TECHNICAL SUPPORT MANUAL:
WATERBODY SURVEYS AND ASSESSMENTS FOR CONDUCTING USE ATTAINABILITY ANALYSES

VOLUME III: LAKE SYSTEMS

U.S. ENVIRONMENTAL PROTECTION AGENCY
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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:

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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). 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:

- Tectonic (depression due to earth movement)
- Volcanos
- Landslides
- Glaciers
- Solution (depressions from soluble rock)
- River activity
- Wind-formed basins
- Shoreline activity
- 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:

- Surface area, A (measured in units of length squared, L^2)
- Volume, V (measured in units of length cubed, L^3)
- Inflow and outflow, Q_{in} and Q_{out} (measured in units of length cubed per time, L^3/T)
- Mean depth, $\bar{d}$
- Maximum depth
- Length
- Length of shoreline
- Depth-area relationships
- Depth-volume relationships
- 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

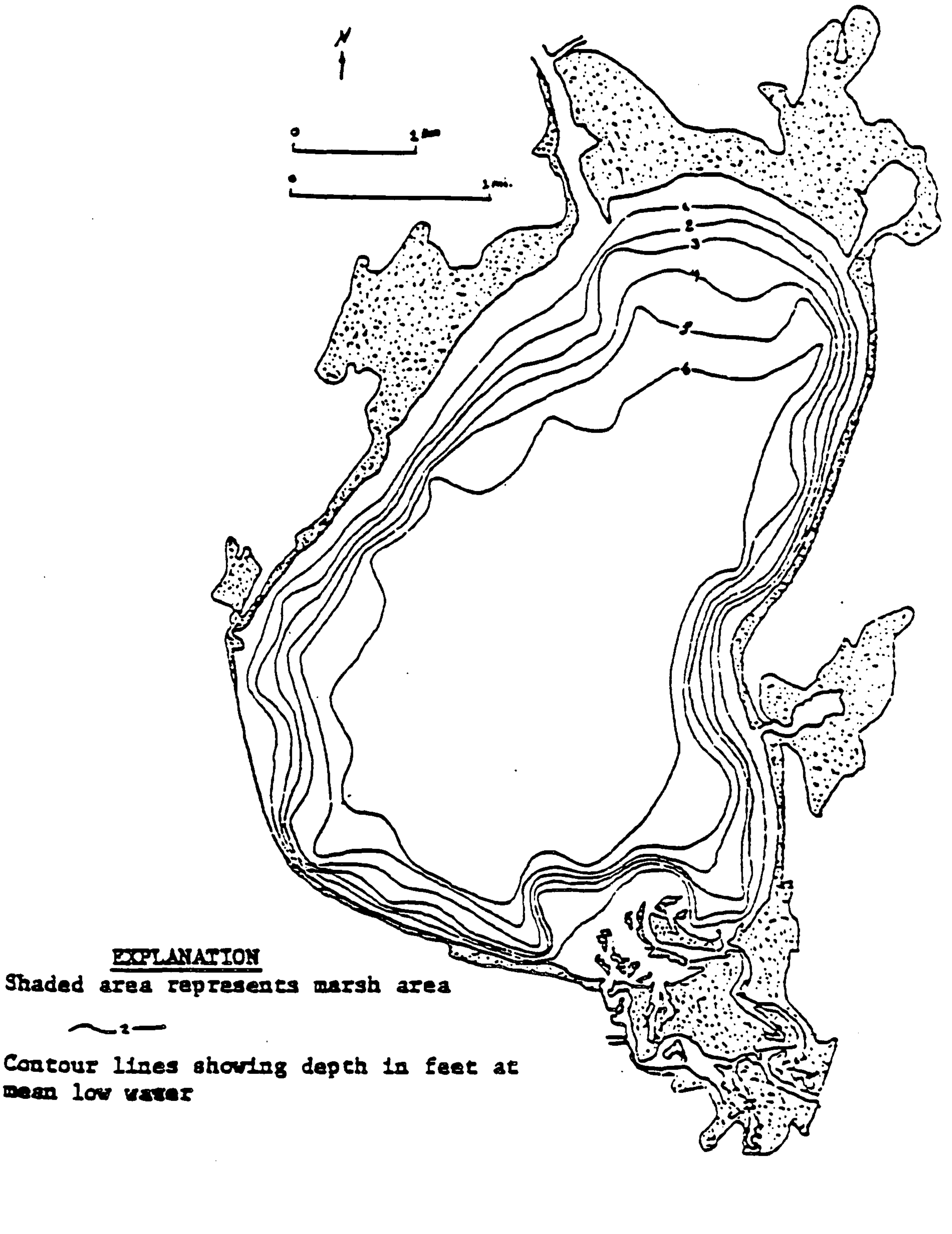


Figure II-1. Bathymetric Map of Lake Harney, Florida (from Brezonik, 1976)

