because largemouth, smallmouth, and spotted bass are omnivorous, any of these three species stocked alone may be fairly successful. They feed on crayfish, large aquatic insects and their own young. These species do well in warmwater ponds if they do not have to compete with prolific species such as bluegills, green sunfish, and black bullheads. Largemouth bass have been stocked in warmwater ponds in combination with minnows, chubsuckers, red-ear sunfish or warmouths. These combinations have proved to be successful.

Walleye stocking reportedly has variable success except in waters devoid of other fishes. In waters such as new reservoirs and renovated lakes, satisfactory survival rates for walleye occur. Bennett (1971) noted that, generally, walleye stocking was unsuccessful in acid or softwater lakes.

CHAPTER IV

SYNTHESIS AND INTERPRETATION

INTRODUCTION

The basic physical and chemical processes of the lake were introduced in Chapter II. Chapter II also includes a discussion of desktop procedures that might be used to characterize various lake properties, and a discussion of mathematical models that are suitable for the investigation of various physical and chemical processes.

The applicability of desktop analyses or mathematical models will depend upon the level of sophistication desired for a use attainability study. Case studies were presented to illustrate the use of measured data and model projections in the use attainability study. The selection of a reference site is discussed later in Chapter IV.

Chapter II also provides a discussion of chemical phenomena that are of importance in lake systems. Most important of these are the processes that control internal phosphorus cycling, and the processes that control dissolved oxygen levels in the epilimnion and the hypolimnion of a stratified lake. Chemical evaluations are also discussed in the earlier Technical Support Manuals (U.S. EPA, 1983b, 1984).

The biological characteristics of the lake are summarized in Chapter III. Specific information on plant, fish and macroinvertebrate lake species is presented to assist the investigator in determining aquatic life uses.

The emphasis in Chapter IV is placed on a synthesis of the physical, chemical and biological evaluations which will be performed to permit an overall assessment of aquatic life protection uses in the lake. A large portion of this discussion is devoted to lake restoration considerations.

Like the two previous Technical Support Manuals (U.S. EPA, 1983b, 1984), the purpose of this Manual is not to specifically describe how to conduct a use attainability analysis. Rather, it is the desire of EPA to allow the states some latitude in such assessments. This Manual provides technical support by describing a number of physical, chemical, and biological evaluations, as well as background information, from which a state may select assessment tools to be used in a particular use attainability analysis.

USE CLASSIFICATIONS

There are many use classifications--navigation, recreation, water supply, the protection of aquatic life--which might be assigned to a water body. These need not be mutually exclusive. The water body survey as discussed in this volume is concerned only with aquatic life uses and the protection of aquatic life in a lake.

The objectives in conducting a use attainability survey are to identify:

1. The aquatic life use currently being achieved in the water body;

2. The potential uses that can be attained, based on the physical, chemical and biological characteristics of the water body; and

3. The causes of any impairment of uses.

The types of analyses that might be employed to address these three points are listed in Table IV-1. Most of these are discussed in detail in this volume, and in the two preceding volumes on estuaries and on rivers and streams.

Use classification systems vary widely from state to state. Use classes may be based on salinity, recreation, navigation, water supply (municipal, agricultural, or industrial), or aquatic life. In some cases geography serves as the basis for use classifications. Aquatic life use classifications found in state standards generally are rather broad (e.g., coldwater fishery, warmwater fishery, fish maintenance, protection of aquatic life, etc.) and offer little specificity. Clearly, little information is required to place a water body into such broad categories.

Far more information may be gathered in a water body survey than is needed simply to assign a classification that is drawn from available state classifications. The additional data that is gathered is required, nevertheless, in order to evaluate management alternatives for the lake and, if appropriate, to refine state use classification systems for the protection of aquatic life.

In general, state water quality standards do not address lakes specifically, so one must assume that standards written to cover surface waters in some states, or rivers and streams in others, are intended to stand for lakes as well. From the standpoint of aquatic life protection uses this may be satisfactory since the types of fish found in lakes are also found in the streams that discharge into lakes. However, the fact that some lakes stratify and others do not suggests that seasonal aquatic life uses in a lake could be more complex than in adjacent streams. In highly stratified lakes, for example, the fish population of the epilimnion might be substantially different from that of the hypolimnion. That a shallow lake may become anoxic during summer stratification may have important implications for the uses of the hypolimnion. That the epilimnion may become anoxic because of diurnal DO fluctuations due to massive algal blooms and decay also has implications for the definition of present and future uses.

Since there may not be an adequate spectrum of aquatic protection use categories available against which to compare the findings of the biological survey; and since the objective of the survey is to compare existing uses with designated uses, and existing uses with potential uses, as seen in the three points listed above; the investigators may need to develop their own system of ranking the biological health of a water body (whether qualitative or quantitative) in order to satisfy the intent of the water body survey. Implicit to the use attainability survey is the development of

TABLE IV-1

SUMMARY OF TYPICAL WATER BODY EVALUATIONS

PHYSICAL EVALUATIONS	CHEMICAL EVALUATIONS	BIOLOGICAL EVALUATIONS
o Size (mean width/depth)	o Dissolved oxygen	o Biological inventory (existing use analysis)
o Flow/velocity	o Nutrients	o Fish
o Total volume	- nitrogen	o Macroinvertebrates
o Reaeration rates	- phosphorus	o Microinvertebrates
o Temperature	o Chlorophyll-a	o Plants
o Suspended solids	o Sediment oxygen demand	- phytoplankton
o Sedimentation	o Salinity	- macrophytes
o Bottom stability	o Hardness	o Biological condition/ health analysis
o Substrate composition and characteristics	o Alkalinity	- diversity indices
o Sludge/sediment	o pH	- primary productivity
o Riparian characteristics	o Dissolved solids	- tissue analyses
o Downstream characteristics	o Toxics	- Recovery Index
		o Biological potential analysis
		o Reference reach comparison

SOURCE: Adapted from EPA 1982a, Water Quality Standards Handbook management strategies or alternatives which might result in enhancement of the biological health of the water body. A clear definition of uses is necessary to weigh the predicted results of one strategy against another in cases where the strategies are defined in terms of protection of aquatic life.

Since one may very well be seeking to define use levels within an existing use category, rather than describe a shift from one use classification to another, the existing state use classifications may not be helpful. Therefore, it may be necessary to develop an internal use classification system to serve as a yardstick during the course of the water body survey, which may later be referenced to the legally constituted use categories of the state.

A scale of biological health classes is presented in Table IV-2 that offers general categories against which to assess the biology of a lake. A descriptive scale is found in Table IV-3 that may be used to assess a water body. This scale was developed by EPA in conjunction with the National Fisheries Survey.

REFERENCE SITES

Selection

Chapter IV-6 of the Technical Support Manual (U.S. EPA, 1983b) presents a detailed discussion on the concept of ecological regions and the selection of regional reference sites. This process is particularly applicable to small and medium size lakes. Use attainability studies for very large lakes are more likely to be concerned with specific segments of the lake than with the lake in its entirety. Resource requirements are an important consideration as well for very large lakes. For example, New York State may be prepared to investigate uses in Lake Ontario near Buffalo, but may not be prepared to study the entire lake. A study of this magnitude could not be done without federal participation, or in the case of Lake Ontario or Lake Erie, international participation. For the scale of study that a state may embark upon, reference sites could well be segments of the same or other large lakes.

The concept of developing ecological regions that are relatively homogeneous can be applied to lakes. This concept is based on the assumption that similar ecosystems occur in definable geographic patterns. Although the biota of particular lakes in close proximity may vary, it is more likely to be similar in a given region than in geographically dissimilar regions.

Within each region various lakes are investigated to determine which sites have a well balanced ecosystem and to note watershed land use and land cover characteristics and the effects of man's activities. A major characteristic to look for in the selection of a reference lake is the level of disturbance in the watershed that feeds the lake. Good reference site candidates are lakes located away from heavily populated areas, such as in protected park land.

TABLE IV-2

BIOLOGICAL HEALTH CLASSES WHICH COULD BE USED
IN WATER BODY ASSESSMENT

Class	Attributes
Excellent	Comparable to the best situations unaltered by man; all regionally expected species for the habitat including the most intolerant forms, are present with full array of age and sex classes; balanced trophic structure.
Good	Fish invertebrate and macroinvertebrate species richness somewhat less than the best expected situation; some species with less than optimal abundances or size distribution; trophic structure shows some signs of stress.
Fair	Fewer intolerant forms of plants, fish and invertebrates are present.
Poor	Growth rates and condition factors commonly depressed; diseased fish may be present. Tolerant macroinvertebrates are often abundant.
Very Poor	Few fish present, disease, parasites, fin damage, and other anomalies regular. Only tolerant forms of macroinvertebrates are present.
Extremely Poor	No fish, very tolerant macroinvertebrates, or no aquatic life.

SOURCE: Modified from Karr, 1981

TABLE IV-3

AQUATIC LIFE SURVEY RATING SYSTEM

A water body that is rated a five has:

- A fish community that is well balanced among the different levels of the food chain.
- An age structure for most species that is stable, neither progressive (leading to an increase in population) or regressive (leading to a decrease in population).
- A sensitive sport fish species or species of special concern always present.
- Habitat which will support all fish species at every stage of their life cycle.
- Individuals that are reaching their potential for growth.
- Fewer individuals of each species.
- All available niches filled.

A water body that is rated a four has:

- Many of the above characteristics but some of them are not exhibited to the full potential. For example, the water body has a well balanced fish community; the age structure is good; sensitive species are present; but the fish are not up to their full growth potential and may be present in higher numbers; an aspect of the habitat is less than perfect (i.e., occasional high temperatures that do not have an acute effect on the fish); and not all food organisms are available or they are available in fewer numbers.

A water body that is a three has:

- A community is not well balanced, one or two trophic levels dominate.
- The age structure for many species is not stable, exhibiting regressive or progressive characteristics.
- Total number of fish is high, but individuals are small.
- A sensitive species may be present, but is not flourishing.
- Other less sensitive species make up the majority of the biomass.
- Anadromous sport fish infrequently use these waters as a migration route.

A water body that is rated a two has:

- Few sensitive sport fish are present, nonsport fish species are more common than sport fish species.
- Species are more common than abundant.
- Age structures may be very unstable for any species.
- The composition of the fish population and dominant species is very changeable.
- Anadromous fish rarely use these waters as a migration route.
- A small percent of the reach provides sport fish habitat.

A water body that is a one has:

- The ability to support only nonsport fish. An occasional sport fish may be found as a transient.

A water body that is rated a zero has:

- No ability to support a fish of any sort, an occasional fish may be found as transient.

For the selection of a reference lake, it is important to seek comparability in physical parameters such as surface area, volume, and mean depth, and in physical processes such as degree of stratification and sedimentation characteristics. It will be important also to seek comparability in detention time, which plays a role in determining the chemical and biological characteristics of the lake. Detention time is determined by lake volume and rate of flow into the lake from both point and nonpoint sources.

The selection of a candidate reference lake could be based on an analysis of existing data. Data for many lakes throughout the country are available from the National Eutrophication Survey conducted by the U.S. EPA in cooperation with state and local agencies. National computerized data bases such as WATSTORE and STORET can provide flow and water quality data. Many states and counties have their own water quality and biological monitoring programs which should be used to obtain the most up-to-date information on the lake.

In addition to the historical data that may be available through WATSTORE or the National Eutrophication Survey, it is very important to obtain current information on a lake in order to evaluate its present characteristics. One must be careful to note trends that may have occurred over time so as to fully understand the extent to which the reference lake represents natural conditions.

<u>Comparison</u>

The reference site will have been selected on the basis of physical similarity with the study area, and upon the determination that it reflects natural conditions or conditions as close to natural as can be found. Subsequent comparisons for the purpose of describing attainable uses will be based on comparisons of the chemical and biological properties of the two water bodies. Similarities and differences in chemical and biological characteristics can be examined to identify causes of use impairment, and potential uses can be determined from an analysis of the lake's response to the abatement of the identified causes of impairment.

Comparisons of individual chemical and biological parameters can be made by using simple statistics such as mean values and ranges for the entire data base or that part of the data base which is considered appropriate to reflect present conditions. Seasonal and monthly statistics can also be used for lakes which demonstrate major changes throughout the year.

In addition to individual parameters, water quality and biological indices are useful for comparisons. Water quality indices summarize a number of water quality characteristics into a single numerical value which can be compared to standard values that are indicative of a range of conditions. The National Sanitation Foundation index, the Dinius water quality index, and the Harkins/Kendall water quality index, each of which may provide insight into the study site, are discussed in Chapter III of the <u>Technical Support Manual</u> (U.S. EPA, 1983<u>b</u>).

Biological indices to be considered include: diversity indices which evaluate richness and composition of species; community comparison indices which measure similarities or dissimilarities between entire communities; recovery indices which indicate the ability of an ecosystem to recover from pollutant stress; and the Fish and Wildlife Service Habitat Suitability Index which examines species habitat requirements. These indices are discussed in detail in Chapter IV of the Technical Support Manual (U.S. EPA, 1983b). Another useful tool which is described in that Manual is cluster analysis, which is a technique for grouping similar sites or sampling stations on the basis of the resemblance of their attributes (e.g., number of taxa and number of individuals).

Statistical tests can be used to determine whether water quality or any other use attainment indicator at the study site is significantly different from conditions at the reference site or sites. Several of these tests are described in Volumes I and II of the Technical Support Manual (U.S. EPA, 1983b, 1984).

CURRENT AQUATIC LIFE PROTECTION USES

The actual aquatic life protection uses of a water body are defined by the resident flora and fauna. The prevailing chemical and physical attributes will determine what biota may be present, but little need be known of these attributes to describe current uses. The raw findings of a biological survey may be subjected to various measurements and assessments, as discussed in Section IV (Biological Evaluations) of the Technical Support Manual (U.S. EPA, 1983b). After performing an inventory of the flora and fauna (preferably an historical inventory to reflect seasonal changes) and considering diversity indices or other measures of biological health, one should be able to adequately describe the condition of the aquatic life in the lake.

CAUSES OF IMPAIRMENT OF AQUATIC LIFE PROTECTION USES

If the biological evaluations indicate that the biological health of the system is impaired relative to a "healthy" reference aquatic ecosystem (as might be determined by reference site comparisons), then the physical and chemical evalutions can be used to pinpoint the causes of that impairment. Figure IV-1 shows some of the physical and chemical parameters that may be affected by various causes of change in a water body. The analysis of such parameters will help clarify the magnitude of impairments to attaining other uses, and will also be important to the third step in which potential uses are examined.

ATTAINABLE AQUATIC LIFE PROTECTION USES

A third element to be considered is the assessment of potential uses of the water body. This assessment would be based on the findings of the physical, chemical and biological information which has been gathered, but additional study may also be necessary. A reference site comparison will be particularly important. In addition to establishing a comparative baseline community, the reference site provides insight into the aquatic life that could potentially exist if the sources of impairment were mitigated or removed.

SOURCE OF MODIFICATION

Stream Parameters	Acid Mine Drainage or Acid Precipitation	Sewage Treatment Plant Discharge (primary or secondary)	Agricultural Runoff (pasture or cropland)	Urban Runoff	Channelization	INDUSTRIES: Pulp and Paper	Textile	Metal Finishing and Electroplating	Petroleum	Iron and Steel	Paint and Ink	Dairy and Meat Products	Fertilizer Production and Lime Crushing	Plastics and Synthetics
pH	D					C	I	C		D	C		D,I	C
Alkalinity	D						I						D,I	
Hardness	I						I						I	
Chlorides		I		I								I		
Sulfates	I								I	I			I	I
TDS	I						I		I		I		I	
TKN		I	I	I					I	I		I	I	I
NH_3-N		I							I	I			I	I
Total-P		I	I	I				I				I	I	
Ortho-P		I		I				I					I	
BOD_5		I				I	I	I	I	I	I		I	
COD	I	I		I		I	I			I	I	I		I
TOC		I	I	I		I			I	I	I	I		I
COD/BOD_5	I			I		I	I		D	I				
D.O.		D				D						D		
Aromatic Compounds			I	I		I			I					I
Fluoride								I					I	I
Cr				I		I	I	I	I		I			I
Cu	I			I		I		I			I			I
Pb				I				I	I	I	I			
Zn	I			I		I		I	I	I	I			I
Cd				I				I			I			I
Fe	I			I				I		I	I		I	I
Cyanide										I				I
Oil and Grease						I	I	I	I	I	I	I		I
Coliforms	D	I	I	I				D		D	D	I		
Chlorophyll	D	I	I		I	D		D	D	D	D	I	I	D
Diversity	D	D		D	D	D	D	D	D	D	D			D
Biomass	D	I	I		I		I	D	D	D	D	I	I	D
Riparian Characteristics					C									
Temperature					I									
TSS			I	I	I	I	I	I			I		I	I
VSS				I		I								
Color						I	I				I	I		I
Conductivity	I												I	
Channel Characteristics					C									

TABLE IV-1. Potential Effects of Some Sources of Alteration on Stream Parameters; D = decrease, I = increase, C = change.

The analysis of all information that has been assembled may lead to the definition of alternative strategies for the management of the lake at hand. Each such strategy corresponds to a unique level of protection of aquatic life, or aquatic life protection use. If it is determined that an array of uses is attainable, further analysis which is beyond the scope of the water body survey would be required to select a management program for the lake.

One must be able to separate the effects of human intervention from natural variability. Dissolved oxygen, for example, may vary seasonally over a wide range in some areas even without anthropogenic effects, but it may be difficult to separate the two in order to predict whether removal of the anthropogenic cause will have a real effect. The impact of extreme storms on a water body, such as the effect of Hurricane Agnes on Pennsylvania lakes and streams in 1972, may completely confound our ability to distinguish the relative impact of anthropogenic and natural influences on immediate effects and long term trends. In many cases the investigator can only provide an informed guess.

If a lake and stream system does not support an anadromous fishery because of dams and diversions which have been built for water supply and recreational purposes, it is unlikely that a concensus could be reached to restore the fishery by removing the physical barriers--the dams--which impede the migration of fish. However, it may be practical to install fish ladders to allow upstream and downstream migration. Another example might be a situation in which dredging to remove toxic sediments may pose a much greater threat to aquatic life than to do nothing. Under the do nothing alternative, the toxics may remain in the sediment in a biologically-unavailable form, whereas dredging might resuspend the toxic fraction, making it biologically available while facilitating wider distribution in the water body.

The points touched upon above are presented to suggest some of the phenomena which may be of importance in a water body survey, and to suggest the need to recognize whether or not they may realistically be manipulated. Those which cannot be manipulated essentially define the limits of the highest potential use that might be realized in the water body. Those that can be manipulated define the levels of improvement that are attainable, ranging from the current aquatic life uses to those that are possible within the limitations imposed by factors that cannot be manipulated.

PREVENTIVE AND REMEDIAL TECHNIQUES

Uses that have been impaired or lost can only be restored if the conditions responsible for the impairment are corrected. In most cases, impairment in a lake can be attributed to toxic pollution or nutrient overenrichment. Uses may also be lost through such activities as the disposal of dredge and fill materials which smother plant and animal communities, through overfishing which may deplete natural populations, and the destruction of freshwater spawning habitat which will cause the demise of various fish species. One might expect losses due to natural phenomena to be temporary although man-made alterations of the environment may preclude restoration by natural processes.

Assuming that the factors responsible for the loss of species have been identified and corrected, efforts may be directed toward the restoration of habitat followed by natural repopulation, stocking of species if habitat has not been harmed, or both. Many techniques for the improvement of substrate composition in streams have been developed which might find application in lakes as well. Further discussion on the importance of substrate composition will be found in the Technical Support Manual (U.S. EPA, November 1983b).

The U.S. EPA National Eutrophication Study and companion National Eutrophication Research Program resulted in the development and testing of a number of lake restoration techniques. In the material to follow, an overview is provided of a number of projects sponsored by the U.S. EPA in which these techniques were applied. This is an overview that is not intended to be exhaustive in detail. For further information, the reader is referred to a manual on lake restoration techniques that is currently in preparation by U.S. EPA and the North American Lake Management Society.

Dredging

Introduction

Dredging to remove sediments from lakes has several objectives: to deepen the lake, to remove nutrients associated with sediment, to remove toxics trapped in bottom sediment, and to remove rooted aquatic plants. Dredged lakes generally show improved aesthetics, and often enjoy improved fish habitat as shown by increased growth of fish (Peterson, 1981). The following sections summarize the objectives of lake dredging programs, the environmental concerns associated with sediment removal, and the methods used in implementing dredging projects.

Lake Conditions Most Suitable for Sediment Removal. Dredging to improve lake conditions is better suited for some lakes than others. Obviously, a lake with a sediment-filled basin is a prime candidate for dredging. Other considerations are lake size, the presence of toxics in the sediment, dredging cost, and sedimentation rate. Toxics are of concern because they may be released to the water column during the dredging operation. Because of dredging costs, the dredging of large areas is not feasible. Lakes that have been dredged in whole or in part range in size from 2 hectares (ha) to 1,050 ha (Peterson, 1981).

The practicality of sediment removal as a lake restoration technique also depends on the depth of sediment to be removed. Lakes with surface sediment that is highly enriched relative to underlying sediment are best suited for dredging projects. Dredging will not be cost effective in lakes with high sedimentation rates. The effect of sediment removal lasts longer in water bodies with smaller ratios of watershed area to lake surface area (Peterson, 1981). One other consideration in dredging projects is the disposal of the dredged material. "Clean" sediment may be sold as landfill to offset the cost of dredging. However, the disposal of contaminated sediment may add considerably to the overall cost of the restoration program.

Purpose

Lakes in colder sections of the United States require a mean depth of about 4.5 m or greater to avoid winter fish kills; thus, lake deepening projects may help assure fish survival (Peterson, 1981). Removal of sediment containing high concentrations of nutrients helps to control algal growth. The resultant decreased algal growth is also beneficial for fish populations. These purposes are explained in greater detail in the following sections. Examples of lakes that have been dredged for the aforementioned purposes are summarized in a separate section, Case Histories.

Removal of Nutrients. The primary nutrient of concern in dredging operations is phosphorus. Removal of enriched sediment reduces the internal phosphorus load, as internal phosphorus cycling can amount to a major portion of the total loading. Peterson (1981) cited these examples of lakes in which a large percentage of the total phosphorus was attributed to internal sources:

(1) Linsley Pond, Connecticut--internal phosphorus was about 45 percent of the total phosphorus loading (Livingston and Boykin, 1962);

(2) Long Lake, Washington--phosphorus loading from sediment was 25-50 percent of the external loading (Welch, et al., 1979); and

(3) White Lake, Michigan--about 40 percent of the total phosphorus loading was contributed by sediment phosphorus regeneration (Jones and Bowser, 1978).

Because such large amounts of phosphorus are found within the sediments, dredging may be a feasible means by which to greatly reduce internal loading.

Lake Deepening. Summer stratification and vertical mixing characteristics change with increasing depth. In addition, a larger volume of hypolimnetic water, and a larger quantity of dissolved oxygen, are present in deeper lakes (Stefan and Hanson, 1981). Therefore, assuming identical rates of benthic oxygen uptake per unit area, the hypolimnion of a shallow lake will be depleted sooner than the hypolimnion of a deeper lake. Summer overturn due to wind-induced mixing may be frequent in shallow lakes. Therefore, dredging to increase depth may help to reduce the frequency of overturn.

Increased lake volume may also help reduce water temperature. Reduced water temperature increases oxygen solubility and decreases metabolic rates of organisms. Therefore, algal growth rates and hypolimnetic oxygen depletion may be slowed (Stefan and Hanson, 1981).

Removal of Toxics. The bottom sediment may be a sink for toxic and hazardous materials as well as nutrients. Toxics in sediments pose a potentially serious problem, although there is a paucity of information concerning the direct effects of contaminated sediment on organisms. Another major concern about sediments containing toxics is the possible introduction of toxics into the food web, and the bioaccumulation and biomagnification of toxics that may follow.

Macrophyte Removal. Rooted aquatic macrophytes can be removed by dredging. Aquatic plants are most often removed for reasons of aesthetics or interference with recreational uses. However, the role of macrophytes in internal nutrient cycling also justifies their removal. Barko and Smart (1980) demonstrated that *Egeria densa*, *Hydrilla verticillata*, and *Myriophyllum spicatum* could obtain their phosphorus nutrition exclusively from the sediments. When the plants die and decompose, nutrients in soluble form may be released to the water column, or be returned to the sediments as particulate matter.

Some researchers contend that healthy aquatic macrophytes obtain nutrients from the sediment and excrete them to the surrounding water (Twilley, et al., 1977; Carignan and Kalff, 1980). There is considerable evidence to show that large quantities of nutrients are recycled to the lake when plants die and decay (Barko and Smart, 1980; Landers, 1982). Landers (1982) found that senescing stands of *Myriophyllum spicatum* contained up to 18 percent of the annual total phosphorus loading in an Indiana reservoir. Because aquatic macrophytes cause mobilization of nutrients from the soil, their removal is a key to reducing the internal phosphorus load.

Environmental Concerns of Lake Dredging

Many of the environmental problems caused by dredging are associated with resuspension of fine particulates. Increased turbidity reduces light penetration; consequently, photosynthesis and phytoplankton production are inhibited. Suspended sediments absorb radiation from the sun and transform it into heat, thereby increasing the water temperature. Increases in temperature affect the metabolic rate of organisms, in addition to reducing the oxygen-holding capacity of the water. Dredging may also cause increased nutrient levels in the water column, and potentially favorable conditions for algal blooms (Peterson, 1981).

Toxic substances may also be liberated during dredging operations. For example, the aldrin concentration in Vancouver Lake, Washington, was 0.012 mg/l prior to dredging and increased by three times at one site and ten times at another site during dredging (Peterson, 1979). Return flow from settling ponds reached even higher concentrations, at times up to 0.336 mg/l.

Resuspended organic matter may present a different type of problem. Rapid decomposition may deplete the available dissolved oxygen. This may be especially important since the organic content of lake sediments can reach 80 percent on a dry weight basis (Wetzel, 1975). Although Peterson (1981) noted that no lake dredging projects have caused this problem, the potential should be recognized.

Implementation of Lake Dredging Projects

Sediment Removal Depth. After it has been determined that sediment removal is a viable lake restoration technique, a removal depth and method must be selected. Sediment removal depth has been determined by several different methods. The following paragraphs briefly describe two methods by which to determine removal depth.

Sediment Characterization. Studies of chemical and physical characteristics of a lake bottom may show distinct stratification of sediment. The greatest concentration of nutrients may be in a single layer, so that removal of the layer will significantly affect the internal nutrient loading. The sediment removal depth may be determined on the basis of nutrient content and release rates for the layers of sediment.

For example, sediment in Lake Trummen, Sweden, was characterized chemically and physically, horizontally and vertically. The study showed a definite layer of FeS-colored (black) fine sediment deposited on a brown layer. Based on aerobic and anaerobic release rates of PO_4-P and NH_4^+-N, it was decided that the black layer would be removed (Peterson, 1981). Born (1979) noted that the ecosystem of Lake Trummen was restored following dredging.

Lake Simulation. Another approach to determining sediment removal depth uses a lake model to predict the lake depth necessary to prevent summer destratification (Stefan and Hanson, 1980). This method of computation is generally used for shallow lakes.

Stefan and Hanson (1981) modeled the Fairmont Lakes, Minnesota, to determine the lake depth that would be required to prevent phosphorus recirculation from the sediments. Using air temperature, dew point temperature, wind direction, solar radiation, and wind speed, plus a consideration of lake morphology, the model predicts temperature with depth. Lake simulation helps determine the appropriate temperature and, therefore, minimum depth for stable seasonal stratification. This method of determining removal depth is based on the concept that shallow eutrophic lakes can be dredged to such a depth that a stable system is formed. In theory, phosphorus released from the sediment into the hypolimnion will be recycled to the photic zone with diminished frequency. By controlling and reducing the phosphorus concentration of the epilimnion, the standing crop of algae will be decreased. The simulation results agreed with the hypothesis of phosphorus release and recycling and the anticipated effects of dredging (Stefan and Hanson, 1981).

The method of lake simulation does not consider sediment release rates. Removal of the upper sediment layer may reduce nutrient levels in the overlying water even though stratification is not stable. Therefore, sediment release rates should also be examined along with the modeling approach (Peterson, 1981).

Dredging Equipment. Barnard (1978) and Peterson (1979) describe various dredges including the Mud Cat, the Bucket Wheel, and others, and their advantages and disadvantages. The reader should refer to these sources, especially Barnard (1978), for more detailed information.

The typical dredges are grab, bucket, and clamshell dredges which are generally operated from a barge-mounted crane. These systems remove sediment at nearly its in-site density, but removal volumes are limited to less than 200,000 m^3. Turbidity is created due to bottom impact of the bucket, the bucket pulling free from the bottom, bucket overflow and leakage both below and above the water surface, and the intentional overflow of water from receiving barges to increase the solids content.

Cutterhead dredges are the most commonly used in the United States. The cutterhead dredge removes material in a slurry that is 10 to 20 percent solids. These hydraulic dredges can remove larger volumes of sediment than bucket dredges. Turbidity from hydraulic dredges is largely dependent on pumping techniques and cutterhead configuration, size and operation.

Sediment Disposal. Dredged material disposal must also be considered in sediment removal projects. Fill permits are required for the filling of low-lying areas when the area exceeds 4.0 ha (10 acres) (Section 404, Public Law 92-500).

Upland disposal sites, which do not require Federal permits, commonly employ dikes to retain dredged material. Dike failure and underdesigned capacity are two major problems with upland disposal areas.

Several documents prepared by the U.S. Army Corps of Engineers contain useful information about dredged material disposal. They include: Treatment of Contaminated Dredged Material (Barnard and Hand, 1978), Evaluation of Dredged Material Pollution Potential (Brannon, 1978), Confined Disposal Area Effluent and Leachate Control (Chen, et al., 1978), Disposal Alternatives for Contaminated Dredged Material as a Management Tool to Minimize Adverse Environmental Effects (Gambrell, et al., 1978), Upland and Wetland Habitat Development with Dredged Material: Ecological Considerations (Lunz, et al., 1978), Guidelines for Designing, Operating, and Managing Dredged Material Containment Areas (Palermo, et al., 1978), and Productive Land Use of Dredged Material Containment Areas (Walsh and Malkasain, 1978).

Lake Dredging Case Studies

Peterson (1981) lists 64 sediment removal projects in the United States that are in various stages of implementation. Several of these projects will be considered in more detail in the following section.

Lilly Lake, Wisconsin. Lilly Lake has a surface area of 35.6 ha, a maximum depth of 1.8 m and a mean depth of 1.4 m. The main problem in Lilly Lake was excessive macrophyte growth, resulting in an accumulation of organic detritus and bottom sediment. Macrophytes also curtailed recreational activities such as boating and fishing. Winter fish kills were common in Lilly Lake.

Dredging began in July 1978 and continued through October of the same year. During dredging operations, the 5-day BOD increased by 1-2 mg O_2/liter, and turbidity rose by 1-3 formazin units. Ammonia concentration increased from 0.01 mg/liter to a high of 5.5 mg/liter when dredging was halted in October. Prior to dredging, chlorophyll-a levels averaged 2.5 ug/liter to 3.0 ug/liter. Immediately after dredging commenced, chlorophyll-a reached a concentration of 27 ug/liter, and then decreased to levels of 12-18 ug/liter. Productivity also increased from pre-dredging levels of about 200 mg C/m^3/d to an average of 750 mg C/m^3/d in 1978 (Peterson, 1981).

Dredging began again in May 1979 and was completed by September. Maximum depth was increased to 6.5 m following dredging. The water quality in 1980 was improved over previous years, and the macrophyte biomass was reduced from 200-300 g dry weight/m^2 to nearly zero.

Steinmetz Lake, New York. Steinmetz Lake is 1.2 ha in area, and has a mean depth of 1.5 m and a maximum depth of 2.1 m. Weed growth, algal growth and highly turbid water were the major concerns.

Restoration included complete drawdown, sediment removal and stormwater drainage diversion. The removed sediment was then replaced with clean quarry sand. This method does not increase lake depth, but produces a new, clean substrate.

Short term results of the restoration project were: increased Secchi disc readings (from 1.25 m to the maximum lake depth), decreased chlorophyll-a levels (from 10.4 ug/liter to 0.1 ug/liter), and reduced aquatic macrophyte biomass (from 30-50 g wet weight/m^2 to virtually zero) (Peterson, 1981). After the treatment, plants grew where tracked vehicles forced organic sediment through the sand cover. The number of people using the lake for recreational purposes increased from almost none to over 3,000.

Lake Herman, South Dakota. Lake Herman has a surface area of 526 ha, a maximum depth of 2.4 m and a mean depth of 1.7 m. The basin has a volume of 8.9×10^6 m^3 (2,642 million gallons). Farming practices in the watershed surrounding the lake have caused high nutrient concentrations and excessive sedimentation. Lake Herman is primarily nitrogen limited and nitrogen frequently declines to zero during algal blooms.

The dredging project was implemented to deepen the lake and remove the nutrients associated with the sediment. Hydraulic dredging removed about 48,000 m^3 of silt from the lake, increasing the mean depth from 1.7 m to about 3.4 m. Dredged material was deposited in an area adjacent to the lake. Shortly after the dredging operation commenced, orthophosphorus concentrations increased from 0.13 mg P/liter to more than 0.56 mg P/liter (Peterson, 1981). Phytoplankton blooms did not accompany the increased phosphorus concentrations because the lake is nitrogen limited. Although no major increase in phytoplankton productivity was observed, the high phosphorus concentrations attributable to phosphorus released to the water column during dredging points out a potentially serious problem that may accompany hydraulic dredging operations.

Nutrient Precipitation and Inactivation

Introduction

Many eutrophic lakes respond slowly following nutrient diversion because of poor flushing rates that facilitate sedimentation, and because of continued internal phosphorus recycling. Phosphorus recycling is controlled by precipitation and inactivation techniques generally used to remove phosphorus from the water column and control its release from bottom sediments. Chemical precipitants used for this purpose include salts of aluminum, iron, and calcium. Calcium (II) has limited use in lakes because it is ineffective below pH 9. Iron salts are not suitable inactivants for long-term phosphorus control, since anoxic conditions reduce iron complexes. This releases phosphorus and iron in the soluble state (Fe III - Fe II). Therefore, aluminum compounds such as aluminum sulfate and sodium aluminate are the most widely used. Zirconium and lanthanum (rare earth elements) have proved effective in phosphorus removal, but more research is needed on direct toxicity and general health effects before this technique receives large-scale use.

Suitable Lake Types. Certain lake types are better suited to nutrient precipitation and inactivation than others. Lakes should have moderate to high retention times (several months or longer), since the treatment will not be effective if there is a rapid flow-through of water. A water-phosphorus budget is useful in assessing the significance of retention time.

Nutrient precipitation and inactivation is generally implemented following nutrient diversion, but this method of lake restoration will not be effective if the diversion is insufficient. Lakes with low alkalinity will exhibit excessive pH shifts unless the lake is buffered or a mixture of alum and sodium aluminate is used as precipitant. Finally, in lakes with large littoral areas, phosphorus that is derived from groundwater, translocated from sediments by macrophytes, or resuspended by some activity that stirs up sediment deposits may cause higher phosphorus concentrations than expected.

Purpose

Phosphorus precipitation and inactivation techniques are used in water bodies with high concentrations of phosphorus in the water column and the sediment. Such a condition is generally indicated by nuisance algal blooms. Immediate results of phosphorus precipitation include decreased turbidity and algal growth. Application of aluminum compounds, primarily aluminum sulfate and sodium aluminate, may also effectively control the release of phosphorus from the sediment.

Environmental Concerns of Nutrient Precipitation

One immediate response of phosphorus precipitation is a reduction in turbidity. The increased light penetration could stimulate increases in rooted plant biomass. Other undesirable side-effects include reduced planktonic microcrustacean species diversity and toxic effects of residual dissolved aluminum (RDA) on aquatic biota. Laboratory research is currently underway to enlarge the aquatic toxicity data base available for the U.S. EPA to develop water quality criteria for aluminum for the protection of aquatic life. Aluminum toxicity is pH dependent and it becomes extremely toxic below pH 5. Cooke and Kennedy (1981) cited the following laboratory studies regarding the possible toxic effects on the biota of phosphorus precipitation using aluminum compounds:

(1) *Daphnia magna* had a 16 percent reproductive impairment at 320 ug Al/l (Biesinger and Christian, 1972);

(2) A few weeks exposure to 5,200 ug Al/l seriously disturbed rainbow trout tested in flow through bioassays (Everhart and Freeman, 1973);

(3) No obvious effect on rainbow trout after long-term exposure to 52 ug Al/l (Kennedy, 1978; Cooke, et al., 1978);

(4) *Daphnia magna* survival was reduced 60 percent in 96-hr tests of concentrations to 80 ug Al/l (Peterson, et al., 1974, 1976); and

(5) No negative effects on fish (Kennedy and Cooke, 1974; Bandow, 1974; Sanville, et al., 1976) or benthic invertebrates (Narf, 1978) after full-scale lake treatments. Cooke and Kennedy (1981) noted that there were no toxic effects on fish as long as the pH remains in an acceptable range and the RDA is less than about 50 ug Al/l.

Implementation of Nutrient Precipitation Projects

The following factors should be considered for phosphorus precipitation/inactivation through chemical application: dose, choice of dry or liquid chemical, depth of application, application procedure, and season (Cooke and Kennedy, 1981).

Dose Determination. Cooke and Kennedy (1980) and Cooke and Kennedy (1981) describe some methods for determining dose. A dose of aluminum that reduces pH to 6.0 is considered "optimal." The residual dissolved aluminum should remain below 50 ug Al/l, the level at which aluminum begins to elicit toxic effects. A simplified method for dose determination is outlined below (Cooke and Kennedy, 1980).

Procedure:

(1) Obtain representative water samples from the lake to be treated. Care should be exercised in selecting sampling stations and depths since significant heterogeneities, both vertical and horizontal, commonly occur in lakes. Samples should be collected as close to the anticipated treatment date as possible.

(2) Determine the total alkalinity and pH of each sample. Total alkalinity, an approximate measure of the buffering capacity of lake water, will dictate the amount of aluminum sulfate (or aluminum) required to achieve pH 6 and thus optimum dose. Additional chemical analyses can be performed, depending on the specific needs of the investigator. For example, phosphorus analyses before and after laboratory treatment would allow estimation of anticipated phosphorus removal effectiveness.

(3) Determine the optimum dose for each sample. Initial estimates of this dose, based on pH and alkalinity, can be obtained from Figure IV-2. More accurate estimates should be made by titrating samples with fresh stock solutions of aluminum sulfate of known aluminum concentration using a standard burette or graduated pipette. The concentration of stock aluminum solutions should be such that pH 6 can be reached with additions of 5 to 10 milliliters per liter of sample. Samples must be mixed (about 2 minutes) using an overhead stirring motor and pH changes monitored continuously using a pH meter. Optimum dose for each sample will be the amount of aluminum, which when added, produces a stable pH of 6.0.