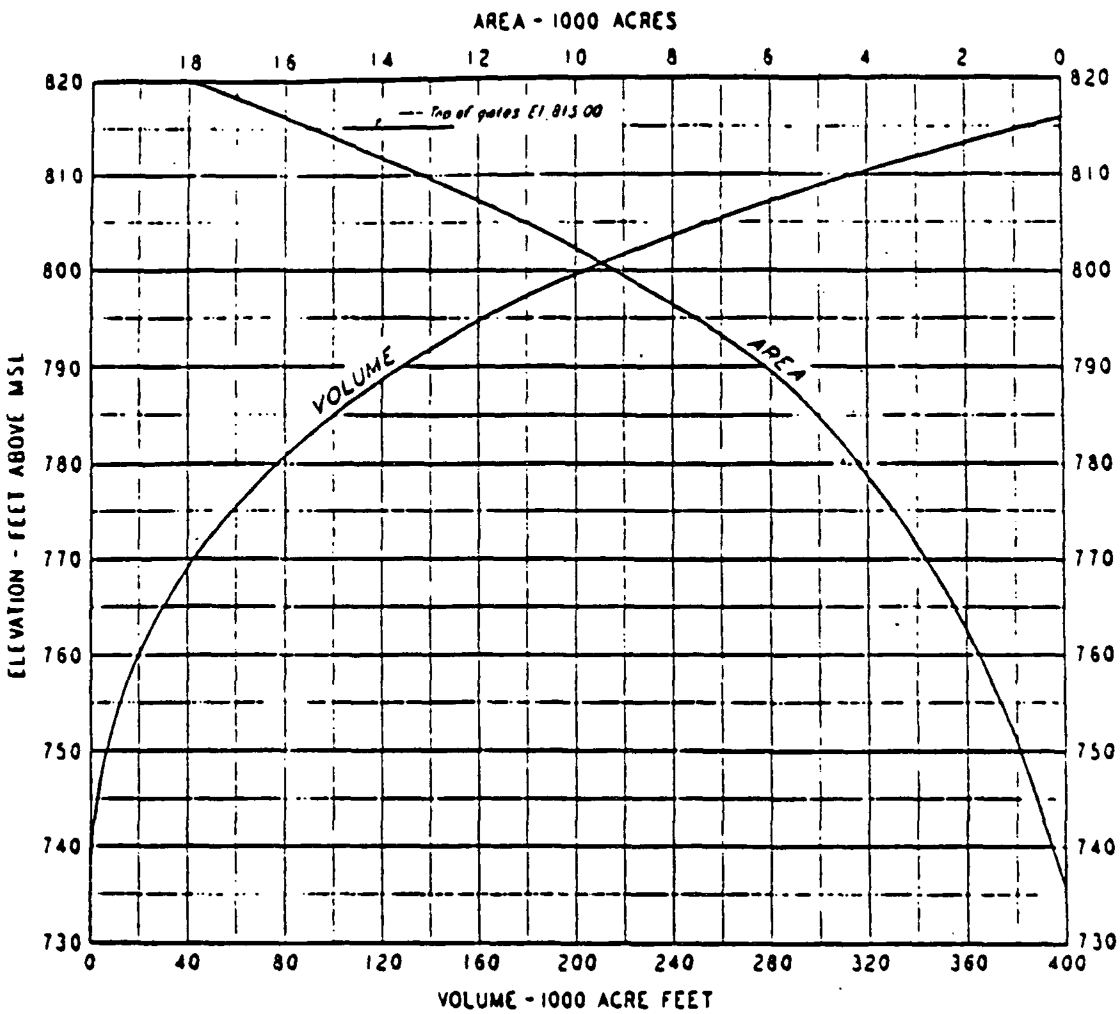


Figure II-2. Fort Loudoun Reservoir Areas and Volumes (from Water Resources Engineers, 1975)

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 motion of the

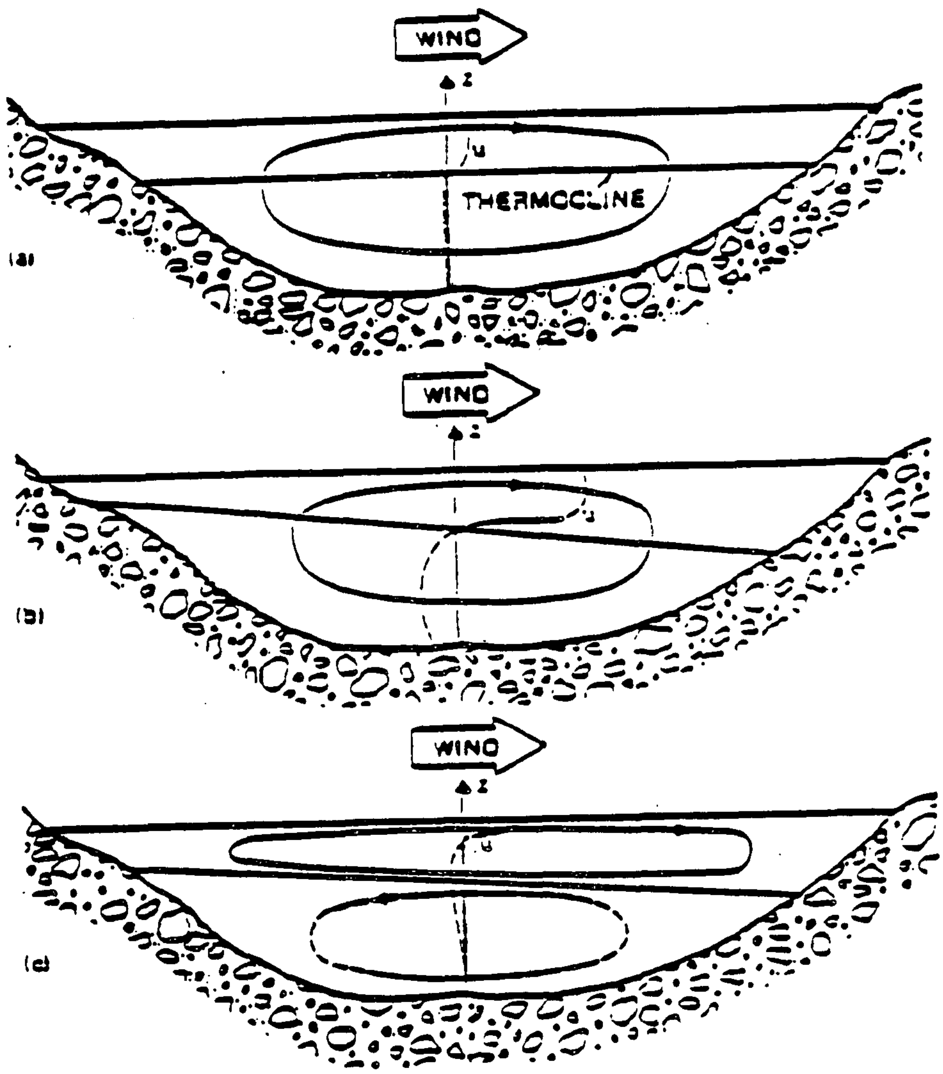


Figure II-3. Formation of baroclinic motions in a lake exposed to wind stresses at the surface: (a) initiation of motion, (b) position of maximum shear across the thermocline (c) steady-state baroclinic circulation (from Fischer, 1979)

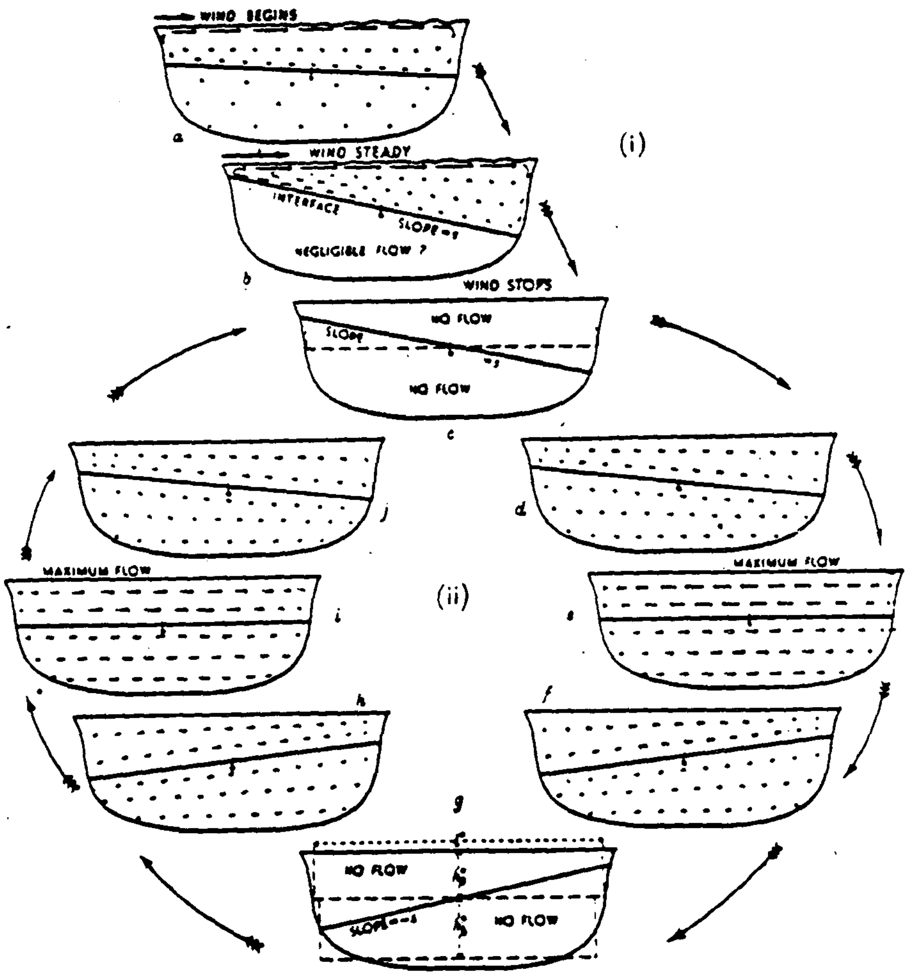


Figure II-4. Movement caused by (i) wind stress and (ii) a subsequent internal seiche in a hypothetical two-layered lake, neglecting friction. Direction and velocity of flow are approximately indicated by arrows. o = nodal section. (from Mortimer, 1952)

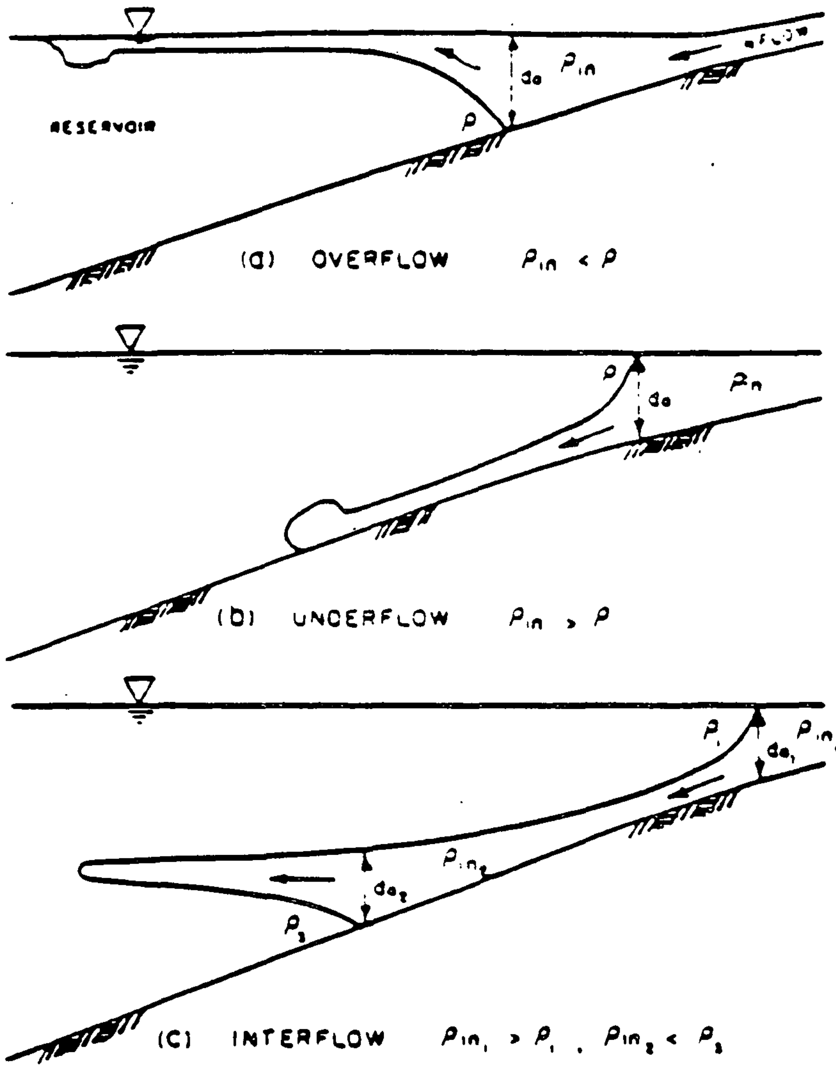


FIGURE II-5. Types of inflow into lakes and reservoirs (from Wunderlich, 1971) earth's rotation) in the Northern Hemisphere and to the left in the Southern Hemisphere. The Coriolis effect causes the surface water to move to the right of the prevailing direction of the wind. Under these conditions in 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:

$$H_N = H_{sn} + H_{an} - (H_b + H_e \pm H_c) \tag{1}$$

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

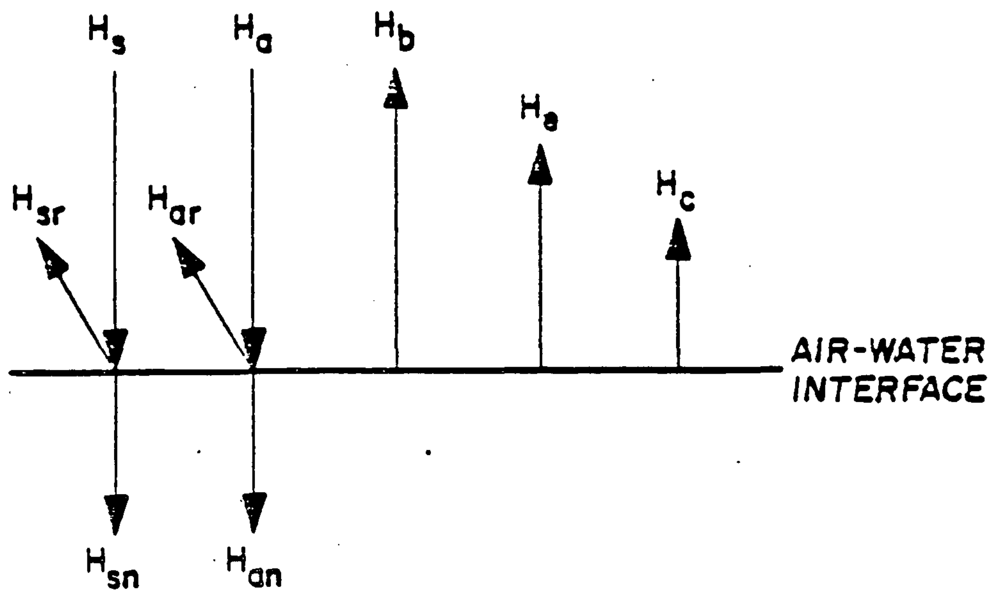


Figure II-6. Heat Transfer Terms Associated with Interfacial Heat Transfer (from Roesner, 1981)

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.