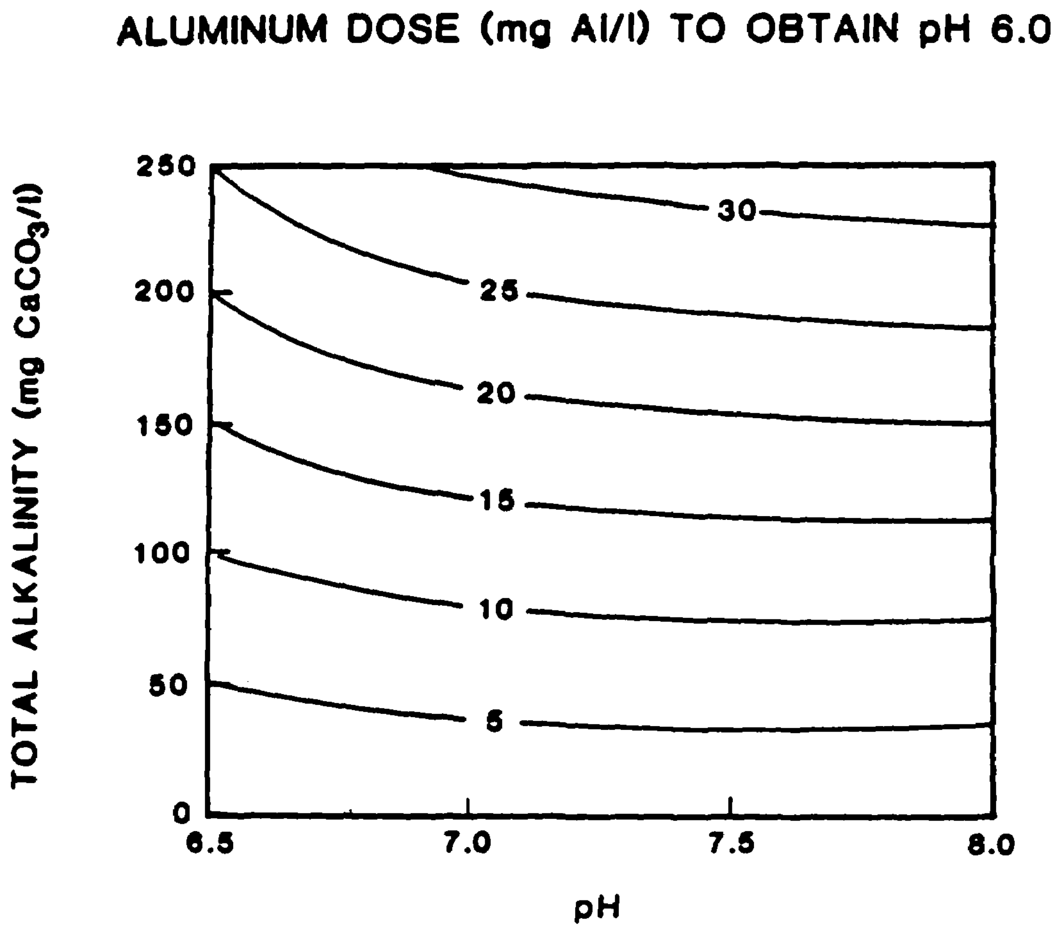


Figure IV-2. Estimated aluminum sulfate dose (mg/1) required to obtain pH 6 in treated water of varying initial alkalinity and pH (from Cooke and Kennedy, 1980).

(4) The relationship between total alkalinity and optimum dose can be determined using information from each of the above titrations by plotting optimum dose as a function of alkalinity. This relationship will allow determination of dose at any alkalinity with the range tested.

Liquid alum and liquid sodium aluminate generally form a better floc and are more effective than the dry forms (Cooke and Kennedy, 1981). If only dry alum is available, it can be mixed in tanks to form a slurry before application.

Depth of Application. Aluminum salts can be applied to surface water, or at predetermined depth(s), depending upon treatment objectives. A surface application is generally needed to remove phosphorus from the water column, whereas hypolimnetic treatment controls the release of phosphorus from sediments.

Time of Application. Both particulate and dissolved forms of phosphorus are efficiently removed by the aluminum floc as it settles to the bottom. Whether there is an optimum season for the application of aluminum salts for the removal of various forms of phosphorus is debatable, as discussed by Cooke and Kennedy (1981).

Nutrient Precipitation Case Studies

Although at least 28 lakes have been reported in the literature that have been treated by the phosphorus inactivation/precipitation technique, there is a paucity of information regarding post-treatment effects. The following sections summarize five case histories that are representative of different approaches, have long-term monitoring, or illustrate strengths and shortcomings of this technique. Information concerning dose, method of application, cost, and long-term effects on additional restoration projects employing inactivation/precipitation techniques is found in Cooke and Kennedy (1981).

Horseshoe Lake, Wisconsin. Horseshoe Lake has a surface area of 8.9 ha, a maximum depth of 16.7 m, and a mean depth of 4.0 m. It is the first reported full scale in-lake inactivation experiment in the United States (Funk and Gibbons, 1979). Prior to treatment, the lake exhibited algal blooms, dissolved oxygen depletions and fish kills. High nutrient levels were attributed to agricultural and natural drainage, and to waste discharges from a cheese-butter factory prior to its closing in 1965.

Alum was applied, just below the water surface, in May 1970. No decrease in phosphorus level was observed until after fall circulation, when concentrations decreased substantially. Reduced phosphorus concentrations were observed in both the epilimnion and the hypolimnion. Although hypolimnetic phosphorus increased slightly every year following treatment, it was controlled for about 8 years. Secchi disc transparency also increased and no fish kills have occurred since the alum application. Additional information about the restoration of Horseshoe Lake is provided by Peterson, et al. (1973).

Medical Lake, Washington. Medical Lake covers an area of 64 ha. It has a maximum depth of 18 m and a mean depth of 10 m. Prior to treatment, the lake exhibited nuisance algal blooms, summer anoxia and high nutrient concentrations, primarily because of internal nutrient cycling. Treatment with alum was chosen as the best method for inactivating phosphorus in Medical Lake.

Alum was applied at the surface or at 4.5 meters, depending upon whether the area was shallow or deep. Application began in August 1977 and continued over a 5-week period.

Water quality monitoring through June 1980 showed that alum treatment successfully reduced phosphorus levels, eliminated algal blooms and increased water clarity. Total and orthophosphorus levels prior to alum treatment were 0.47 mg/liter and 0.32 mg/liter, respectively. These levels decreased about 87 and 97 percent, respectively. Chlorophyll-a decreased from a mean monthly value of 25.2 mg/m^3 prior to alum treatment, to 3.2 mg/m^3 following treatment. Secchi disc transparency improved from a mean depth of 2.4 meters to 4.9 meters. Whereas the lake did not support a fishery prior to treatment, a rainbow trout population flourished after phosphorus precipitation/inactivation. No negative impacts on biota were observed although the concentration of dissolved aluminum increased to 700 ug Al/l during treatment. Post-treatment levels fell to 30-50 ug/l (Cooke and Kennedy, 1981). Detailed results of water quality monitoring following phosphorus precipitation/inactivation treatment are presented in Gasperino, et al. (1980a) and Gasperino, et al. (1980b).

Annabessacook Lake, Maine. Annabessacook Lake, located in central Maine, covers an area of about 575 ha, and has a hypolimnetic area of 130 ha. The mean lake depth is 5.3 m and the maximum depth is 14.9 m. High levels of phosphorus in the water column and sediments were believed to be responsible for blue-green algal blooms. Industrial and municipal wastewater inputs contributed to high phosphorus levels prior to 1972, and internal nutrient cycling caused continued high nutrient levels in the lake (Dominie, 1980).

Annabessacook Lake underwent an extensive lake restoration program, including nutrient diversion, agricultural waste management and in-lake nutrient inactivation. Point sources were diverted from the lake and agricultural waste management plans were implemented. Laboratory testing showed that aluminum treatment was a feasible alternative for lake restoration. Because the lake water has a low alkalinity, a combination of aluminum sulfate and sodium aluminate was used to provide sufficient buffering capacity to moderate potential pH shifts.

After the aluminum application and commencement of waste management programs, the following changes were observed (Dominie, 1980):

- Total phosphorus mass in the lake was reduced from over 2,200 kilograms (kg) in 1977 to 1,030 kg in 1978.

- Internal recyclable phosphorus was reduced 65 percent from 1,800 kg in 1977 to 625 kg in 1979.

- The average June chlorophyll-a concentration decreased from 11.5 ug/l (1977) to 6.2 ug/l (1978).

- Secchi disc depth for June (monthly mean) increased from 2.0 m (1977) to 3.1 m (1978).

Additional information on the restoration of Annabessacook Lake is found in Dominie (1980), Gordon (1980), Cooke and Kennedy (1981), and U.S. EPA (1982).

Liberty Lake, Washington. Liberty Lake, in Spokane County, has a surface area of 316 ha. The lake has a mean depth of 7.0 m, and a maximum depth of 9.1 m. A combination of septic tank drainage, urban runoff, and poor solid waste disposal practices caused excessive nutrient levels and heavy blooms of blue-green algae in the lake.

In 1974, Liberty Lake was treated with aluminum sulfate to precipitate and inactivate phosphorus. Jar tests and in situ tests were made to determine dosage. The alum slurry was applied to the surface. After application of aluminum sulfate, total phosphorus was reduced from 0.026 mg/l to less than 0.015 mg/l. Water clarity increased following the treatment. Although alkalinity and pH dropped, the effect was short lived and these parameters returned to pretreatment levels within 24 to 48 hours (Funk and Gibbons, 1979).

The treatment effectively controlled algal blooms from 1974 to 1977. Heavy blooms equivalent to those prior to treatment did occur in the fall of 1977.

Dollar Lake and West Twin Lake, Ohio. Dollar Lake has a surface area of 2.22 ha, a mean depth of 3.89 m and a maximum depth of 7.5 m. West Twin Lake, which is adjacent to Dollar Lake, is larger, with a surface area of 34.02 ha, a mean depth of 4.34 m and a maximum depth of 7.50 m. Septic tank drainage was largely responsible for eutrophic conditions. Although septic effluent was diverted in 1971-72, algal blooms continued, partly because of internal cycling of phosphorus.

Aluminum sulfate was applied to the hypolimnion of the lakes to inactivate and precipitate phosphorus. Following the alum application, both lakes showed decreased phosphorus content in the water column and improved water transparency. Blue-green algae dominance in West Twin Lake was reduced by 80 percent (Funk and Gibbons, 1979; Cooke and Kennedy, 1981). Zooplankton populations were affected, and the dominant species shifted from Cladocera to Copepoda. Hypolimnetic phosphorus concentration in Dollar and West Twin Lakes remained low for four years after treatment.

Aeration/Circulation

Introduction

Aeration/circulation is a potentially useful technique for treating symptoms of eutrophication. The range of aeration/circulation techniques can be divided into two major groups: artificial circulation and hypolimnetic aeration. Both of these techniques increase the dissolved oxygen concentration of hypolimnetic waters. The two techniques differ in that hypolimnetic aeration aerates hypolimnetic waters without mixing them with surface waters while artificial circulation breaks down stratification by mixing the upper and lower strata of the water column. These techniques can be used to enhance the habitat of aquatic biota and improve water quality by alleviating problems created by stratification and deoxygenation of the hypolimnion.

Both techniques restore oxygen to anaerobic bottom waters. These restoration procedures lead to habitat expansion for zooplankton, benthos and fish. Destratification is usually benefical for warmwater fish, promoting an increase in the depth distribution. However, complete mixing may eliminate coldwater habitats and fish such as salmonids may disappear from the lake.

Lakes Best Suited for Aeration/Circulation. Anaerobic bottom waters of a stratified lake can be oxygenated by aeration/circulation techniques. Either method may be implemented when the primary purpose of treatment is to alleviate "taste and odor" problems resulting from high concentrations of Fe, Mn, H_2S and other chemicals in an anoxic hypolimnion. Both methods expand or improve habitat for zooplankton, benthos, and warmwater fish. However, artificial circulation and hypolimnetic aeration do not produce the same effects in lakes.

Artificial aeration may cause the replacement of blue-green algae communities by more desirable communities of green algae, while hypolimnetic aeration generally does not have an effect on phytoplankton. Since hypolimnetic aeration does not effect mixing of surface and hypolimnetic waters, nutrient concentrations in the euphotic zone are basically unaffected when this technique is employed. Consequently, hypolimnetic aeration generally does not affect the phytoplankton community. In contrast, artificial circulation vertically mixes the water column and can increase nutrient concentrations in the euphotic zone. In a series of experiments, Shapiro (1973) showed that natural populations of blue-green algae were replaced by green algae after enrichment with phosphorus and nitrogen when carbon dioxide was added or pH was lowered. These results indicate that green algae can outcompete blue-green algae under enriched nutrient conditions as long as CO_2 is abundantly available.

When control of algal blooms is not a prime consideration and a coldwater supply is necessary, the preferred method is hypolimnetic aeration. A cold hypolimnion is needed for survival of coldwater fish, and thus hypolimnetic aeration is recommended when improvement of fisheries is the only consideration. In southern lakes, high water temperatures in the epilimnion and metalimnion often preclude survival of coldwater fish; therefore, it is necessary to preserve the integrity of the water layers, including the colder hypolimnion, and artificial destratification would not be appropriate.

Artificial circulation is preferred when limitation of algal biomass is desired, oxygenation of the metalimnion is needed, or a completely mixed water column is acceptable. Artificial circulation is also suitable for northern lakes where the temperature of surface waters does not exceed 22°C during the summer (Pastorak, et al., 1981).

Purpose

Artificial Circulation. Anaerobic conditions in the hypolimnion of a stratified lake restrict the vertical distribution of fish, eliminate certain benthic organisms, and may cause the release of nutrients and toxic substances to the overlying water. Artificial circulation alleviates these problems by destratifying and oxygenating bottom waters of the lake. The water becomes oxygenated primarily through atmospheric exchange at the water surface. Except in very deep lakes, the transfer of oxygen from air bubbles of diffused air systems is relatively small.

By aerating and destratifying lakes, artificial circulation improves water quality, decreases algal growth, and improves fish habitat. These effects are described below.

Elimination of Taste and Odor Problems. Generally, artificial destratification oxygenates anaerobic hypolimnetic waters. Anaerobic conditions near the lake bottom cause the release of reduced chemical species from sediments to the water column. Water supply utilities experience water quality control problems resulting from the accumulation of iron (Fe), manganese (Mn), carbon dioxide (CO_2), hydrogen sulfide (H_2S), ammonium ions (NH_4+) and other chemicals in the hypolimnion. As hypolimnetic waters are brought to the lake surface during artificial circulation, gases such as CO_2, H_2S and NH_3 are released to the atmosphere. Artificial circulation increases hypolimnetic oxygen, and raises the redox potential near the lake bottom. The result is decreased concentrations of reduced chemical species, thereby eliminating taste and odor problems.

Decreased Algal Growth. In some cases, algal production is reduced through artificial circulation. Pastorak, et al. (1981) cited Fast (1975) for several mechanisms that cause reduced algal growth. Internal nutrient loading may be reduced through the elimination of anaerobic conditions that cause nutrient regeneration. Artificial circulation also increases the mixed depth of the algae, thereby reducing algal growth through light limitation. When mixing is induced during an algal bloom, the algae are distributed through a greater water volume, and lake water transparency will increase immediately. In addition, as water is pumped to destratify the lake, rapid changes in hydrostatic pressure and turbulence serve to destroy phytoplankton.

Artificial circulation does not consistently decrease algal populations, and may cause increased algal biomass in some instances. Pastorak, et al. (1981) surveyed the literature covering 40 experiments in which destratification was relatively complete. Only 26 experiments exhibited significant changes in phytoplankton biomass, and of these, about 30 percent exhibited increases in algae.

Forsberg and Shapiro (1981) found that changes in algal species composition during artificial aeration depend primarily on the mixing rate. With slow mixing rates, surface levels of total phosphorus and pH generally increased, and the relative abundance of blue-green species such as *Anabaena circulinus* and *Microcystis aureginosis* increased.

The abundance of green algae and diatoms increased when faster mixing rates were used. Complete chemical destratification caused by high mixing rates was accompanied by large increases in surface total phosphorus and CO_2 concentration. The green algae Sphaerocystis schroederi, Ankistrodesmus falcatus and Scenedesmus spp., and the diatoms Nitzchia spp., Synedra spp., and Melosira spp. grew particularly well under these conditions (Forsberg and Shapiro, 1981).

Benefits to Fish Populations. Artificial circulation may enhance fish habitat and food supply, thereby potentially improving growth of fish, environmental carrying capacity, and overall yield.

Low oxygen levels in the hypolimnion may prevent fish from using the entire potential habitat. Destratification and aeration of bottom waters may allow fish to inhabit a greater portion of the water column, expanding the vertical distribution of warmwater fish.

Salmonids in particular may be restricted to a layer of metalimnetic habitat, with warm water above and anaerobic conditions below. If surface water temperatures remain below 22°C throughout the summer, as in northern lakes, artificial circulation should increase habitat for cold-water fish. In addition, summer-kill of fish due to anoxic conditions and toxic gases may be prevented by artificial circulation.

Artificial circulation has also proved to be an effective method of preventing over-winter mortality of salmonids. Whereas natural oxygen concentrations may be depleted during the winter, aeration prior to ice formation can provide sufficient oxygen for fish survival. Winter mortalities of fish in Corbett Lake, British Columbia, were prevented in this way (Pastorak, et al., 1981).

Hypolimnetic Aeration and Oxygenation. Hypolimnetic aeration and oxygenation add dissolved oxygen to the bottom waters without destratifying the lake. Aeration of the hypolimnion occurs through oxygen transfer between air bubbles and water, and oxygenation occurs more slowly than with artificial circulation.

Major goals of programs employing hypolimnetic aeration and oxygenation are to improve water quality and provide habitat for coldwater fish. Unlike artificial circulation, there is no evidence that hypolimnetic aeration will control algal blooms.

Improvement of Water Quality. Hypolimnetic aeration minimizes taste, odor and corrosion problems by oxygenating bottom waters, which raises the pH and lowers concentrations of reduced compounds. Although artificial circulation aerates the water column more rapidly, hypolimnetic aeration maintains stratification, thereby retaining a coldwater resource.

Improvement of Fisheries. Hypolimnetic aeration creates habitat for coldwater fish by oxygenating the cold bottom layers of a lake. Because the lake does not become completely mixed as a result of hypolimnetic aeration, a two-story fishery can develop. Aeration also enhances fish food supply, since the distribution and abundance of macroinvertebrates increases.

Planktivorous fish may also find an increased food supply following hypolimnetic aeration. While phytoplankton abundance is generally unaffected, zooplankton populations may expand their vertical range after treatment. Fast (1971) found a significant increase in the population of Daphnia pulex following aeration of Hemlock Lake, Michigan. He attributed the population change to an expanded habitat, which allowed Daphnia to inhabit dimly lit depths of the lake and avoid predation by trout.

Environmental Concerns of Aeration/Circulation

Most of the environmental concerns are associated with the use of artificial destratification systems, whereas very few adverse impacts of hypolimnetic aeration are known. Hypolimnetic aeration has very little influence on depth of mixing, pH of the water, sediment resuspension, and algal densities. Adverse impacts of aeration/circulation, including effects on water quality, nuisance algae, macrophytes and fisheries, are described in the following sections. Examples of impacts of aeration/circulation on lakes are presented later in a section on Case Histories. The purpose of the present discussion of environmental concerns is to point out adverse consequences that might occur as a result of artificial destratification. Although these effects will not necessarily be seen, it is instructive to recognize the potential problems that could arise, on a site-specific basis.

Water Quality. Artificial circulation may cause several chemical and physical changes that adversely affect water quality. The mixing of nutrient rich hypolimnetic water could increase the concentrations of nutrients in the upper water layers. Heightened concentrations of the gases NH_3 and H_2S may also occur in surface water.

Turbulence due to mixing and aeration systems may further affect water quality by resuspending silt, thereby increasing turbidity. Decreases in water transparency after mixing may also be associated with surface algal blooms (Pastorak, et al., 1980).