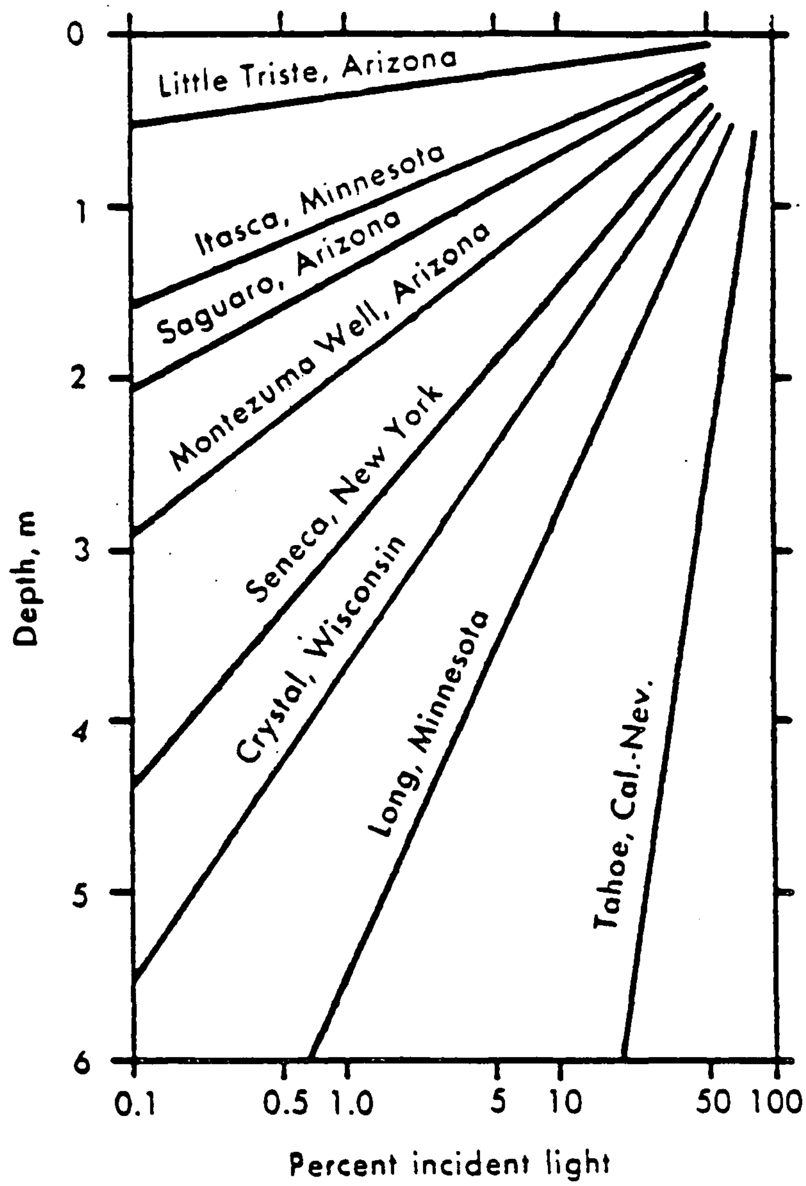


FIGURE II-7. Vertical penetration of light in various bodies of water showing percentage of incident light remaining at different depths (from Cole, 1978)

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

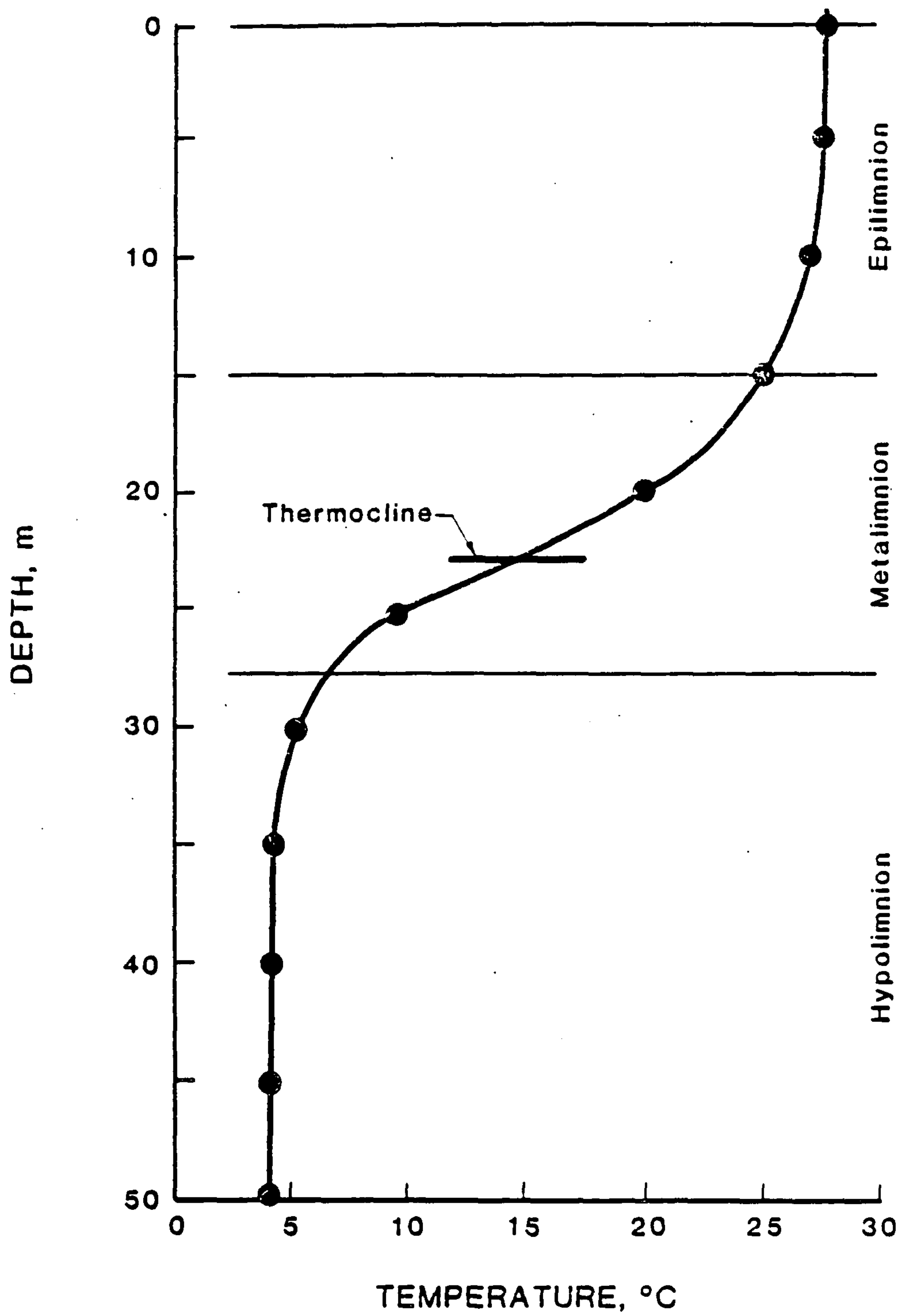


FIGURE II-8. Vertical temperature profile showing direct stratification and the lake regions defined by it (from Cole, 1979).

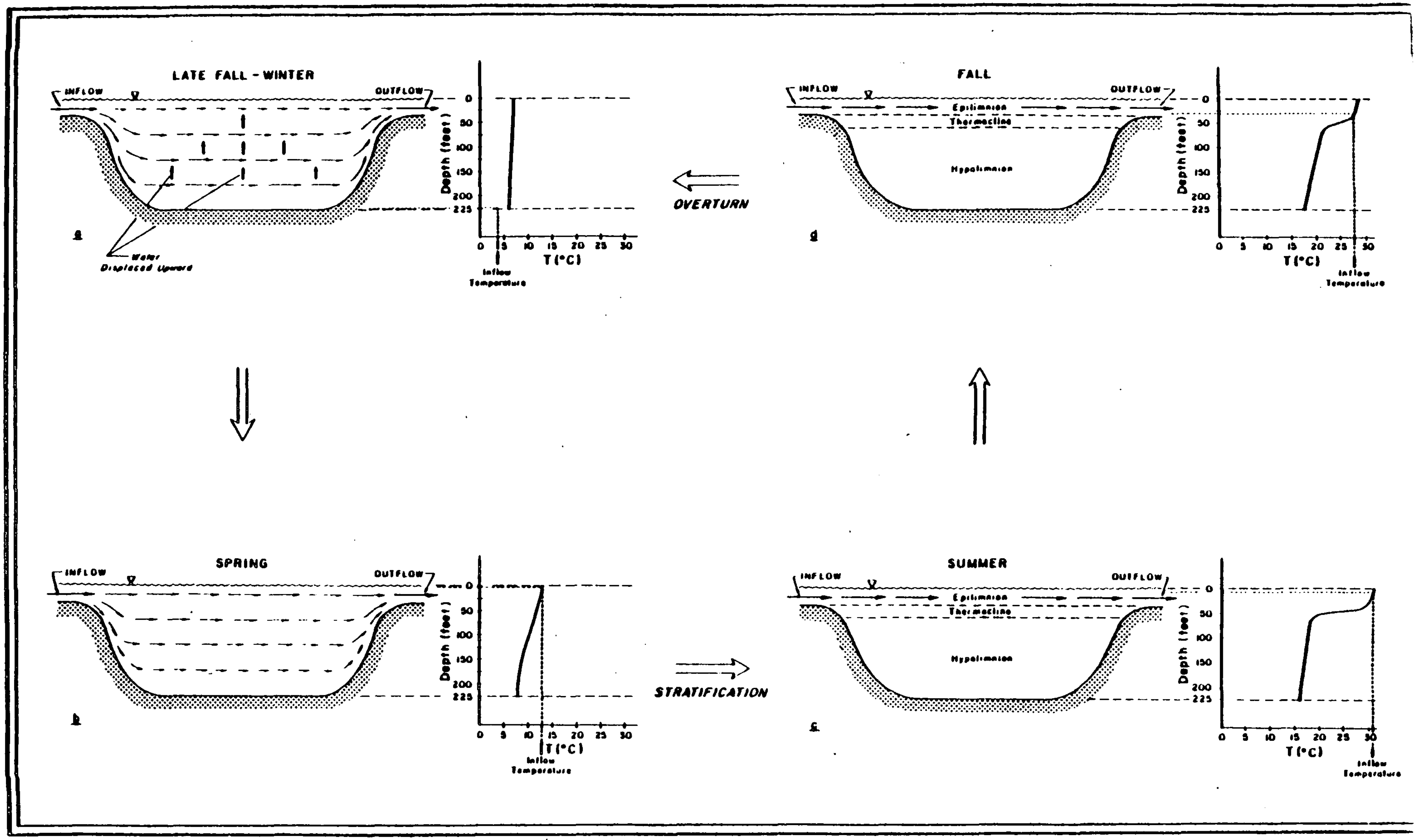


Figure II-9. Annual Cycle of Thermal Stratification and Overturn in an Impoundment (from Zison et al, 1977) dissolved oxygen concentrations in the epilimnion can also occur due to daytime photosynthetic oxygen production superimposed over the continuous oxygen demand from biotic respiration.

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.

Amixis — Amictic lakes never circulate. They are permanently covered with ice, and are mostly restricted to the Antarctic and very high mountains.

Holomixis — In holomictic lakes, wind-driven circulation mixes the entire lake from surface to bottom. Several types of holomictic lakes have been described.

Oligomictic lakes 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.

Monomictic lakes undergo one regular period of circulation per year. Cold monomictic 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. Warm monomictic lakes circulate in the winter at or above 4°C and stratify directly during the summer. Warm momomixis 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:

$$F = (LQ/DV)\ (\rho_0/g\beta)^{1/2} \qquad (2)$$

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	$4.7x10^4$	70	$1.2x10^{-8}$	0.0026	Deep
Detroit	$1.5x10^4$	56	$3.5x10^{-8}$	0.0030	Deep
Lake Roosevelt	$2.0x10^5$	70	$5.0x10^{-7}$	0.46	Weakly Stratified
Priest Rapids*	$2.9x10^4$	18	$4.6x10^{-6}$	2.4	Completely Mixed
Wells*	$4.6x10^4$	26	$6.7x10^{-6}$	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