Nuisance Algae. Artificial circulation/destratification may produce undesirable changes in phytoplankton communities. For example, temporary algal blooms may occur because of recycling of hypolimnetic nutrients and elevation of total phosphorus. Such a rise in algal biomass may favor blue-green algae by depleting CO_2 and keeping pH levels high.

Macrophytes. Improved water transparency following artificial circulation may allow increased macrophyte growth. Rooted aquatic plants could expand to nuisance levels, especially in lakes with shallow littoral shelves.

Fisheries. Where coldwater fish exist in the metalimnetic region, artificial circulation and the subsequent warming of bottom waters may eliminate habitat for certain species. The surface temperatures of northern lakes generally remain below 22°C, and thus the bottom waters will not be warmed (as might occur in southern lakes), and habitat for coldwater fish will be enhanced during circulation. Destratification and mixing can also lead to dissolved oxygen decreases in the whole lake. In this instance, resuspension of bottom detritus increases the biochemical oxygen demand (BOD) beyond the rate of reaeration (Pastorak, et al., 1981). Extensive depletion of dissolved oxygen may be responsible for fish mortalities. Aeration of Stewart Lake initially caused a decline in bluegill population, presumably because of reduced dissolved oxygen (Pastorak, et al., 1981).

Fish kills may also be caused by supersaturated concentrations of nitrogen, which may result from circulation or hypolimnetic aeration. In spring, N_2 levels generally equilibrate at 100 percent saturation with respect to surface temperature and pressure. Warming of the hypolimnion during the summer results in supersaturation of N_2 relative to surface temperature and ambient temperature at depth. This supersaturation of N_2 may induce gas bubble disease in fish, causing stress or mortality (Pastorak, et al., 1981). Although this has not been documented in lakes, dissolved nitrogen concentrations of 115-120 percent saturation induced salmonid mortalities in rivers (Rucker, 1972).

Implementation of Aeration/Circulation Projects

Aeration/circulation is a relatively inexpensive and efficient restoration technique. The following sections briefly describe methods and equipment used in restoration projects employing artificial circulation or hypolimnetic aeration.

Artificial Circulation. Lake circulation techniques can be broadly classified in the categories of diffused air systems or mechanical mixing systems (Lorenzen and Fast, 1977). Diffused air systems employ the "air-lift" principle, as water is upwelled by a plume of rising air bubbles. Mechanical systems move water by using diaphragm pumps, fan blades, or water jets. Lorenzen and Fast (1977) reviewed the design and field performance of various circulation techniques, and concluded that diffused air systems are less expensive and easier to operate than mechanical mixing systems.

Diffused Air Systems. Diffused air systems inject compressed air into the lake through a perforated pipe or other simple diffusers. Johnson and Davis (1980) reviewed submerged jetted inlets and perforated pipe air-mixing systems used in reservoirs. Hypolimnetic water is upwelled by the rising air bubbles. Upon reaching the surface, this water flows out horizontally and sinks, mixing with the warm surface water in the process. The amount of water flow induced by the rising bubbles is a function of air release depth and air flow rate. Artificial circulation is generally most effective if air is injected at the maximum depth possible (Pastorak, et al., 1981). In a thermally stratified lake, mixing will normally be induced only above the air release depth. However, while an aerator located near the surface of the lake may be unsuitable for destratifying a lake, it may effectively prevent the onset of stratification (Pastorak, et al., 1981).

Mechanical Mixing. Mechanical mixing devices such as pumps, fans and water jets are employed less frequently than diffused air systems. Pastorak, et al. (1981) notes several instances in which mechanical mixing devices have been successfully employed:

(1) Stewart Hollow Reservoir and Vesuvius Reservoir, Ohio--a pumping rate of 10.9 m^3/min was sufficient to destratify the reservoirs within 8 days (Irwin, et al., 1966);

(2) Ham's Lake, Oklahoma--an axial-flow pump with a capacity of 102 m^3/min completely destratified the lake, which has a mean depth of 2.9 m, after 3 days of operation (Toetz, 1977).

On the other hand, mechanical mixing may not always be successful:

(1) West Lost Lake--a pumping capacity of 1.3 m^3/min over a period of 10.1 days was not sufficient to completely mix the lake (Hooper, et al., 1953);

(2) Arbuckle Lake, Oklahoma--an array of 16 pumps (total capacity 1,600 m^3/min) did not completely mix the lake, which has a mean depth of 9.5 m (Toetz, 1979).

Artificial circulation techniques should be started before full development of thermal stratification, because nutrients that become trapped in the hypolimnion and then are recycled may cause increased algal growth. Lorenzen and Fast (1977) recommend about 9.2 m^3/min of air per 10^6 m^2 of lake surface (= 30 SCFM per 10^6 ft^2) to adequately mix and aerate the water column.

Hypolimnetic Aeration. Fast and Lorenzen (1976) reviewed designs of hypolimnetic aerators, and proposed the following divisions: mechanical agitation systems, pure oxygen injection, and air injection systems (which include full air-lift designs, partial air-lift designs, and downflow air injection systems). Hypolimnetic aeration systems generally remove water from the hypolimnion, aerate and oxygenate it, and then return the water to the hypolimnion.

Mechanical Agitation. Mechanical agitation systems generally draw hypolimnetic water up a tube and aerate it at the surface through mechanical agitation. Fast and Lorenzen (1976) noted that a surface agitator design is most efficient for hypolimnetic aeration of shallow lakes where water depth is insufficient to provide a large driving force for gas dissolution.

Oxygen Injection Systems. As in other hypolimnetic aeration systems, water is removed from and returned to the hypolimnion. In oxygen injection systems, nearly pure oxygen becomes almost completely dissolved when it is returned to the hypolimnion (Fast and Lorenzen, 1976).

Air Injection Systems. The full air lift design is the least costly system to construct, install and operate (Fast and Lorenzen, 1976; Fast, et al., 1976; Pastorak, et al., 1981). In these systems, compressed air is injected near the bottom of the aerator, and the air/water mixture rises. At the water surface, air separates from the mixture and water is returned to the hypolimnion.

Partial air lift designs are less efficient than full air lift designs. Partial air lift systems aerate and circulate hypolimnetic water by an air injection system, but the air/water mixture does not upwell to the surface.

Air and water separate below the lake's surface and air rises to the atmosphere while water returns to the hypolimnion (Fast and Lorenzen, 1976).

Aeration/Circulation Case Studies

Three case studies are presented in this section to summarize the effects of artificial circulation on lakes.

Parvin Lake, Colorado. Parvin Lake is a 19 ha mesotrophic reservoir, with a maximum depth of 10 m and a mean depth of 4.4 m. Summer surface temperatures remain less than 21°C year-round.

The effects of artificial circulation on Parvin Lake were studied for two years (Lackey, 1973). November 1968 to October 1969 was the control period during which phytoplankton were sampled to provide baseline information. The treatment year, when the destratification system operated continuously, extended from November 1969 to October 1970.

Phytoplankton in Parvin Lake were affected in the following ways (Lackey, 1973):

- Abundance of green algae significantly decreased during treatment;
- Anabaena, a nuisance blue-green algae, followed a similar pattern of abundance during both control and treatment years;
- Planktonic diatoms decreased in abundance during the treatment winter.

Ham's Lake, Oklahoma. Pastorak, et al. (1981) summarized the effects of artificial destratification on Ham's Lake, Oklahoma. The lake, which has a maximum depth of 10 m, and a mean depth of 2.9 m, covers an area of 40 ha. Following destratification, the lake showed an increase in Secchi disc depth, dissolved oxygen concentration, and phosphate concentration. Both the density and the diversity of benthic organisms increased. Decreases in concentrations of ammonium, nitrate, iron and manganese in the water column were noted. No changes in algal density, chlorophyll-a, green algae, blue-green algae, or the ratio of green algae/blue-green algae was observed.

Kezar Lake, New Hampshire. Kezar Lake has an area of 73 ha, a maximum depth of 8.4 m, and a mean depth of 2.8 m. Artificial circulation was imposed from July 16 to September 12, 1968, and became completely destratified (Haynes, 1973). The responses of the lake to artificial circulation were:

- Increases in Secchi disc depth, pH, dissolved oxygen concentration, phosphate, and total phosphorus;
- Decreases in ammonium, iron and manganese concentrations;
- Reductions in algal density, algal standing biomass, and blue-green algae;

- Increases in green algae, and the ratio of green algae/blue-green algae; and

- No change in mean chlorophyll-a concentration.

Ottoville Quarry, Ohio. Ottoville Quarry is a small (0.73 ha) water-filled quarry, with a maximum depth of 18 m. Prior to treatment, rainbow trout (*Salmo gairdneri*) were unable to survive the summer because of high water temperature and oxygen depletion. A program employing hypolimnetic oxygenation was implemented in 1973 (from July to September), and increased summer dissolved oxygen concentrations from nearly zero to 8 mg/l (Overholtz, et al., 1977). Aeration from May to October, 1974, caused dissolved oxygen concentrations in the hypolimnion to exceed 20 mg/l by September.

Overholtz, et al. (1977) found that hypolimnetic aeration created an environment suitable for rainbow trout survival while maintaining thermal stratification in the quarry.

Lake Drawdown

Introduction

The primary purpose in restoration programs employing lake drawdown is to control the growth of nuisance aquatic macrophytes. In general, the water level in a lake is lowered sufficiently to expose the nuisance plants while retaining an adequate amount of water in the lake to protect desirable fish populations. This technique is effective for short-term control (1-2 years) of susceptible aquatic macrophytes. Secondary objectives include turbidity control by sediment consolidation, reduction of nutrient release from sediments (through sediment consolidation or removal), management of fish populations and waterfowl habitats, repair of shoreline structures and simultaneous use of other restoration methods such as covering sediment with new clean material (Cooke, 1980a, 1980b). Sediment consolidation may also cause a slight increase in lake depth. The following sections expand upon the technique of lake drawdown, including methods and case studies.

Lake Conditions Most Suitable for Lake Drawdown. Drawdown and sediment consolidation may be feasible for the restoration of shallow lakes if two conditions are met. The lake basin should have a shallow slope, so that a small vertical decline in water level exposes a large part of lake bottom, and the source of water must be controlled (Dooris, et al., 1982).

The nature of the lake sediment is particularly important to the success of drawdown projects. The sediment that will be exposed must be able to dry and consolidate quickly so that a prolonged dewatering period is not required, and the dried and compacted sediment should not rehydrate significantly after the refilling of the lake basin. However, the sediment should be of a consistency which would allow colonization by desirable plants and benthic organisms (Dooris, et al., 1982).

Purpose

The main objective of lake level drawdown is to marage nuisance macrophytes by destroying seeds and vegetative reproductive structures through exposure to drying and/or freezing conditions. In addition, dewatering and consolidation of sediments alters the substrate, thereby eliminating conditions required for the growth of certain aquatic plants. Sediment consolidation also helps control turbidity, reduces nutrient release from sediments and causes a slight deepening of the lake.

Lake drawdown can be used to enhance fisheries and waterfowl habitats. The simultaneous use of other restoration techniques, such as sediment covering or removal, will be even more effective for control of vegetation. The period of dewatering may also be used to repair shoreline structures, such as dams, docks and swimming beaches.

Environmental Concerns of Lake Drawdown

There may be negative impacts of lake drawdown as well as desirable effects. Negative environmental changes that may occur following drawdown include establishment of resistant macrophytes, algal blooms, fish kills, changes in littoral fauna, failure to refill, and decline in attractiveness to waterfowl.

Algal blooms that occur after reflooding may be one of the undesirable effects of drawdown. Geiger (1983) observed increases in total nitrogen, total phosphorus, and chlorophyll-a following drawdown of Blue Lake, Oregon. The cause of such increases is unclear although it is postulated that drawdown and exposure of sediments, and the subsequent aeration and oxidation bring about nutrient release when the basin is reflooded. The released nutrients are then available for algal growth.

Fish kills may be caused by drawdown, especially if the water level is lowered during the summer. The warmer temperatures cause increased rates of metabolism and heighten the sediment oxygen demand. However, Cooke (1980a) noted that a 2 m summer drawdown of Long Lake, Washington (maximum depth 3.5 m) did not cause fish kills, and the dissolved oxygen remained above 5 mg/l.

Drawdown and reflooding may cause changes in the diversity and density of benthic fauna. Increases in invertebrate density, but decreases in species diversity, have been observed following drawdown and reflooding (Cooke, 1980a). Summer drawdown and subsequent hardening of littoral soils may reduce repopulation by insects. These changes may be detrimental to fish and waterfowl.

The basin may not refill because of an insufficient watershed drainage area, unexpected drought and, in the case of reservoirs, failure to close the dam at the proper time. Failure to refill may have a great impact on the aquatic biota, interrupting the life cycles of those species dependent at some time upon littoral areas.

While drawdown brings about short-term control of most rooted species, some species are strongly resistant to exposure and may even be stimulated by it. Those species that are strongly resistant to drawdown and exposure include *Myriophyllum spicatum*, *Ceratophyllum demersum*, *Lemna minor*, *Najas flexilis*, and *Potamogeton pectinatus*. Cooke (1980a) compiled the following list of responses of some common nuisance aquatic macrophytes to drawdown:

o Increased: Alternanthera philoxeroides (alligatorweed)
Najas flexilis (naiad)
Potamogeton spp. (pondweed)

o Decreased: Chara vulgaris (muskgrass)
Eichornia crassipes (water hyacinth)
Nuphar spp. (water lily)

o No clear response or change: Cabomba caroliniana (fanwort)
Elodea canadensis (elodea)
Myriophyllum spp. (milfoil)
Utricularia vulgaris (bladderwort)

Information on the responses of 63 aquatic plants to drawdown is available in Cooke (1980a).

Additional negative effects of drawdown may include lowered levels in potable water wells, and the loss of open water or access to open water for recreation.

Implementation of Drawdown Projects

Lake drawdown should not be considered without first conducting a number of laboratory and other investigations to determine the feasibility of the technique. These investigations should include simulations of lake drawdown, and laboratory studies of nutrient solubilization. Lake drawdown is applicable only to lakes in which water input and output may be controlled. The extent of macrophyte growth is important in specifying the depth to which the lake level will be lowered.

Laboratory Experiments. Drawdown simulations are performed to determine the extent to which sediments will dry and consolidate. Containers that have been used in lake simulations range in size from Plexiglass tubes that are 4.45 cm (ID) and 0.3 m high (Dooris, et al., 1982), to columns 0.3 m (ID) and 1.2 m high (Fox, et al., 1977). Fox, et al. (1977) also used plastic swimming pools (2.4 m in diameter, 45 cm deep) in lake simulation experiments. The containers of sediment are exposed to air and light for a period of time, during which sediment shrinkage and water loss are measured. The drying rate of the sediment can then be determined.

The container of dried sediment should be refilled, and the orthophosphate, total phosphorus and total nitrogen levels measured. Ideally, only small amounts of nitrogen and phosphorus compounds should be released from the consolidated sediment. Large releases of nutrients may presage algal blooms that may occur when the lake basin is refilled following drawdown.

Drawdown. The level of the lake should be lowered sufficiently to expose most of the nuisance macrophytes, but to allow enough water for fish survival (if desired). It may be advantageous to combine drawdown with other restoration techniques such as sediment removal and sediment covering.

Certain species of aquatic macrophytes may be more susceptible to drawdown during one season than another. The decision to employ summer or winter drawdown should be based upon the severity of the climate in a particular area, and upon consideration of lake uses and secondary management objectives. For example, winter drawdown is advantageous because there will be no invasion by terrestrial plants nor development of aquatic emergents, and little interference with lake recreational uses. In addition, water bodies drawn down in winter can usually be refilled in spring. In contrast, refilling in the autumn after a summer drawdown may not be possible.

Complete dewatering of sediment is problematic during the winter, especially in regions of heavy snow or frequent winter rain. Winter drawdown may also defeat other objectives such as the establishment of emergent vegetation for waterfowl habitat, since these species may be susceptible to the cold.

Lake Drawdown Case Studies

Lake level drawdown is a multipurpose improvement technique. The major objective is generally to control the growth of rooted aquatic vegetation, with secondary objectives of fish management, sediment consolidation, and turbidity control. The following case histories exemplify the effects of drawdown on lake biota.

Murphy Flowage, Wisconsin. Murphy Flowage (303 ha) was drawn down for two consecutive winters in an effort to control the macrophyte species *Potamogeton robbinsii* (Robbin's pondweed), *Ceratophyllum demersum* (coontail), *Nuphar* sp. (water lily), *Potamogeton natans* (floating-leaf pondweed), and *Myriophyllum* sp. (water milfoil). In 1967 and 1968, the water level of the Flowage was lowered 1.5 m from November to March, and restored in April. There was an 89 percent reduction in area covered by macrophytes following the first drawdown, and an additional 3 percent reduction occurred following the second drawdown. The species that had been dominant were controlled or nearly eliminated. No fish kills occurred during drawdown. Following the second drawdown, resistant species such as *Megalondonta beckii* (bur marigold), *Najas flexilis* (naiad), and *Potamogeton diversifolius* (pondweed) began to spread. The extent to which resistant species may have spread is unknown, because a flood destroyed the Flowage in 1970 and evaluations were ended (Cooke, 1980a).

Blue Lake, Oregon. Blue Lake is an oxbow lake with a surface area of 26.3 ha, a maximum depth of 7.3 m, and a mean depth of 3.4 m. Prior to drawdown, Eurasian water milfoil, *Myriophyllum spicatum*, dominated the littoral areas of the lake. During the winter of 1981-1982, the lake level was dropped 2.7 m to the base of most of the milfoil beds.

Drawdown reduced the standing crop biomass by 47 percent at depths less than 1.2 m, and by 57 percent at depths from 2.4-3.7 m. The death of shoots by drying and freezing during drawdown served to reduce milfoil biomass. However, drawdown alone did not eliminate the milfoil, and regrowth from surviving rootcrowns was widespread. The herbicide 2,4-D was applied in 1982 to reduce milfoil growth.

Water quality effects that may be seen following reflooding include a decrease in Secchi disc transparency and an increase in total suspended solids, turbidity, chlorophyll-a and total nitrogen and total phosphorus concentrations (Geiger, 1983).

Additional In-Lake Treatment Techniques

Several additional methods of lake restoration are available, but have not been applied as widely as the techniques noted in the previous sections. The techniques that will be discussed in this section include dilution/flushing, techniques to control nuisance aquatic vegetation (chemical applications, harvesting, habitat manipulation and biological controls), and liming of acidified water bodies.

Dilution/Flushing

Dilution/flushing improves lake water quality by reducing the concentration of the limiting nutrient and increasing the water exchange rate in the lake. The result is a reduction in the biomass of planktonic algae because the loss rate exceeds algal growth rate. The technique is implemented by adding low-nutrient water to the lake in order to reduce the concentration of the limiting nutrient and thereby reduce algal growth. In addition, nutrients and algal biomass are washed from the lake because the water exchange rate is increased (Welch, 1979, 1981a, 1981b).

The purpose of dilution, as suggested earlier, is to deter blue-green algal blooms by decreasing total phosphorus and total nitrogen, and by eliminating biomass at a greater rate than the growth rate can supply new cells. The reduction of allelopathic substances excreted by blue-green algae may also contribute to the increased abundance of diatoms and green algae (Welch and Tomasek, 1980).

Use of the dilution/flushing method is most feasible when large quantities of low-nutrient water are available for transport to the lake that is to be restored. This condition was met in the instances of Moses and Green Lakes in Washington State. Case histories of these two lakes are discussed below.

Moses Lake, Washington. Moses Lake has an area of 2,753 ha and a mean depth of 5.6 m. Prior to restoration by dilution/flushing, the lake was eutrophic and experienced blue-green algal blooms because of high nutrient concentrations. Inflowing water (Crab Creek, [P]=92 ug/l) was diluted with low nutrient water from the Columbia River ([P]=30 ug/l) with about a 3:1 dilution of Crab Creek. Following dilution/flushing, Secchi disc depth in the lake increased from 0.5 m to 1.1 m (April-July values). Total phosphorus, which had a mean value of 142 ug/l prior to dilution, was reduced to 53 ug/l. Chlorophyll-a also decreased from 55 ug/l (mean values for April-July) to 9 ug/l (April-July mean).

Green Lake, Washington. Green Lake, which is located in King County, Washington State, has a surface area of 104 ha, a mean depth of 3.8 m, and a maximum depth of 8.8 m. Prior to dilution, Green Lake had a high level of blue-green algal production, and high nutrient levels caused by subsurface seepage (U.S. EPA, 1982).

Dilution began in 1962 with the Seattle city water supply as the source of low nutrient water. The technique applied to Green Lake was one of long-term dilution at a relatively low rate. Post-dilution monitoring did not begin until three years after dilution was begun, and only one pre-dilution

Habitat Manipulation. Dredging may be used to mechanically remove the whole plant from shallow waters, or it may be used to increase the depth to a point below which plants are unable to grow. Dredging may also remove sediment nutrient sources for aquatic plant growth.

Shades, dyes, bottom coverings and drawdown are also included in habitat manipulation techniques to control aquatic weeds. Black plastic sheeting that floats on the water surface has reportedly controlled growth of *Myriophyllum spicatum* (Nichols and Shaw, 1983). Following four weeks of shading, the plants were brown and dead, and there was little or no regrowth during the rest of the summer. Cooke (1980b) reviewed the various methods that are encompassed by the general category of covering bottom sediments. Included within these techniques are sheeting and screening, and smothering with sand or fly ash. Cooke (1980b) concluded:

- Plastic sheeting appears to be effective in retarding macrophyte growth, but there are problems with application methods and in anchoring the material;

- Fiberglass screens hold promise as effective means of controlling macrophytes, but further evaluation is recommended;

- Sand is apparently not effective if enriched sediment is not first removed because the sand particles sink into flocculent sediments; and

- Fly ash was not recommended because of the negative water quality effects (elevated pH, low dissolved oxygen, high concentrations of heavy metals) and subsequent effects on the biota.

The aniline dye nigrosine has been used in attempts to control macrophytes. Although the toxicity of aniline dyes to other organisms is not known, they are very toxic to humans. Other considerations associated with the use of dyes include aesthetics, loss of effect through dilution, loss of dye through plant uptake and loss by sorption to suspended solids and sediment.

Biological Controls. Biological controls include the use of fish, shellfish, insects, and disease. Some fish that have been suggested for control of aquatic weeds are the common carp (*Cyprinus carpio*), roach (*Rutilus rutilus*), rudd (*Scardinus erythopthalmus*), some species of tilapia (*Tilapia zillii*, *T. mossambica*), silver dollar fish (*Metynnis roosevelti*, *Mylossoma argenteum*), white amur (*Ctenopharyngodon idella*) and hybrids of the white amur (Mulligan, 1969; Nichols and Shaw, 1983). It should be noted that the introduction of exotic species is strictly regulated in many states.

Carp are not primarily herbivores, but they serve to decrease plant growth by uprooting plants when searching for benthic organisms or when spawning, and by increasing turbidity in the water. Although carp have been shown to effectively control elodea and curly-leaved pondweed, they cause water quality problems (suspended sediment, turbidity) which can lead to the demise of sportfish populations (Nichols and Shaw, 1983).

Herbivorous fish can be used to control certain species of aquatic weeds. For example, roach and rudd prefer elodea over milfoil. Milfoil is also the least preferred food of *Tilapia* spp. The introduction of grass carp at Red Haw Lake, Iowa, resulted in control of *Elodea*, *Potamogeton*, *Ceratophyllum* and *Najas*. The biomass of aquatic macrophytes in the lake decreased from 2,438 g/m^2 in 1973 to 211 g/m^2 in 1976 (Mitzner, 1978). Since milfoil is not the preferred food of herbivorous fish, there is a possibility that persistent monocultures of *Myriophyllum spicatum* will develop.

Herbivorous snails have been suggested as potential controls for macrophytes. Although native snail species in temperate regions do not eat macrophytes, two South American species (*Marisa cornuarielisi* L. and *Pomacea australialis*) are macrophyte herbivores that may potentially be used to control pest species. The crayfish *Orconectes causeyi*, which consumes both *Elodea canadensis* and *Myriophyllum exalbescens*, has also been suggested as a means of biological control of macrophytes (Nichols and Shaw, 1983).

Several insects have also been investigated as predators on Eurasian water milfoil. Some of the promising species noted are *Parapoynx stratiota*, *P. allionealis*, *Acentria nivea*, *Litodactylus leucogaster* and all aquatic moths. However, most of these insects are not specific to milfoil. Diseases that may cause declines in milfoil populations include "Lake Venice" disease and "Northeast" disease. The causes of these two diseases are not known nor are the long-term consequences of artificial introduction of disease. Thus, the use of pathogens to control milfoil is not recommended (Nichols and Shaw, 1983).

Neutralization of Acidified Lakes

Causes of Acidity and Problem Definition. Acidity of surface waters is largely caused by two nonpoint sources: acid mine drainage and acid precipitation. Acid mine drainage results when mine water comes in contact with sulfur-containing minerals. Acid precipitation is caused by atmospheric sulfur that is released by electric utilities and urban and industrial operations that use sulfur-containing fuel. Oxidation of sulfuric compounds produces sulfuric acid, which dissociates to form H^+ and SO_4^{2-} ions in surface or atmospheric water (Novotny and Chesters, 1981).

Acid mine drainage and acid precipitation cause undesirable "oligotrophication" (a severe loss of productivity caused by the low pH conditions), including loss of natural fish populations. Salmonid fisheries, particularly lake trout, are susceptible to acidification (Goodchild and Hamilton, 1983).

The ability of surface waters to neutralize acidic inputs is largely a function of the chemical composition and solubility of the surrounding soils and underlying rocks. For example, limestones ($CaCO_3$) and dolomites ($CaMg(CO_3)_2$) yield infinite acid neutralizing capacity, whereas hard rocks such as granites (i.e., quartz - SiO_2, feldspar - $KAlSi_3O_8$) and related igneous rocks, crystalline metamorphic rocks (i.e., gneisses and schists) and calcareous sandstone are associated with water that contain very low concentrations of neutralizing compounds (Novotny and Chesters, 1981; Lewis and Olem, 1983). Areas of the United States where lakes are highly sensitive to acidification are in New England, the Adirondack Mountains of New York, the Appalachians, and the Rockies.

Neutralization. Several materials have been considered for use in neutralizing acid lakes. These include lime (CaO, $Ca(OH)_2$), limestone ($CaCO_3$), dolomite, lime slags, basic flyash, soda ash, and phosphorus. Of these, lime and limestone are the most widely employed to neutralize surface waters (Driscoll, et al., 1982). Dolomite, dolomitic hydrated lime, and dolomite quicklime (each exceeding a 35 percent magnesium content) may also be used. However, limestones containing more than 10 percent magnesium carbonate dissolve slowly and are not practical for use in neutralizing surface waters. Agricultural limestone, while not as effective as quicklime or hydrated lime, has several advantages: it is noncaustic, relatively inexpensive, relatively free of harmful contaminants, and does not produce harmful alkaline conditions (Britt and Fraser, 1983).

Application. Techniques for lime application in lakes include using trucks (blowers), boats (blowers, slurries, bags), aircraft, and sediment injection systems. The proper time and place to apply neutralizing agents depends upon two main factors: the time and location of acidic episodic events (e.g., snowmelt, autumnal rains); and relationships between such events and the critical life stages of aquatic biota. For example, in dimictic lakes, mixing and distribution of lime is enhanced when it is applied during the spring overturn. However, spring acidic snowmelt creates two problems. First, neutralization may occur too late to prevent fish embryo and fry mortality that is caused by acidic snowmelt. Second, the colder snowmelt water may be less dense than deeper lake water, and mixing with neutralized water may be inhibited (Britt and Fraser, 1983).

Liming the entire lake area is desirable, but may not be feasible because of time and other resource constraints. Alternatively, application of lime over the deepest part of the lake allows the particles of $CaCO_3$ more time to react within the water column. Another alternative may be to distribute limestone in shallow littoral zones where wave action enhances dissolution (Britt and Fraser, 1983). An alternative liming strategy involves chemically treating watersheds, thereby neutralizing the associated aquatic ecosystem. Methods to estimate lime requirements are found in Boyd (1982) and Driscoll, et al. (1982).

Liming Effects. The biological consequences of liming have been summarized by Hultberg and Andersson (1982) and Britt and Fraser (1983). Case histories of limed lakes show the following changes in lake biota:

o Decreases in acidophilic algae and mosses, with concurrent increases in diversity of planktonic algae;

o Predominance of cladocerans shifts to a predominance of copepods after neutralization;

o Reduction in benthic biomass after liming, but eventual recovery with repopulation of less acid tolerant species;

- o Most fish species respond positively, with enhanced survival due to successful spawning and hatching.

Some chemical changes caused by neutralization may be of concern. Toxicity changes of metals, especially aluminum, may have serious environmental consequences. Aluminum toxicity varies with pH changes; gill damage to fish may be caused when aluminum reacts with hydroxides from pH 4.4 to 5.2, while other studies indicate that aluminum is most toxic to fish from pH 5.2 to 5.4 (Britt and Fraser, 1983). The sediments of a limed lake may become sinks for aluminum and other toxic metals as pH is raised and the metals are removed from the water column. If the lake is allowed to reacidify after several years of treatment, the remobilization of metals may cause serious biological problems.

Watershed Management

The quality of a lake's water is often a direct manifestation of the number and types of pollution sources in the surrounding watershed. Agricultural practices such as tillage, the use of fertilizers, and operations of confined animal feedlots may potentially increase the loss of sediments and nutrients from the land and accelerate the natural process of lake eutrophication. In urban areas, many pollutants are carried to lakes in stormwater runoff, via combined sewers, storm sewers and direct surface runoff.

The effectiveness of in-lake restoration techniques would be short-lived if the cause of eutrophication (high nutrient input) was not corrected. Watershed pollution control techniques are important corrective and often preventive measures. The following sections highlight watershed management techniques that help control nonpoint sources of pollution from agricultural and urban areas.

Agricultural Pollution Control

Control of Sediment Input and Associated Nutrients. One of the most important water pollutants that results from agricultural activities is the sediment input from eroding croplands. Sediment itself is a physical pollutant, and in addition serves as a vehicle to transport nutrients, pesticides, toxic chemicals, organic matter, and inorganic matter to water bodies. Techniques to reduce soil loss from agricultural lands have been discussed in the U.S. Environmental Protection Agency publication entitled Effectiveness of Soil and Water Conservation Practices for Pollution Control (1979b) and in a publication by Stewart, et al. (1975). Several Soil and Water Conservation Practices (SWCP) will be discussed in the following paragraphs.

No-Till Planting. Planting is accomplished by placing seeds in the soil without tillage, using a fluted coulter that leaves the vegetative cover virtually undisturbed. Chemical herbicides are used to control weeds and previously planted crops. No-till planting can reduce soil loss to less than 5 percent as compared to conventional plowing and planting practices (Novotny and Chesters, 1981). However, this method requires a greater use of herbicides, and lower yields may be expected on some soils. Because vegetative cover is left to decompose on the surface, the loss of soluble plant nutrients is greater in runoff from no-till than from conventionally-tilled plots (U.S. EPA, 1982).

In summary, no-till farming reduces runoff and erosion losses. Therefore, losses of strongly adsorbed and solid phase pollutants (total phosphorus and organic nitrogen) are decreased. Losses of weakly adsorbed pesticides and plant nutrients (dissolved phosphorus) may increase; but overall the no-till technique is effective in reducing losses of both phosphorus and nitrogen.

Conservation Tillage. This technique replaces conventional plowing with a form of noninversion tillage that retains some of the plant residue on the surface. A chisel, field cultivator, or disk can be used for tilling. The organic residue cover protects the soil surface from erosion and decreases the volume and velocity of runoff (U.S. EPA, 1979b). Because runoff volume and soil loss are reduced, losses of strongly adsorbed organic phosphorus, organic nitrogen and insecticides are decreased.

Sod-Based Rotations. This system involves the periodic rotation of row crops and a sod crop such as alfalfa, other legumes, or grasses. Plowing the sod improves filtration and reduces erodibility. Increased soil porosity helps decrease surface runoff, and the reduction in runoff can continue for several years of continuous row crops after the sod crop is plowed under (U.S. EPA, 1982).

An additional benefit of sod-based rotations is that crop rotations lessen the need for applications of fertilizers and pesticides by increasing soil organic matter and species diversity. Also, legumes help restore nitrogen to soils through fixation of atmospheric nitrogen.

Cover Crops. Shredded stalks of corn or sorghum can be left on fields during the non-growing season, thereby reducing runoff and soil loss from normally fallow fields. More protection from surface runoff is provided from the cover crop that is left in place than by late-seeded small-grain winter cover on plowed fields (Novotny and Chesters, 1981).

Terraces. Terraces divide the field into segments with lesser or near-horizontal slopes, thereby reducing the slope effect on erosion rates. Generally, terraces consist of an embankment or a combination of an embankment and a channel that diverts or stores surface runoff.

Terraces are more effective in reducing erosion than in decreasing surface runoff. Consequently, terraces are most effective in reducing strongly adsorbed substances such as total phosphorus and paraquat (Smith, et al., 1979). Impoundment terraces, which retain runoff in surface storage areas, reduce both runoff volume and sediment loss, but the eventual percolation of the stored water may increase the nitrogen loading to the groundwater.

Other Methods to Prevent Sediment and Nutrient Losses. Contouring, ridge planting, contour listing, and strip cropping are methods that are designed to create barriers perpendicular to the natural direction of flow. Runoff volume and water velocity are thus decreased. In the technique of contour plowing, crop rows and plowing follow the natural contour of the land. This practice provides excellent erosion control for moderate rainstorms

(Novotny and Chesters, 1981). Ridge planting involves planting crops on preformed ridges that follow the natural contours of the field. Crop residues are pushed into the furrows between rows, further deterring runoff and erosion (U.S. EPA, 1982).

A special plow (lister) is required to form alternating ridges and furrows for contour listing. Row crops are then planted either in the bottom furrows or the ridge tops. Contour strip cropping is accomplished by alternating the cultivated crops with strips of grass or close growing crops.

The principal erosion control practices for use on croplands are summarized in Table IV-4.

Waste Management Planning. The planning of a waste management system helps prevent the owner from investing in unnecessary components. Evaluations include estimations of liquid and solid waste sources on a farm and development of a complete system to manage them without degrading air, soil or water resources. An operation plan, which provides specific details for operation of the system, should include:

1. Timing, rates, volumes, and locations for applications of waste and, if appropriate, approximate number of trips for hauling equipment and an estimate of the time required.

2. Minimum and maximum operation levels for storage and treatment practices and other operations specific to the practice, such as estimated frequency of solids removal.

3. Safety warnings, particularly where there is danger of drowning or exposure to poisonous or explosive gases.

4. Maintenance requirements for each of the practices.

Waste Storage Ponds. The purpose of waste storage ponds is to temporarily store liquid and solid wastes, wastewater, and polluted runoff until it can be applied to land without polluting surface or ground water. Common uses of waste storage ponds are storage of milkhouse wastes and manure and storage of polluted runoff from feedlots and barnyards.

Diversions or dikes are usually combined with systems employing waste storage ponds. Clear water diversion systems direct water from upland watersheds away from feedlots or barnyards. Polluted runoff may be collected and directed to storage ponds by constructing a system of curbs, gutters or terraces. Design of waste storage ponds should consider the maximum period of time between emptying, which varies according to precipitation, runoff, and waste volume.

Waste Storage Structures. Waste storage structures such as storage tanks and manure stacking facilities serve the same purposes as waste storage ponds, and while storage structures are more expensive they offer several advantages. Advantages include preservation of nutrient content of stored wastes, minimization of odors, management flexibility and improved aesthetics.

TABLE IV-4
PRINCIPAL TYPES OF CROPLAND EROSION CONTROL PRACTICES AND THEIR HIGHLIGHTS (Continued)

E9	Contouring	Can reduce average soil loss by 50% on moderate slopes, but less on steep slopes; loses effectiveness if rows break over; must be supported by terraces on long slopes; soil, climatic, and topographic limitations; not compatible with use of large farming equipment on many topographies. Does not affect fertilizer and pesticide rates.
E10	Graded rows	Similar to contouring but less susceptible to row breakovers.
E11	Contour strip cropping	Rowcrop and hay in alternate 50- to 100-ft strips reduce soil loss to about 50% of that with the same rotation contoured only; fall seeded grain in lieu of meadow about half as effective; alternating corn and spring grain not effective; area must be suitable for across-slope farming and establishment of rotation meadows; favorable and unfavorable features similar to E3 and E9.
E12	Terraces	Support contouring and agronomic practices by reducing effective slope length and runoff concentration; reduce erosion and conserve soil moisture; facilitate more intensive cropping; conventional gradient terraces often incompatible with use of large equipment, but new designs have alleviated this problem; substantial initial cost and some maintenance costs.
E13	Grassed outlets	Facilitate drainage of graded rows and terrace channels with minimal erosion; involve establishment and maintenance costs and may interfere with use of large implements.
E14	Ridge planting	Earlier warming and drying of row zone; reduces erosion by concentrating runoff flow in mulch-covered furrows; most effective when rows are across slope.
E15	Contour listing	Minimizes row breakover; can reduce annual soil loss by 50%; loses effectiveness with postemergence corn cultivation; disadvantages same as E9.
E16	Change in land use	Sometimes the only solution. Well managed permanent grass or woodland effective where other control practices are inadequate, lost acreage can be compensated for by more intensive use of less erodible land.
E17	Other practices	Contour furrows, diversions, subsurface drainage; land forming, closer row spacing, etc.

SOURCE: Stewart, et al., 1975

TABLE IV-4
PRINCIPAL TYPES OF CROPLAND EROSION CONTROL PRACTICES AND THEIR HIGHLIGHTS

	Erosion Control Practice	Benefits and Impact
E1	No-till plant in prior-crop residues	Most effective in dormant grass or small grain; highly effective in crop residues; minimizes spring sediment surges and provides year-round control; reduces man, machine, and fuel requirements; delays soil warming and drying; requires more pesticides and nitrogen; limits fertilizer- and pesticide-placement options; some climatic and soil restrictions.
E2	Conservation tillage	Includes a variety of no-plow systems that retain some of the residues on the surface; more widely adaptable but somewhat less effective than E1; advantages and disadvantages generally same as E1 but to lesser degree.
E3	Sod-based rotations	Good meadows lose virtually no soil and reduce erosion from succeeding crops; total soil loss greatly reduced but losses unequally distributed over rotation cycle; aid in control of some diseases and pests; more fertilizer-placement options; less realized income from hay years; greater potential transport of water-soluble P; some climatic restrictions.
E4	Meadowless rotations	Aid in disease and pest control; may provide more continuous soil protection than one-crop systems; much less effective than E3.
E5	Winter cover crops	Reduce winter erosion where corn stover has been removed and after low-residue crops; provide good base for slot-planting next crop; usually no advantage over heavy cover of chopped stalks or straw; may reduce leaching of nitrate; water use by winter cover may reduce yield of cash crop.
E6	Improved soil fertility	Can substantially reduce erosion hazards as well as increase crop yields.
E7	Timing of field operations	Fall plowing facilitates more timely planting in wet springs, but it greatly increases winter and early spring erosion hazards; optimum timing of spring operations can reduce erosion and increase yields.
E8	Plow-plant systems	Rough, cloddy surface increases infiltration and reduces erosion; much less effective than E1 and E2 when long rain periods occur; seedling stands may be poor when moisture conditions are less than optimum. Mulch effect is lost by plowing.