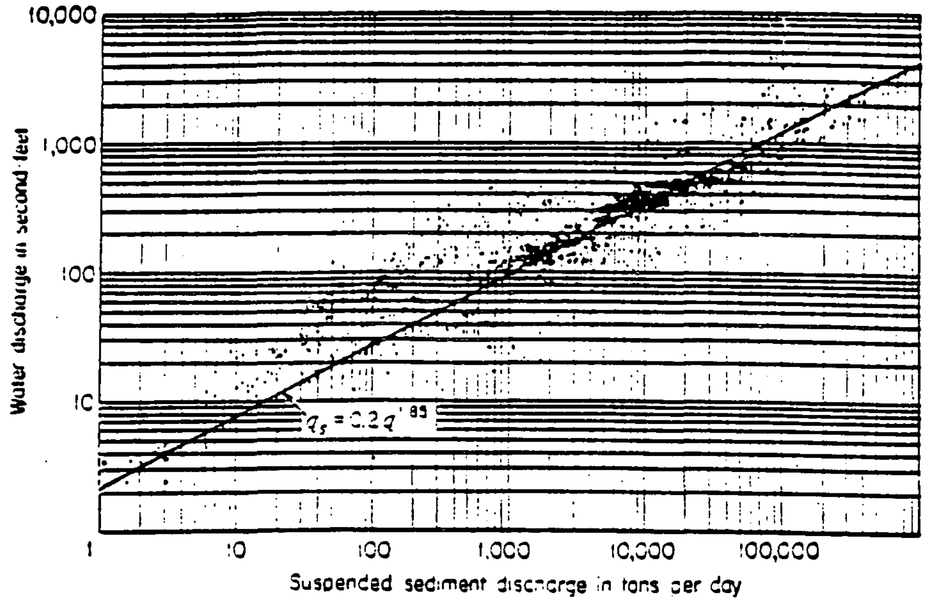


Figure II-10. Sediment-rating curve for the Powder River at Arvada, Wyoming (from Fleming, 1969)

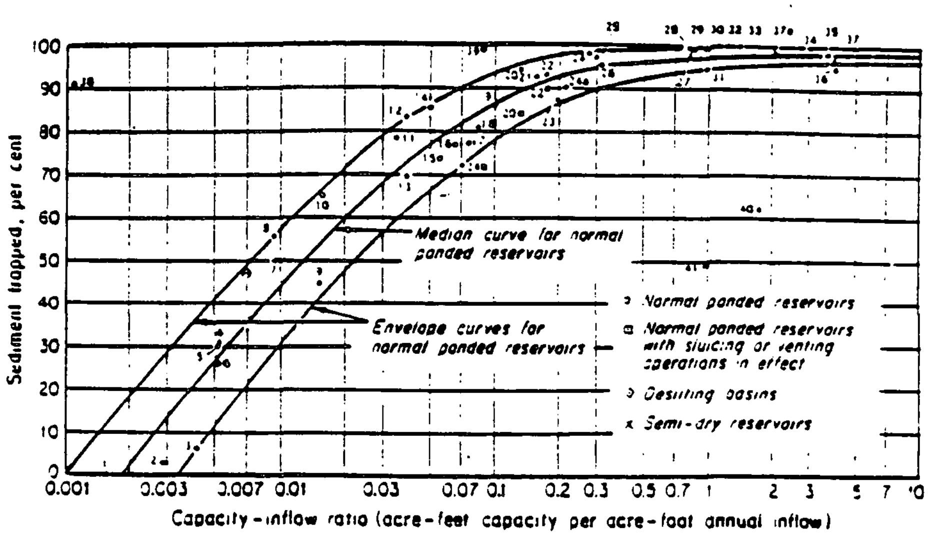


Figure II-11. Reservoir trap efficienty as a function of the capacity-inflow ratio (from Brune, 1953) higher salinities than will be found in the lake, many of the interrelationships of biology and nutrient cycling in the estuary have their counterparts in the lake.

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 reoxygenation 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:

- o Oligotrophic - the water body is low in plant nutrients, and may be well oxygenated
- o Eutrophic - the water body is rich in plant nutrients, and the hypolimnion may be deficient in DO
- o 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.

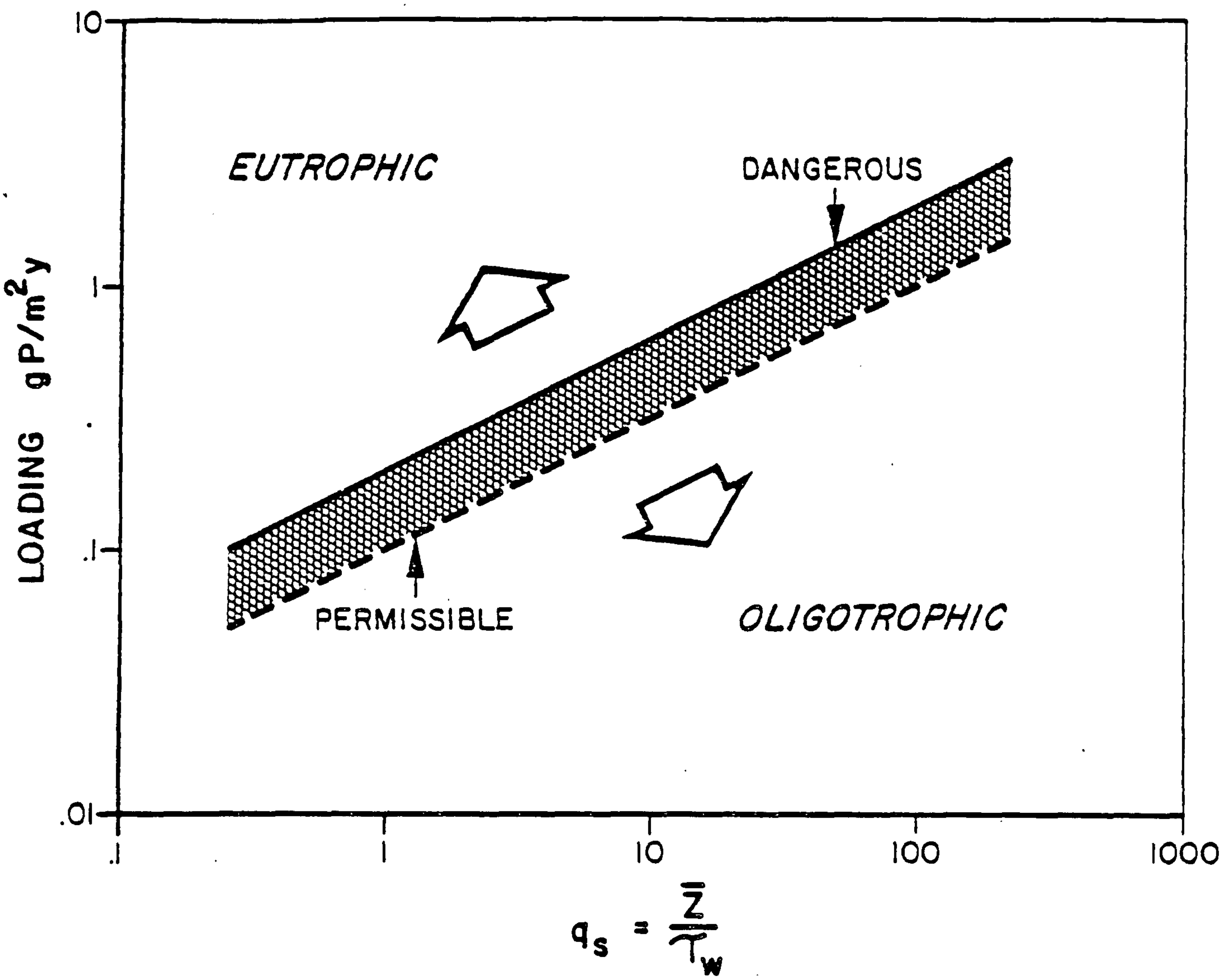


Figure II-12a. The Vollenweider Model (from Zison, et al., 1977).