Waste Treatment Lagoons. Treatment lagoons may be designed as anaerobic, aerobic, or aerated lagoons. They are used principally to treat liquid wastes.

Anaerobic lagoons are the most commonly used. They require less area than aerobic lagoons, and do not need require electricity for operation, as do aerated systems. Treated wastes may be lower in nitrogen due to ammonia volatilization; therefore, the waste may be applied over a smaller land area.

Aerobic lagoons are used for weak agricultural wastes, such as those originating from milk centers. They require large surface areas, and the effluent is rarely suitable for discharge to surface water.

Filter Strips. In this method, runoff from feedlots and barnyards flows over grassy strips. The strips help reduce the volume and pollution content by soil percolation, the filtration capability of the grass, and volatilization.

Waste Utilization. Waste utilization refers to where and when manure should be applied to land. Its purpose is to use the wastes as fertilizer for crops, forage and fiber production, to prevent erosion, to improve or maintain soil structure, to produce energy, and to safeguard water resources.

Factors to be considered include the land areas available, and the crops that will be grown. Other factors that should be considered are the timing of application, nutrient release rates, soil types, and climate.

Urban Runoff Pollution Control

Lakes in urban areas are subject to pollution from stormwater runoff which enters lakes via combined sewers, storm sewers, and direct surface runoff. The runoff contains high concentrations of sediment, nutrients, heavy metals and toxic chemicals.

During storm events, the capacity of combined sewer lines may be exceeded, and overflow structures at sewage treatment plants or in the sewerage system are designed to discharge the excess into surface water bodies. The "first flush effect" refers to the phenomenon in combined sewer overflow samples whereby the highest concentrations of BOD_5, suspended solids, grease and other pollutants are found during the earliest part of a storm event. Accumulated solid deposits that contain organic matter undergoing decay in combined, sanitary and storm sewers may increase BOD_5 concentrations to levels greater than those of normal untreated dry-weather wastewater (Lager and Smith, 1974). Long periods between rainfall, low sewer slopes, infrequent cleaning, and failure to block off or clean catch basins magnify pollutant concentrations in combined sewer overflows, and (to a lesser extent) storm sewer discharges.

Several management alternatives are available to alleviate problems caused by urban stormwater. Techniques may be grouped into three categories: land management, collection system modifications, and storage. While detailed descriptions of urban runoff control measures are beyond the scope of this manual, several components of each category will be briefly summarized in the following paragraphs.

Land Management. Land management practices include those measures designed to reduce urban and construction site stormwater runoff at the source, by employing Best Management Practices (BMPs). On-site measures can be further divided into low structural or non-structural controls.

Low structural control measures require physical modifications in a construction or urbanizing area. The most common on-site control is storage. Storage attenuates peak runoff flows, treats runoff (detention/sedimentation), or contains the flow in combination with another treatment process such as retention/percolation (Lynard, et al., 1980).

Non-structural control measures include surface sanitation, chemical use control, use of natural drainage, and certain erosion/sedimentation control practices (Field, et al., 1977). Surface sanitation (street sweeping operations) may have a significant impact on the quantity of pollutants washed off by stormwater. Certain street cleaning techniques are able to remove 93 percent of the dry weight solids, which make up a significant portion of the overall pollution potential (Field, et al., 1977; Lager and Smith, 1974). A frequently overlooked measure for reducing the pollution potential from urban areas is reduction in the use of fertilizers, pesticides and deicing materials. Suggestions for methods to reduce such inputs can be found in Lager and Smith (1974) and Field, et al. (1977).

Construction in urbanized areas replaces areas of natural infiltration and drainage with impervious areas. The result is increased runoff and flowrates, and decreased infiltration to the groundwater. Use of natural drainage helps reduce drainage costs and pollution, while it enhances groundwater supplies and flood protection (Field, et al., 1977).

Non-structural erosion/sedimentation controls include cropping (seeding and sodding), use of mulch blankets, nettings, chemical soil stabilizers and earthen berms. These measures are described in Lager and Smith (1974), Field, et al. (1977), and Lynard, et al. (1980).

Collection System Controls. Collection system controls include sewer separation, inflow control, flushing and polymer injections, regulators, and remote flow monitoring and control. Several of these alternatives are briefly described below.

Sewer Separation. Sewer separation refers to the conversion of a combined sewer system into separate sanitary and storm sewer systems. The practice of sewer separation has been used for many years, but Lager and Smith (1974) note two main reasons for reevaluating sewer separation. The first reason stems from changes in physical conditions and quality standards from the past, which include: (1) increases in urban impervious areas and municipal water usage, causing overflows of increased duration and quantity; (2) rapid industrial expansion, causing increased quantities of industrial wastewaters in the overflows; (3) increasing environmental concern for better water quality; and (4) the realization that the total amount of available fresh water is limited and that complete reclamation of substantial portions of the flow may be necessary in the future. The second reason includes: (1) separated storm sewer discharges contain pollutants that affect the receiving water and create new problems; and (2) storm sewer discharges occur more frequently and last longer than combined sewer overflows because combined sewer regulators prevent overflows during minor events.

Lager and Smith (1974) concluded that in many cases the separation of existing combined sewer systems is not practically or economically feasible to resolve combined sewer problems. A feasibility study including the cost of alternative methods would indicate the practicality of each option.

Infiltration/Inflow Control. Problems result from infiltration into sewers from groundwater sources, and high inflow rates through direct connections from sources other than those which the sewers are intended to serve. Examples of infiltration are the volumes of water that enter the sewer system through manhole walls, cracks, defective joints, and illegal connections.

Remote Flow Monitoring and Control. Computerized collection system control can be applied to upgrade combined sewer systems. Control systems are intended to assist in routing and storing combined sewer flows to effectively use interceptor and line capacities (Lager and Smith, 1974). The control system is able to sense and report minute-to-minute system status, including flow levels, quantities, treatment rates, pumping rates, gate (regulator) positions, and characteristics at significant locations in the system. Such observations may assist in determining where necessary overflows can be discharged with the least impact. The control system also provides a means for manipulating the system to maximum advantage.

Storage. Storage of runoff effectively prevents or reduces stormwater runoff from entry into combined sewers and surface water bodies. Storage facilities can provide complete or short-term retention of stormwater flows. Retention facilities may incorporate infiltration systems such as gravel bottoms or tile drains.

Detention basins are capable of reducing peak flow volumes from storms, and providing a sediment trap for suspended solids. The gradual release of stormwater lessens impacts caused by flooding, erosion, and disruption of aquatic habitats (U.S. EPA, 1982).

Stormwater flows to treatment plants, and subsequent overflows, may be controlled by in-line or off-line storage facilities. Storage facilities have several advantages: they are basically simple in design and operation, they respond without difficulty to intermittent and random storm behavior, they are relatively unaffected by flow and quality changes, and they are capable of providing flow equalization (Lager and Smith, 1974). Drawbacks of storage basins include their large size (real estate requirements and therefore cost), visual impact and the need to provide for solids dewatering and disposal.

Storage facilities may be in-line, in which regulators and pumping stations are used to store stormwater runoff in areas of the sewer system with extra capacity, or off-line, which may be concrete vaults, or storage basins such as described earlier. Detailed information concerning storage facilities is available in Lager and Smith (1974), Field (1977), and Lynard, et al. (1980).

CHAPTER V

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APPENDIX A

PALMER'S LISTS OF POLLUTION TOLERANT ALGAE

Source: Palmer, 1969

APPENDIX A

PALMER'S LISTS OF POLLUTION TOLERANT ALGAE

TABLE A-1

POLLUTION-TOLERANT GENERA OF ALGAE
LIST OF THE 60 MOST TOLERANT GENERA, IN ORDER OF DECREASING EMPHASIS BY 165 AUTHORITIES

No.	Genus	Group[a]	No. authors	Total Points
1	Euglena	F	97	172
2	Oscillatoria	B	93	161
3	Chlamydomonas	F	68	115
4	Scenedesmus	G	70	112
5	Chlorella	G	60	103
6	Nitzchia	D	58	98
7	Navicula	D	61	92
8	Stigeoclonium	G	50	69
9	Snynedra	D	44	58
10	Ankistrodesmus	G	36	57
11	Phacus	F	39	57
12	Phormidium	B	37	52
13	Melosira	D	37	51
14	Gomphonema	D	35	48
15	Cyclotella	D	35	47
16	Closterium	G	34	45
17	Micractinium	G	27	44
18	Pandorina	F	32	42
19	Anacystis	B	28	39
20	Lepocinclis	F	25	38
21	Spirogyra	G	26	37
22	Anabaena	B	27	36
23	Cryptomonas	F	27	36
24	Pediastrum	G	28	35
25	Arthrospira	B	18	34
26	Trachelomonas	F	26	34
27	Carteria	F	21	33
28	Chlorogonium	F	23	33
29	Fragilaria	D	24	33
30	Ulothrix	G	25	33
31	Surirella	D	27	33
32	Stephanodiscus	D	22	32
33	Eudorina	F	23	30
34	Lyngbya	B	17	28
35	Oocystis	G	20	28
36	Agmenellum	B	19	27
37	Spirulina	B	17	25
38	Pyrobotrys	F	16	24

TABLE A-1 (CONTINUED)

No.	Genus	Group[a]	No. authors	Total Points
39	Cymbella	D	19	24
40	Actinastrum	G	20	24
41	Coelastrum	G	21	24
42	Cladophora	G	22	24
43	Hantzschia	D	18	23
44	Diatoma	D	19	22
45	Spondylomorum	F	16	21
46	Golenkinia	G	14	19
47	Achnanthes	D	16	19
48	Synura	F	14	18
49	Pinnularia	D	15	18
50	Chlorococcum	G	13	17
51	Asterionella	D	14	17
52	Cocconeis	D	14	17
53	Cosmarium	G	14	17
54	Gonium	F	15	17
55	Tribonema	G	10	16
56	Stauroneis	D	14	16
57	Selenastrum	G	13	15
58	Dictyosphaerium	G	11	14
59	Cymatopleura	D	13	14
60	Crucigenia	G	13	14

[a]Groups: B, blue-green; D, diatom; F, flagellate; G, green.

SOURCE: Palmer, 1969.

TABLE A-2

POLLUTION-TOLERANT GENERA OF ALGAE
LIST OF THE 80 MOST TOLERANT SPECIES,
IN ORDER OF DECREASING EMPHASIS BY 165 AUTHORITIES

No.	Genus	Group[a]	No. authors	Total Points
1	Euglena viridis	F	50	93
2	Nitzschia palea	D	45	69
3	Oscillatoria limosa	B	29	42
4	Scenedesmus quadricauda	G	26	41
5	Oscillatoria tenuis	B	26	40
6	Stigeoclonium tenue	G	25	34
7	Synedra ulna	D	25	33
8	Ankistrodesmus falcatus	G	21	32
9	Pandorina morum	F	23	30
10	Oscillatoria chlorina	B	17	29
11	Chlorella vulgaris	G	19	29
12	Arthrospira jenneri	B	15	28
13	Melosira varians	D	22	28
14	Cyclotella meneghiniana	D	20	27
15	Euglena gracilis	F	18	26
16	Nitzschia acicularis	D	18	26
17	Navicula cryptocephala	D	19	25
18	Oscillatoria princeps	B	16	24
19	Oscillatoria putrida	B	13	23
20	Gomphonema parvulum	D	14	23
21	Hantzschia amphioxys	D	18	23
22	Oscillatoria chalybea	B	14	22
23	Stephanodiscus hantzschii	D	16	22
24	Euglena oxyuris	F	15	21
25	Closterium acerosum	G	16	21
26	Scenedesmus obliquus	G	16	21
27	Chlorella pyrenoidosa	G	11	20
28	Cryptomonas erosa	F	15	20
29	Eudorina elegans	F	16	20
30	Euglena acus	F	16	20
31	Surirella ovata	D	16	20
32	Lepocinclis ovum	F	14	19
33	Oscillatoria formosa	B	14	19
34	Oscillatoria splendida	B	14	19
35	Phacus pyrum	F	11	18
36	Micractinium pusillum	G	12	18
37	Agmenellum quadriduplicatum	B	13	18
38	Melosira granulata	D	14	18
39	Pediastrum boryanum	G	15	18
40	Diatoma vulgare	D	17	18
41	Lepocinclis texta	F	12	17
42	Euglena deses	F	13	17

TABLE A-2 (CONTINUED)

No.	Genus	Group[a]	No. authors	Total Points
43	Spondylomorum quaternarium	F	13	17
44	Phormidium uncinatum	B	15	17
45	Chlamydomonas reinhardii	F	10	16
46	Chlorogonium euchlorum	F	10	16
47	Euglena polymorpha	F	11	16
48	Phacus pleuronectes	F	11	16
49	Navicula viridula	D	13	16
50	Phormidium autumnale	B	13	16
51	Oscillatoria lauterbornii	B	8	15
52	Anabaena constricta	B	9	15
53	Euglena pisciformis	F	11	15
54	Actinastrum hantzschii	G	13	15
55	Synedra acus	D	9	14
56	Chlorogonium elongatum	F	10	14
57	Synura uvella	F	11	14
58	Cocconeis placentula	D	12	14
59	Nitzschia sigmoidea	D	12	14
60	Coelastrum microporum	G	13	14
61	Achnanthes minutissima	D	10	13
62	Cymatopleura solea	D	12	13
63	Scenedesmus dimorphus	G	8	12
64	Fragilaria crotonensis	D	9	12
65	Anacystis cyanea	B	10	12
66	Navicula cuspidata	D	10	12
67	Scenedesmus acuminatus	G	10	12
68	Euglena intermedia	F	11	12
69	Pediastrum duplex	G	11	12
70	Closterium leibleinii	G	8	11
71	Oscillatoria brevis	B	8	11
72	Trachelomonas volvocina	F	8	11
73	Dictyosphaerium pulchellum	G	9	11
74	Fragilaria capucina	D	9	11
75	Cladophora glomerata	G	10	11
76	Cryptomonas ovata	F	10	11
77	Gonium pectorale	F	10	11
78	Euglena proxima	F	7	10
79	Pyrobotrys gracilis	F	7	10
80	Tetraedron muticum	G	7	10

[a]Groups: B, blue-green; D, diatom; F, flagellate; G, green.

SOURCE: Palmer, 1969.

APPENDIX B

U.S. ENVIRONMENTAL PROTECTION AGENCY'S PHYTOPLANKTON TROPIC INDICES

Source: U.S. EPA, 1979a

All genus-trophic-values used in formulating the phytoplankton trophic indices are presented in Table B-1. The genus-trophic-values, total phosphorus (TOTALP), chlorophyll-a (CHLA), and total Kjeldahl nitrogen (KJEL) in Table B-1 are simply mean photic zone values associated with the dominant occurrences of each genus. TOTALP/CONC, CHLA/CONC, and KJEL/CONC were calculated by dividing the TOTALP, CHLA, and KJEL values by the corresponding mean cell count. Also given in Table B-1 is a genus-trophic-multivariate-value (MV) calculated for each genus using the following formula:

MV = Log TOTALP + Log CHLA + Log KJEL - Log SECCHI

TABLE B-1
TROPHIC VALUES OF SELECTED GENERA BASED UPON MEAN PARAMETER VALUES ASSOCIATED WITH THEIR OCCURRENCES AS DOMINANTS.

GENUS	DOMINANT OCCURRENCES	TOTALP	CHLA	KJEL	TOTALP / CONC	CHLA / CONC	KJEL / CONC	MV
Achnanthes	6	29	11.5	734	.027	.001	.689	3.53
Actinastrum	2	56	3.5	594	.142	.009	1.508	3.62
Anabaena	33	183	19.7	1015	.098	.011	.545	4.82
Anabaenopsis	7	70	32.9	1393	.008	.004	.165	5.01
Ankistrodesmus	9	75	17.9	573	.082	.020	.626	4.25
Anomoeoneis	3	10	5.4	364	.005	.002	.166	2.32
Aphanizomenon	41	147	37.6	1437	.058	.015	.569	5.18
Aphanocapsa	4	242	21.1	1427	.034	.003	.200	5.04
Aphanothece	3	65	32.4	1493	.009	.004	.203	4.98
Arthrospira	2	51	21.0	1227	.022	.009	.519	4.37
Asterionella	36	36	9.6	491	.023	.006	.310	3.87
Attheya	1	70	1.4	473	1.892	.038	12.784	3.23
Binuclearia	1	42	6.7	425	.038	.006	.384	3.37
Botryococcus	2	56	10.3	1049	.013	.002	.250	4.20
Carteria	2	509	44.5	1513	.176	.015	.523	6.04
Ceratium	2	140	5.2	1046	3.784	.141	28.270	3.84
Chlamydomonas	4	847	55.1	3143	.162	.011	.601	6.75
Chlorella	3	70	53.1	991	.015	.012	.215	5.13
Chromulina	1	8	10.0	348	.008	.010	.336	2.46
Chroococcus	19	163	46.6	1630	.028	.008	.283	5.37
Chroomonas	1	116	32.9	1421	.084	.024	1.032	5.50
Chrysocapsa	1	10	7.9	261	.015	.012	.380	2.16
Chrysococcus	2	1580	75.0	4631	.197	.009	.576	7.32
Closterium	4	20	19.8	698	.007	.007	.249	3.60
Coelastrum	6	60	13.4	1208	.077	.017	1.549	4.36
Coelosphaerium	6	44	11.7	888	.097	.026	1.965	3.82
Coscinodiscus	3	138	62.7	1267	.053	.024	.488	5.25
Cosmarium	3	14	9.9	586	.003	.002	.115	3.27
Crucigenia	2	361	11.8	1048	.696	.023	2.019	4.67

Continued

TABLE B-1

TROPHIC VALUES OF SELECTED GENERA BASED UPON MEAN PARAMETER VALUES ASSOCIATED WITH THEIR OCCURRENCES AS DOMINANTS (Continued)

GENUS	DOMINANT OCCURRENCES	TOTALP	CHLA	KJEL	TOTALP / CONC	CHLA / CONC	KJEL / CONC	MV
Cryptomonas	72	115	16.5	798	.102	.015	.711	4.53
Cyclotella	83	185	29.9	1053	.073	.012	.418	4.10
Dactylococcopsis	58	178	25.0	1041	.026	.004	.153	5.05
Dictyosphaerium	1	18	10.8	949	.050	.030	2.658	3.45
Dinobryon	31	27	8.1	594	.043	.013	.938	3.16
Euglena	8	318	24.5	1481	.190	.015	.884	5.70
Eunotia	1	178	8.6	1199	3.296	.159	22.204	4.88
Fragilaria	45	64	17.5	843	.019	.005	.247	4.13
Glenodinium	4	8	6.4	403	.020	.016	1.025	2.34
Gloeocystis	6	35	10.9	639	.057	.018	1.034	3.50
Gloeothece	2	9	4.0	412	.069	.031	3.169	2.23
Golenkinia	2	615	26.9	1040	.195	.009	.330	5.60
Gomphonema	1	10	7.4	782	.019	.014	1.507	--
Gomphosphaeria	4	25	8.3	1270	.123	.041	6.225	3.65
Gymnodinium	2	9	2.8	256	.053	.016	1.506	1.68
Kirchneriella	8	139	7.6	755	.123	.007	.669	4.15
Lyngbya	99	99	29.5	1498	.008	.002	.115	4.98
Mallomonas	6	87	6.0	642	.798	.055	5.890	3.62
Melosira	255	94	18.1	774	.034	.006	.277	4.49
Merismopedia	22	183	33.6	1387	.059	.011	.444	5.34
Mesostigma	1	57	12.8	571	.131	.029	1.310	4.04
Micractinium	1	101	52.8	1098	.041	.021	.446	5.22
Microcystis	53	148	37.5	1457	.056	.014	.547	5.27
Mougeotia	2	76	29.2	990	.058	.022	.757	5.09
Navicula	6	74	8.2	490	.127	.014	.838	3.93
Nitzschia	29	92	26.5	883	.042	.012	.402	4.78
Oocystis	5	38	14.0	1098	.005	.002	.157	3.97
Oscillatoria	105	125	39.2	1356	.014	.004	.150	5.27
Peridinium	6	16	8.4	595	.054	.029	2.024	3.01
Phacus	2	2523	22.8	4049	3.955	.036	6.346	7.59