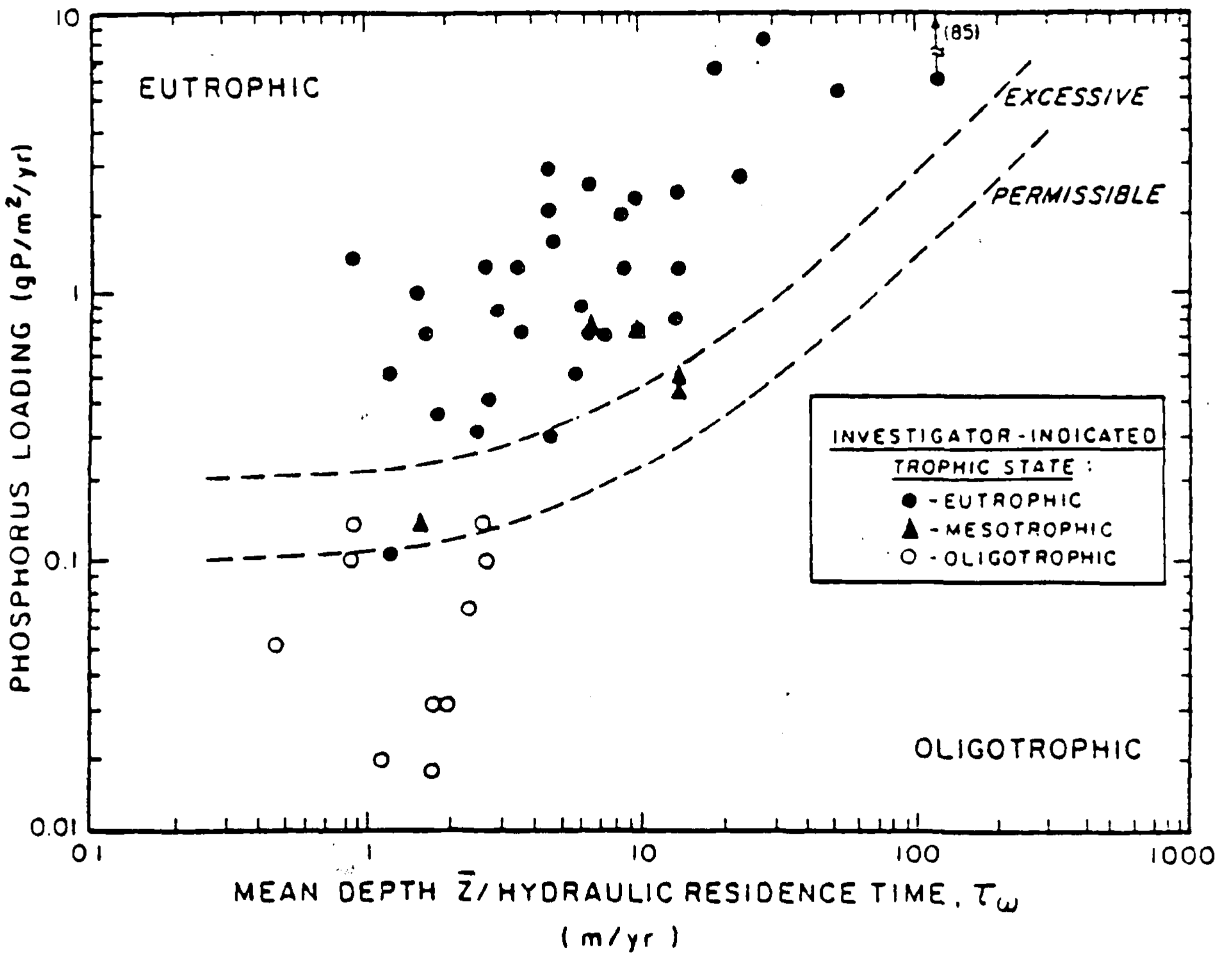


Figure II-12b. The Vollenweider-OECD Model (from Rast and Lee, 1978).

An example application of this type of approach is given by Zison, et al. (1977), where the characteristics of a reservoir are given as:

<u>Bigger Reservoir</u>

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.