Continued

TABLE

TROPHIC VALUES OF SELECTED GENERA BASED UPON MEAN PARAMETER VALUES ASSOCIATED WITH THEIR OCCURRENCES AS DOMINANTS (Continued)

GENUS	DOMINANT OCCURRENCES	TOTALP	CHLA	KJEL	TOTALP/CONC	CHLA/CONC	KJEL/CONC	MV
Phormidium	3	172	113.2	1955	.102	.067	1.164	5.77
Pinnularia	1	4	0.5	264	.400	.050	26.400	0.78
Raphidiopsis	45	106	30.5	1073	.010	.003	.097	4.88
Rhizosolenia	1	31	15.9	1161	.014	.007	.519	4.19
Roya	1	7	2.4	332	.030	.010	1.437	1.68
Scenedesmus	50	351	60.4	1826	.058	.010	.303	6.01
Schroederia	2	17	4.1	552	.063	.015	2.060	2.54
Selenastrum	1	99	9.3	465	.116	.011	.546	4.13
Spermatozoopsis	2	65	8.8	1631	.085	.012	2.132	4.13
Sphaerellopsis	1	57	6.4	532	.594	.067	5.542	3.56
Sphaerocystis	2	46	11.3	1274	.032	.008	.897	4.23
Sphaerozosma	1	13	16.6	750	.002	.003	.128	3.61
Spondylosium	1	21	6.4	599	.058	.018	1.659	--
Staurastrum	1	13	16.6	750	.004	.006	.251	3.61
Stauroneis	1	79	1.9	557	9.875	.238	69.625	3.62
Stephanodiscus	73	166	37.0	1112	.045	.010	.304	5.27
Synedra	48	82	19.0	797	.027	.006	.261	4.42
Synura	1	131	8.9	1449	1.056	.072	11.685	5.11
Tabellaria	20	22	7.7	455	.015	.005	.307	2.86
Tetraëdron	5	18	5.2	384	.040	.012	.859	2.66
Tetrastrum	1	28	6.9	625	.043	.011	.963	3.53
Trachelomonas	4	97	6.0	867	.292	.018	2.611	4.38
GENERAL CATEGORIES								
centric diatoms	32	142	24.9	1000	.033	.006	.234	4.97
pennate diatoms	17	254	46.8	1615	.036	.007	.227	5.81
flagellate	108	154	13.7	882	.075	.007	.427	4.55
flagellates	199	99	14.6	749	.054	.008	.411	4.30
chrysophytan	5	54	10.5	635	.010	.002	.118	3.73

TABLE B-2
PROCEDURE FOR CALCULATING THE TOTALP(PD) PHYTOPLANKTON TSI USING FOX LAKE, ILLINOIS, AS AN EXAMPLE

Dominant Genera in Fox Lake (STORET No. 1755)	Percent Occurrence	V (TOTALP, from Table 8)
Aphanizomenon	41.2	147
Melosira	15.9	94
Stephanodiscus	15.5	166
		Sum Total = 406

$$\text{TOTALP(PD) phytoplankton TSI} = \frac{406}{3} = 135.6$$

TABLE B-3
PROCEDURE FOR CALCULATING THE TOTALP/CONC(P) PHYTOPLANKTON TSI
USING FOX LAKE, ILLINOIS, AS AN EXAMPLE

Genera Counted in Fox Lake, Illinois (STORET No. 1755)	Percent of Count	C (Algal Units per ml)	V (TOTALP/CONC, Table 8)	V x C
Anabaena	3.7	237	.098	23
Aphanizomenon	41.2	2631	.058	153
Closterium	0.3	22	.007	0
Crucigenia	0.3	22	.696	15
Cyclotella	1.0	65	.073	5
Flagellates	0.3	22	.054	1
Glenodinium	1.7	108	.020	2
Gomphosphaeria	1.7	108	.123	13
Melosira	15.9	1014	.034	34
Microcystis	5.1	324	.056	18
Oocystis	4.1	259	.005	1
Oscillatoria	4.1	259	.014	4
Phormidium	0.3	22	.102	2
Scenedesmus	3.7	237	.058	14
Sphaerocystis	0.7	43	.032	1
Stephanodiscus	15.5	992	.045	45
Synedra	0.3	22	.027	1
			SUM TOTAL =	332

TOTALP/CONC(P) phytoplankton TSI = 332

TABLE B-4
PROCEDURE FOR CALCULATING THE TOTALP/CONC(PD) PHYTOPLANKTON TSI USING FOX LAKE, ILLINOIS, AS AN EXAMPLE

Dominant Genera in Fox Lake, Illinois (STORET No. 1755)	Percent of Count	C (Algal Units Per ml)	V (TOTALP/CONC Table 8)	V x C
Aphanizomenon	41.2	2631	.058	153
Melosira	15.9	1041	.034	34
Stephanodiscus	15.5	992	.045	45
				SUM TOTAL = 232

TOTALP/CONC(PD) phytoplankton TSI = 232

APPENDIX C

CLASSIFICATION, BY VARIOUS AUTHORS, OF THE TOLERANCE OF VARIOUS MACROINVERTEBRATE TAXA TO DECOMPOSABLE WASTES: TOLERANT (T), FACULTATIVE (F), AND INTOLERANT (I)

Source: Weber, 1973

CLASSIFICATION, BY VARIOUS AUTHORS, OF THE TOLERANCE OF VARIOUS MACROINVERTEBRATE TAXA TO DECOMPOSABLE ORGANIC WASTES; TOLERANT (T), FACULTATIVE (F), AND INTOLERANT (I)

Macroinvertebrate	T	F	I
Porifera			
Demospongiae			
Monaxonida			
Spongillidae			9 *
Spongilla fragile		11	
Bryozoa			
Ectoprocta			
Phylactolaemata			
Plumatellidae			
Plumatella repens		13	
P. princeps var. mucosa	11		
P. p. var. mucosa spongiosa		11	
P. p. var. fruticosa	11		
P. polymorpha var. repens			11
Cristatellidae			
Cristatella mucedo		13	
Lophopodidae			
Lophopodella carteri			9
Pectinatella magnifica			†11,9
Endoprocta			
Urnatellidae			
Urnatella gracilis		†11,9	
Gymnalaemata			
Cienostomata			
Paludicellidae			
Paludicella chrenbergi		11	
Coclenterata			
Hydrozoa			
Hydroida			
Hydridae			
Hydra		9	
Clavidae			
Cordylophora lacustris		9	
Platyhelminthes			
Turbellaria		9	
Tricladida			
Planariidae			
Planaria		11	
Nematoda		9	
Nematomorpha			
Gordioida			
Gordiidae		11	
Annelida			
Oligochaeta	4,3	11	
Plesiopora			
Naididae		11	
Nais		9	
Dero		11	
Ophidonais	14		
Stylaria		9	
Tubificidae			
Tubifex tubifex	11,9		
Tubifex	11,6,14		
Limnodrilus hoffmeisteri	11,2,9		
L. claparedianus	11		
Limnodrilus	11,6,14		
Branchiura sowerbyi	9		

Macroinvertebrate	T	F	I
Prosopora			
Lumbriculidae	14		
Hirudinea			
Rhynchobdelida			
Glossiphoniidae			
Glossiphonia complanata	11		
Helobdella stagnalis	11,9		
H. nepheloidea	11		
Placobdella montifera	14		
P. rugosa		11	
Placobdella		9	
Piscicolidae			
Piscicola punctata		14	
Gnathobdellida			
Hirudidae			
Macrobdella	8		
Pharyngobdellida			
Erpobdellidae			
Erpobdella punctata	11		
Dina parva	11		
D. microstoma	11		
Dina		9	
Mooreobdella microstoma	9		
Hydracarina			4
Arthropoda			
Crustacea			
Isopoda			
Asellidae			
Asellus intermedius		11	
Asellus	14	9	4,3
Lirceus		9	
Amphipoda		3	
Talitridae			
Hyallela azteca		4,2 3,9	
H. knickerbockeri	11		
Gammaridai			
Gammarus		9	
Crangonyx pseudogracilis		9	
Decapoda			
Palaemonidae			
Palaemonetes paludosus		4,2 3	
P. exilipes	11		
Astacidae			
Cambarus striatus	7		
C. fodiens	1		
C. bartoni bartoni		1	1
C. b. cavatus		1	
C. conasaugeansis			1
C. asperimanus			1
C. latimanus		1	
C. acuminatus			1
C. hiwassensis			1
C. extraneus			1
C. diogenes diogenes	1		
C. cryptodytes †			1

* Numbers refer to references enumerated in the "Literature" section immediately following this table.
† Albinistic

(Continued)

Macroinvertebrate	T	F	I
C. floridanus		1	
C. carolinus‡	1		
C. longulus longirostris			1
Procambarus raneyi			1
P. acutus acutus	1		
P. paeninsulanus		1	
P. spiculifer			1
P. versutus			1
P. pubescens		1	
P. liosternum		1	
P. enoplosternum		1	
P. angustatus		1	
P. seminolae		1	
P. truculentus‡	1		
P. advena‡	1		
P. pygmaeus‡	1		
P. pubischelae		1	
P. barbatus		1	
P. howellae		1	
P. troglodytes	1		
P. epicyrtus		1	
P. fallax	1		
P. chacei		1	
P. lunzi		1	
Orconectes propinquus		9	
O. rusticus		9	
O. juvenilis			1
O. erichsonianus		1	
Faxonella clypeata		1	
Insecta			
Diptera			
Chironomidae			
Pentaneura inculta		14	2,3
P. carneosa		14,10	14,5
P. flavifrons	4		
P. melanops	10,5		
P. americana			10,5
Pentaneura			9,10
Ablabesmyia janta		2,3 9	
A. americana		11,14	4
A. illinoense	5	10	
A. mallochi		9	3
A. ornata			3
A. aspera			3
A. peleensis		3	
A. auriensis			3
A. rhamphe		9	
Ablabesmyia			9
Procladius culiciformis	14	10,5	
P. denticulatus	9		
Procladius	5	3,10 5	
Labrundinia floridana			3
L. pilosella			9
L. virescens			3
Guttipelopia		9	
Conchapelopia		9	
Coelotanypus scapularis		9	
C. concinnus	9	11,14 10,5	10

Macroinvertebrate	T	F	I
Psilotanypus bellus	9		
Tanypus stellatus	10,5	6,14	4
T. carinatus		9	
T. punctipennis		10,5	
Tanypus		10,5	
Psectrotanypus dyari	10,5	11	
Psectrotanypus		10	
Larsia lurida		3	
Clinotanypus caliginosus			10,5
Clinotanypus		3	
Orthocladius obumbratus			14
Orthocladius		4,10	14,9 10,5
Nanocladius			3,9
Psectrocladius niger		9	
P. julia		9	
Psectrocladius			3,10
Metriocnemus lundbecki			3
Cricotopus bicinctus			2,3 10,5
C. bicinctus group	9		
C. exilis		10	5
C. exilis group		9	
C. trifasciatus		10	5
C. trifasciatus group		9	
C. politus			10,5
C. tricinctus		10	5
C. absurdus			6,10 5
Cricotopus			10
Corynoneura taris			3
C. scutellata			10,5
Corynoneura			4,9 5
Thienemanniella xena			3,9
Thienemanniella			3,10
Trichocladius robacki			2,3
Brillia par			3
Diamesa nivoriunda			6,9 10
Diamesa			14
Prodiamesa olivacea			5
Chironomus attenuatus group	4,3 9,5		10
C. riparius	6,10 5		
C. riparius group	9		
C. tentans			5
C. tentans-plumosus	14		
C. plumosus	11,6 14		11,5
C. plumosus group	9		
C. carus	3		
C. crassicaudatus	3		
C. stigmaterus	3		
C. flavus		14	
C. equisitus		14	
C. fulvipilus	3		
C. anthracinus			5
C. paganus			5
C. staegeri		5	

‡Not usually inhabitant of open water; are burrowers.

(Continued)

Macroinvertebrate	T	F	I
Chironomus	4	14	
Kiefferulus dux	3		10,5
Cryptochironomus fulvus	2,3		10,5
C. fulvus group		9	
C. digitatus		11	5
C. sp. B (Joh.)			4
C. blarina		9	5
C. psittacinus			14
C. nais		9	
Cryptochironomus	4		
Chaetolabis atroviridis			5
C. ochreatus			5
Endochironomus nigricans		3,9	10,5
Stenochironomus macateei			9,10
S. hilaris			2,3
Stictochironomus devinctus			3,5
S. varius			10
Xenochironomus xenolabis			9
X. rogersi		9	
X. scopula			10,5
Pseudochironomus richardsoni			10,5
Pseudochironomus			5
Parachironomus abortivus group		9	
P. pectinatellae		9	
Cryptotendipes emorsus		9	
Microtendipes pedellus			10,5
Microtendipes			9
Paratendipes albimanus			10,5
Tribelos jucundus			5
T. fuscicornis			9
Harnischia collator		9	
H. tenuicaudata			10
Phaenopsectra			9
Dicrotendipes modestus		9	
D. neomodestus		10	9,5
D. nervosus		9	5
D. incurvus	9		
D. fumidus			9,5
Glyptotendipes senilis			9
G. paripes	3		5
G. meridionalis		9	
G. lobiferus	11,3 9		10,5
G. barbipes	9		
G. amplus		9	
Glyptotendipes	5		
Polypedilum halterale		9	3,5
P. fallax		4,10 5	3
P. scalaenum	3	9	
P. illinoense		2,3 9,10	10,5
P. tritum		9	
P. simulans		9	5
P. nubeculosum			5
P. vibex			10
Polypedilum		11,10	5
Tanytarsus neoflavellus		10,5	6
T. gracilentus			5
T. dissimilis			9
Rheotanytarsus exiguus	4		2,3
Rheotanytarsus		9	

Macroinvertebrate	T	F	I
Cladotanytarsus		9	
Micropsectra dives		14	5
M. deflecta			9
M. nigripula			10,5
Calopsectra gregarius	4		
Calopsectra			10,5
Stempellina johannseni		10	5
Culicidae	3		
Culex pipiens	6,10		
Anopheles punctipennis			10
Chaoboridae			
Chaoborus punctipennis		14,9	10
Ceratopogonidae	4,3	9	
Palpomyia tibialis		14	
Palpomyia		11,14	
Bezzia glabra	10		
Stilobezzia antenalis	10		
Tipulidae	3	9	
Tipula caloptera			10
T. abdominalis			10
Pseudolimnophila luteipennis			10
Hexatoma			10
Eriocera		14	
Psychodidae	3		
Psychoda alternata	10		
P. schizura	10		
Psychoda	9		
Telmatoscopus albipunctatus	14		
Telmatoscopus			10
Simuliidae	9	10	4,3
Simulium vittatum		6,10	
S. venustrum			10
Simulium			2
Prosimulium johannseni			10
Cnephia pecuarum			10
Stratiomyidae	3		
Stratiomys discalis	10		
S. meigeni	10		
Odontomyia cincta		10	
Tabanidae	3		
Tabanus atratus	6	10	
T. stygius		10	
T. benedictus	10		
T. giganteus			10
T. lineola	10		
T. variegatus			10
Tabanus			10
Syrphidae	3		
Syrphus americanus	10		
Eristalis bastardi	6,10		
E. aeneus	10		
E. brousi	10		
Eristalis	10		
Empididae		9	
Ephydridae			
Brachydeutera argentata	10		
Anthomyiidae		9	
Lepidoptera			
Pyralididae		4,3	
Trichoptera			
Hydropsychidae			
Hydropsyche orris		9	

(Continued)

Macroinvertebrate	T	F	I
H. bifida group		9	
H. simulans			9
H. frisoni			9
H. incommoda		11	4,2,3
Hydropsyche			4,3
Cheumatopsyche		4,14	
		2,3	
		9	
Macronemum carolina			4,2,3
Macronemum			9
Potamyia flava		9	
Psychomyiidae			
Psychomyia			9
Neureclipsis crepuscularis			9
Polycentropus		9	4,11
			3
Cyrnellus fraternus		9	
Oxyethira			4,3
Rhyacophilidae			
Rhyacophila			11
Hydroptilidae			
Hydroptila waubesiana			9
Hydroptila			4,2,3
Ochrotrichia			9
Agraylea			9
Leptoceridae			11
Leptocella		4,3	9
Athripsodes			9
Oecetis		4,3	
Philopotamidae			
Chimarra perigua			2,3
Chimarra			4,3
Brachycentridae			
Brachycentrus			3
Molannidae			11
Ephemeroptera			
Heptageniidae			
Stenonema integrum		15,9	
S. rubromaculatum			15
S. fuscum			15
S. pulchellum		15	
S. ares		15	
S. scitulum		9	
S. femoratum		6,9	15
S. terminatum			9
S. interpunctatum			15,9
S. i. ohioense			15
S. i. canadense			15
S. i. heterotarsale		15	
S. exiguum			4,2,3
S. smithae			4,2,3
S. proximum			2
S. tripunctatum			15
Stenonema			15
Hexageniidae			
Hexagenia limbata			9
H. bilineata		14	11
Pentagenia vittigera			9
Baetidae			
Baetis vagans			9
Callibaetis floridanus	3		
Callibaetis		6	

Macroinvertebrate	T	F	I
Caenidae			
Caenis diminuta	3		
Caenis		9	11
Tricorythidae		9	
Siphlonuridae			
Isonychia			9
Plecoptera			4,3
Perlidae			
Perlesta placida		6	2
Acroneuria abnormis		9	
A. arida			9
Nemouridae			
Taeniopteryx nivalis			9
Allocapnia vivipara		6	
Perlodidae			
Isoperla bilineata			9
Neuroptera			
Sisyridae			
Climacia areolaris			9
Megaloptera			
Corydalidae			
Corydalus cornutus		9	4,2,3
Sialidae			
Sialis infumata			11
Sialis		9	
Odonata			
Calopterygidae			
Hetaerina titia			3
Agrionidae			
Argia apicalis		9	
A. translata		9	
Argia			4,3
Ischnura verticalis	11	9	
Enallagma antennatum		9	
E. signatum		9	11
Aeshnidae			
Anax junius			11
Gomphidae			
Gomphus pallidus		4,2,3	
G. plagiatus			11
G. externus			11
G. spiniceps		9	
G. vastus		9	
Gomphus		4,3	
Progomphus			4,3
Dromogomphus		9	
Erpetogomphus		9	
Libellulidae			
Libellula lydia		6	
Neurocordulia moesta		9	
Plathemis		9	
Macromia		4,9	3
Hemiptera	3		
Corixidae		9	
Corixa	6		
Hesperocorixa	6		
Gerridae			
Gerris	6		
Belostomatidae			
Belostoma	6,2		
Hydrometridae			
Hydrometra martini	2		

(Continued)

Macroinvertebrata	T	F	I
Coleoptera	3§		
Elmidae			
Stenelmis crenata			6,12
S. sexlineata		9,12	6
S. decorata	12		
Dubiraphia		9,12	
Promoresia			12
Optioservus		12	
Macronychus glabratus			12
Ancyronyx variegatus			12
Microcylloepus pusillus			12
Gonielmis dietrichi		12	
Hydrophilidae			
Berosus	9		
Tropisternus natator	6		
T. lateralis	2		
T. dorsalis			11
Dytiscidae			
Laccophilus maculosus	6		
Gyrinidae			
Gyrinus floridanus	2		
Dineutus americanus	6		
Dineutus		9	
Mollusca			
Gastropoda			
Mesogastropoda			
Valvatidae			
Valvata tricarinata		8	11,8
V. piscinalis		8	
V. bicarinata			11
V. b. var. normalis			11
Viviparidae			
Viviparus contectoides			11
V. subpurpurea			11
Campeloma integrum		8	
C. rufum		8	
C. contectus		8	
C. fasciatus		8	
C. decisum			8
C. subsolidum		11,8	
Campeloma		14	
Lioplax subcarinatus			11
Pleuroceridae			
Pleurocera acuta		11,8	
P. elevatum		8	
P. e. lewisi		8	
Pleurocera		8	
Goniobasis livescens		11,8	
G. virginica	8		
Goniobasis		8	4,3
Anculosa		8	
Bulimidae			
Bulimus tentaculatus		8	
Amnicola emarginata			11
A. limosa			11
Somatogyrus subglobosus			11
Basommatophora			
Physidae			
Physa integra	6,8	8	
P. heterostropha	8	8	
P. gyrina		8	
P. acuta		8	8
P. fontinalis		8	8
P. anatina	8		
P. halei	8		
P. cubensis	8		
P. pumilia	2		
Physa	4,3		
Aplexa hypnorum		8	8
Lymnaeidae			
Lymnaea ovata	8		
L. peregra		8	
L. caperata		8	
L. humilis		8	
L. obrussa		8	
L. palustris		8	8
L. auricularia		8	
L. stagnalis		8	8
L. s. appressa			8
Lymnaea	3	9	
Pseudosuccinea columella		8	
Galba catascopium	8		
Fossaria modicella	8		
Planorbidae			
Planorbis carinatus			
P. trivolvis	8		
P. parvus	8		
P. corneus		8	8
P. marginatus			8
Planorbis		8	
Segmentina armigera	8		
Helisoma anceps		8	
H. trivolvis		8	
Helisoma	2,3		
Gyraulus arcticus		8	
Gyraulus		8	
Ancylidae			
Ancylus lacustris		8	8
A. fluviatilis		8	8
Ferrissia fusca		8	
F. tarda		8	
F. rivularis			8
Ferrissia	4,2,3	9	
Bivalvia			
Eulamellibranchia			
Margaritiferidae			
Margaritifera margaritifera			8
Unionidae			
Unio complanata	8		
U. gibbosus	8	8	
U. batavus			8
U. pictorum			8
U. tumidus		8	
Lampsilis luteola		8	
L. alata		8	
L. anodontoides		8	
L. gracilis		11	
L. parvus			
Lampsilis		11,9	
Quadrula pustulosa		8,9	

§ Except riffle beetles

(Continued)

Macroinvertebrate	T	F	I
Q. undulata		8	
Q. rubiginosa		8	
Q. lachrymosa		8	
Q. plicata		8	
Truncilla donaciformis			11
T. elegans			11
Tritigonia tuberculata		8	
Symphynota costata		8	
Strophitus edentulus		8	
Anodonta grandis		8,9	
A. imbecillis		8,9	
A. mutabilis			8
Alasmodonta costata		8	
Proptera alata			9
Leptodea fragilis			9
Amblema undulata		8	
Lasmigona complanata		8	
Obliquaria reflexa			14
Heterodonta			
Corbiculidae			
Corbicula manilensis			9
Sphaeriidae	4,3		
Sphaerium notatum	8		
S. corneum		8	
S. rhomboideum		8	
S. striatinum		8	
S. s. var. corpulentum		11	
S. s. var. illycashense		11	
S. sulcatum		8	
S. stamineum		11,8	
S. [illegible]		8	8
S. vivicolum		8	8
S. solidulum			8
Sphaerium		9	
Musculium securis		8	
M. transversum	11,8	8	
M. truncatum	11	8	
Musculium	14		
Pisidium abditum	8		
P. fossarinum			8
P. pauperculum crystalense		11,8	
P. amnicum		8	8
P. casertanum			
P. compressum	11	8	
P. fallax		8	
P. henslowanum		8	
P. idahoense	8		
P. complanatum	11,8	11,8	
P. subtruncatum		8	
Pisidium		11	
Dreissenidae			
Mytilopsis leucophaeatus		8	
Mactridae			
Rangia cuneata		8	

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APPENDIX D

KEY TO CHIRONOMID ASSOCIATIONS OF THE PROFUNDAL ZONES OF PALEARCTIC AND NEARARCTIC LAKES

Source: Seather, 1979

APPENDIX D

Key to chironomid associations of the profundal zones of Palaearctic and Nearctic lakes

In the key "absent" means less than 1% as accidental occurrence may take place, "present" means more than 1%. The limit of 2% is regarded as the level above which the species can be regarded as a persistant non-accidental member of the community, while the 5% limit is a level above which the species can be said to be a common member of the community. These limits should of course not be regarded rigidly if the samples are few.

1. Pseudodiamesa and/or Oliveria tricornis present ...... α-oligotrophic
   The above absent 2

2. Heterotrissocladius, Protanypus, Micropsectra or Paracladopelma present and making up at least 2% of the profundal chironomids ..... oligo- mesotrophic lakes 3
   The above absent or making up less than 2% of the profundal chironomids .... eutrophic lakes 10

3. Heterotrissocladius subpilosus - group present, tribe Chironomini absent from the true profundal zone .... β-oligotrophic
   H. subpilosus group present or absent, tribe Chironomini present 4

4. Heterotrissocladius subpilosus group, Protanypus caudatus group, Micropsectra groenlandica or Paracladius spp. present and making up more than 5% of the profundal chironomids 5
   The above absent or making up less than 5% of the profundal chironomids 7

5. Protanypus caudatus group or Paracladius usually present, Chironomus absent, Phaenopsectra (including Sergentia) and Stictochironomus at most present in very low numbers (<2%) .... γ-oligotrophic
   When Protanypus caudatus group or Paracladius present, Chironomus, Phaenopsectra or Stictochironomus present in low numbers (>2%) 6

6. Heterotrissocladius subpilosus group plus H. maeaeri group more common than H. marcidus group; Chironomus making up less than 2% .... δ-oligotrophic
   Heterotrissocladius subpilosus group plus H. maeaeri group absent or less common than H. marcidus group: Chironomus usually makes up more than 2% .... ϵ-oligotrophic

7. Heterotrissocladius, Paracladopelma nigritula, P. galaptera, Micropsectra notescens group, Monodiamesa tuberculata, Macropelopia fehlmanni and/or Tanytarsus bathophilus common (>5%) ζ-oligotroph
   The above at most present in very low numbers 8

8. *Micropsectra* and/or *Monodiamesa* common, more or about as common as *Stictochironomus* and *Phaenopsectra*, or *Chironomus* except *salinarius* or *semireductus* types .... η -mesotrophic
Micropsectra and/or *Monodiamesa* less common than *Stictochironomus* and *Phaenopsectra* or spp. of *Chironomus* except *salinarius* or *semireductus* types 9

9. *Monodiamesa*, *Protanypus*, *Heterotrissocladius*, *Stictochironomus*, *Phaenopsectra* or *Chironomus salinarius* and *semireductus* types more common than other *Chironomus* spp .... θ -mesotrophic
The above less common than other *Chironomus* .... ι -mesotrophic

10. *Heterotrissocladius*, *Protanypus*, *Micropsectra*, *Paracladopelma nigritula* or *P. galaptera* present in low numbers ..... κ -eutrophic
The above absent 11

11. No chironomids present .... o -eutrophic
Chironomids present 12

12. Only *Chironomus plumosus* type and *Tanypodinae* present ξ -eutrophic
Other chironomids also present 13

13. Only *Chironomus* and subfam. *Tanypodinae* present ...... ν -eutrophic
Other groups also present 14

14. Only tribe Chironomini, *Tanytarsus* spp. and subfam. Tanypodinae present .... μ -eutrophic
Other groups also present .... λ -eutrophic.

TABLE D-1
CHARACTERISTIC PROFUNDAL CHIRONIMIDS IN NEARARCTIC(-----) AND PALEEARCTIC(......) LAKES. FULLY DRAWN LINES AND FILLED CIRCLES: DISTRIBUTION UNDER GOOD TO EXCELLENT CONDITIONS. BROKEN LINES AND DOTS: MAXIMUM RANGE OR SINGLE FINDINGS. A: IN EUROPE, ALPINE. B: IN EUROPE, BOREAL.

SPECIES	A	B
Pseudodiamesa nivosa Goetgh.	x	x
Pseudodiamesa arctica (Mall.)		
Oliveria tricornis (Ol.)		x
Lauterbornia sedna (Ol.)		
Paracladius quadrinodosus Hirv.		
Protanypus caudatus (Edw.)		x
Heterotrissocladius subpilosus (Kieff.)		x
Heterotrissocladius oliveri Saeth.		
Monodiamesa ekmani Brund.		x
Protanypus saetheri Wied.		
Tanytarsus palmeni Lind.		x
Lauterbornia coracina Kieff.	x	x
Paracladius alpicola (Zett.)	x	x
Monodiamesa alpicola Brund.	x	
Protanypus forcipatus Egg.	x	
Micropsectra groenlandica And.	x	x
Heterotrissocladius maeaeri Brund.		x
Protanypus hamiltoni Saeth.		
Micropsectra lindebergi Säw.		x
Tanytarsus lugens Kieff.	x	x
Heterotrissocladius sp. *A* near *subpilosus*		
Heterotrissocladius sp. *B* near *maeaeri*		
Protanypus ramosus Saeth.		
Micropsectra contracta Reiss	x	
Micropsectra insignilobus Kieff.		x
Paracladopelma galaptera (Town.)		
Paracladopelma nigritula (Goetgh.)	x	x
Monodiamesa tuberculata Saeth.		
Macropelopia fehlmanni (Kieff.)	x	
Tanytarsus bathophilus Kieff.	x	x
Protanypus morio Zett.	x	x
Heterotrissocladius changi Saeth.		
Heterotrissocladius scutellatus Goetgh.	x	
Heterotrissocladius sp. *D* near *changi*		
Protanypus sp. *A* near *morio*		
Protanypus sp. *B* near *morio*		
Tanytarsus decipiens Lind.		x
Monodiamesa nitida (Kieff.)	x	
Heterotrissocladius grimshawi Edw.		x
Monodiamesa sp. poss. *prolilobata* Saeth.		
Monodiamesa bathyphila (Kieff.)	x	x
Stictochironomus rosenschoeldi (Zett)	x	x
Phaenopsectra coracina (Zett.)	x	x
Tanytarsus n. sp. *lestagei* - aggl.		
Monodiamesa depectinata Saeth.		
Chironomus atritibia Mall		
Chironomus anthracinus Zett.	x	x
Tanytarsus inaequalis Goetgh.	x	x
Tanytarsus gregarius Kieff.	x	x
Chironomus plumosus f. *semireductus*		
Cryptotendipes casuarius (Town.)		
Chironomus decorus Joh.		
Cryptotendipes darbyi (Subl.)		
Chironomus plumosus L.	x	x
Zalutschia zalutschicola Lip.		x
Chironomus tenuistylus Brund.		x

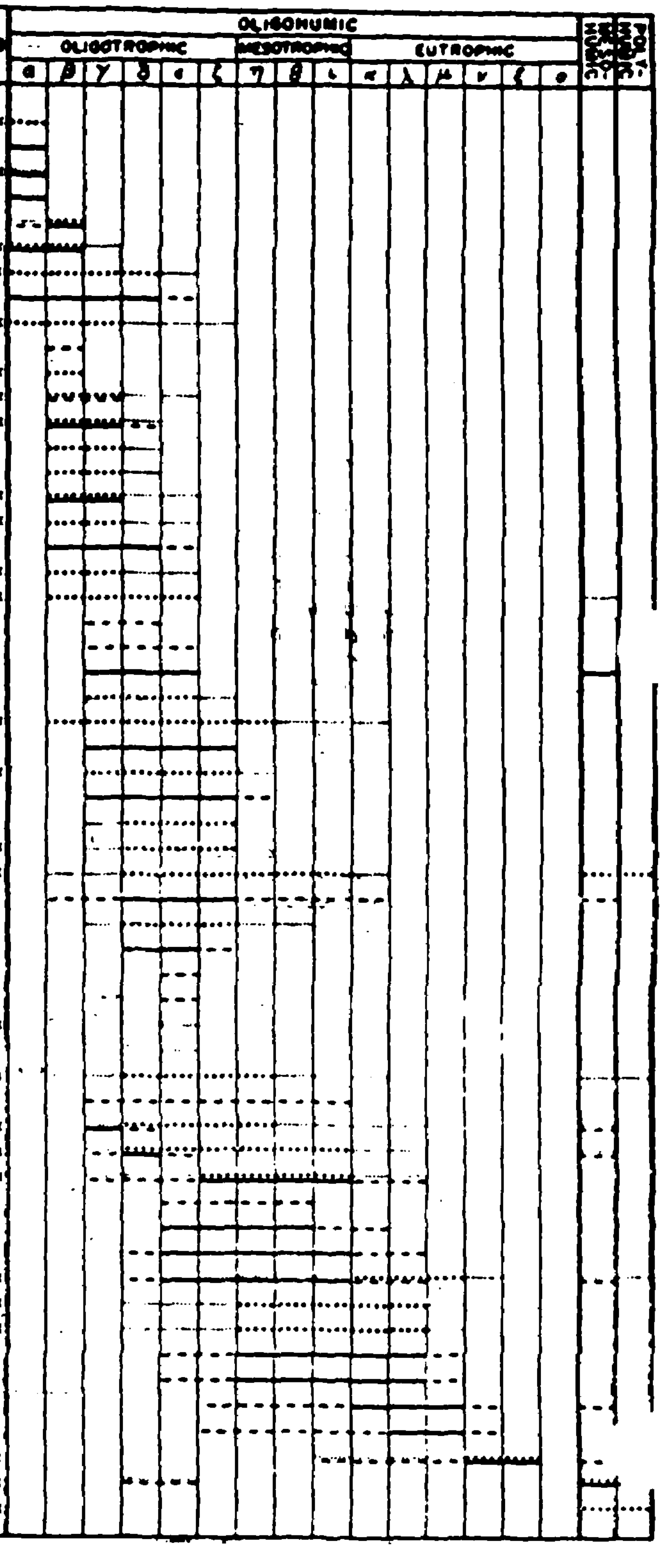

TABLE D-2

CHARACTERISTIC SUBLITTORAL AND LITTORAL CHIRONOMID HABITATS IN NEARACRTIC AND PALEARCTIC LAKES.

SPECIES	A	B
Heterotrissocladius subpilosus (Kieff.)		x
Heterotrissocladius oliveri Sæth.		
Hydrobaenus fusistylus (Goetgh.)		
Zalutschia trigonacies Sæth.		
Abiskomyia virgo Edw.		x
Oeklandia borealis Kieff.		
Orthocladius (O.) trigonolabis Edw.		x
Orthocladius (P) consobrinus Holmgr.)	x	x
Heterotrissocladius maeaeri Brund.		x
Oliveria tricornis (Ol.)		x
Lauterbornia sedna Ol.		
Hydrobaenus martini Sæth.		x
Hydrobaenus conformis conformis (Holmgr.)		x
Hydrobaenus conformis labradorensis Sæth.		
Monodiamesa ekmani Brund.		x
Paracladius quadrinodosus Hirv.		x
Zalutschia tornetraeskensis (Edw.)		x
Tanytarsus lugens Kieff.	x	x
Paratanytarsus hyperboreus Brund.		x
Tanytarsus niger And.		x
Paracladius alpicola (Zett.)	x	x
Paracladopelma nigritula (Goetgh.)	x	x
Stictochironomus rosenschoeldi (Zett.)	x	x
Micropsectra groenlandica And.	x	x
Arctopelopia barbitarsis (Zett.)	x	x
Micropsectra lindebergi Säw.		x
Thienemannimyia fusciceps (Edw.)		x
Mesocricotopus thienemanni (Goetgh.)		x
Lauterbornia coracina Kieff.	x	x
Micropsectra insignilobus Kieff.		x
Heterotrissocladius marcidus (Walk.)	x	x
Paracladopelma galaptera (Town.)		
Zalutschia obsepta (Webb)		
Heterotrissocladius hirtapex Sæth.		
Micropsectra contracta Reiss	x	
Heterotanytarsus perennis Sæth.		
Heterotanytarsus nudalus Sæth.		
Nanocladius (N.) rectinervis (Kieff.)	x	x
Paracladopelma nais (Town.)		
Stempellina bausei (Kieff.)	x	x
Zalutschia zalutschicola Lip.		x
Nanocladius (N.) incomptus Sæth.		
Nanocladius (N) minimus Sæth.		
Protanypus morio (Zett.)	x	x
Phaenopsectra coracina (Zett.)	x	x
Paratanytarsus nelvigi (Goetgh.)		x
Heterotanytarsus apicalis (Kieff.)	x	x
Paratanytarsus penicillatus Goetgh.		x
Stempellina n sp near almi Brund.		
Nanocladius (N.) andersoni Sæth.		

TABLE D-2
CHARACTERISTIC SUBLITTORAL AND LITTORAL CHIRONOMIDS OF HABITATS IN NEARACRTIC AND PALEARCTIC LAKES (Continued)

SPECIES	A	B
Paracladopelma winnelli Jacks.		
Paracladopelma undine (Town)		
Stempellinella minor (Edw)	x	x
Stempellinella brevis Edw.	x	x
Pagastiella orophila (Edw.)	x	x
Pagastiella ostansa (Webb)		
Nanocladius (N.) distinctus (Mall)		
Cladopelma edwardsi (Krus.)	x	x
Zalutschia lingulata Saeth.		
Saetheria tylus (Town.)		
Heterotrissocladius latilaminus Saeth.		
Pseudochironomus rex Haub.		
Phaenopsectra albescens (Town.)	x	x
Psectrocladius (P.) psilopterus Kieff.	x	x
Monopelopia tenuicalcar (Kieff.)	x	x
Cladopelma viridula (Fabr.)	x	x
Nanocladius (N.) bicolor (Zett.)	x	x
Robackia demeijerei (Krus.)	x	x
Cryptotendipes casuarius (Town.)		
Monodiamesa depectinata Saeth.		
Pseudochironomus fulviventris (Joh)		
Psectrocladius (P.) simulans (Joh)		
Chironomus plumosus f. *semireductus*		x
Pseudochironomus pseudoviridis (Mall.)		
Nanocladius (N.) balticus (Palm.)	x	x
Chironomus anthracinus Zett.	x	x
Harnischia curtilamellata (Mall.)	x	x
Stictochironomus histrio (Fabr.)	x	x
Demicryptochironomus vulneratus (Zett.)	x	x
Cryptotendipes darbyi (Subl.)		
Chironomus decorus (Joh.)		
Dicrotendipes nervosus (Staeg.)	x	x
Endochironomus subtendens (Town.)		
Endochironomus nigricans (Joh.)		
Endochironomus albipennis (Meig)	x	x
Cricotopus (I.) sylvestris (Fabr)	x	x
Chironomus plumosus L.	x	x
Glyptotendipes (P.) paripes Edw.	x	x
Polypedilum (Pe.) nubeculosum (Meig)	x	x
Cladotanytarsus wexionensis Brund.	x	x
Cladotanytarsus near *wexionensis*		
Tanytarsus usmaensis Pag.	x	x
Einfeldia synchrona (Ol.)		
Einfeldia dissidens (Walk.)	x	x
Chironomus (Ca) tentans Fabr.	x	x
Tanypus punctipennis (Meig.)	x	x
Labrundinia longipalpis (Goetgh)		x
Psectrocladius (A.) platypus Edw.	x	x
Zalutschia mucronata (Brund)		x