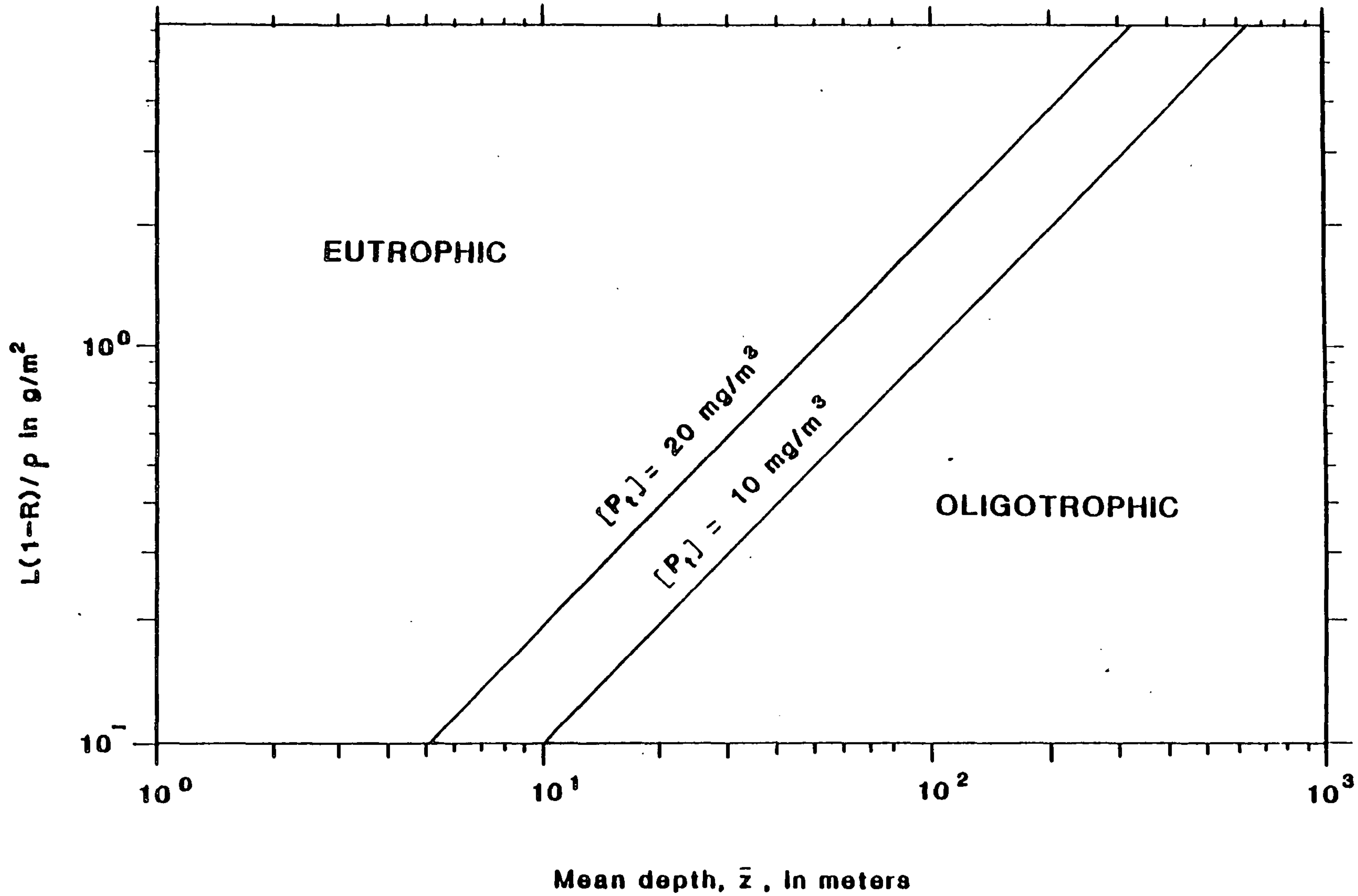


Figure II-13. The Dillon-Rigler Model (from Dillon and Rigler, 1974).

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:

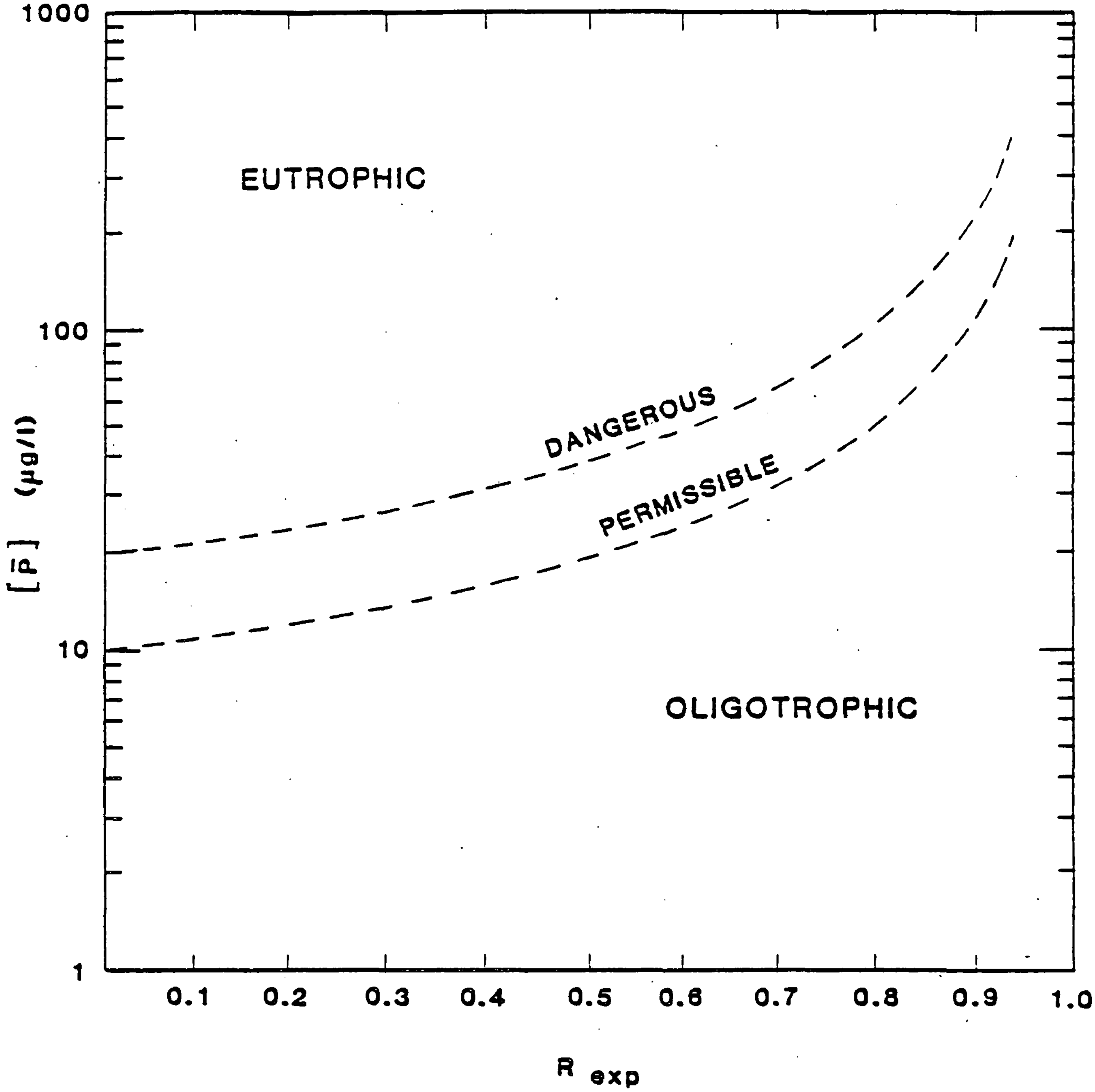


Figure II-14. The Larsen-Mercier Model (from Tapp, 1978).

$$L/(16 + \bar{z}\rho) < P_{true} < L/13.2 \quad (8)$$

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:

$$\overline{TP}_E = \frac{ETP}{\overline{1-R}} \quad \text{or} \tag{9}$$

$$\overline{TP}_M = \frac{MTP}{\overline{1-R}} \tag{10}$$

where

$\overline{TP}_E$ = the minimum mean tributary TP concentration in ug/l which will cause a lake to be eutrophic at equilibrium,

$\overline{TP}_M$ = 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,

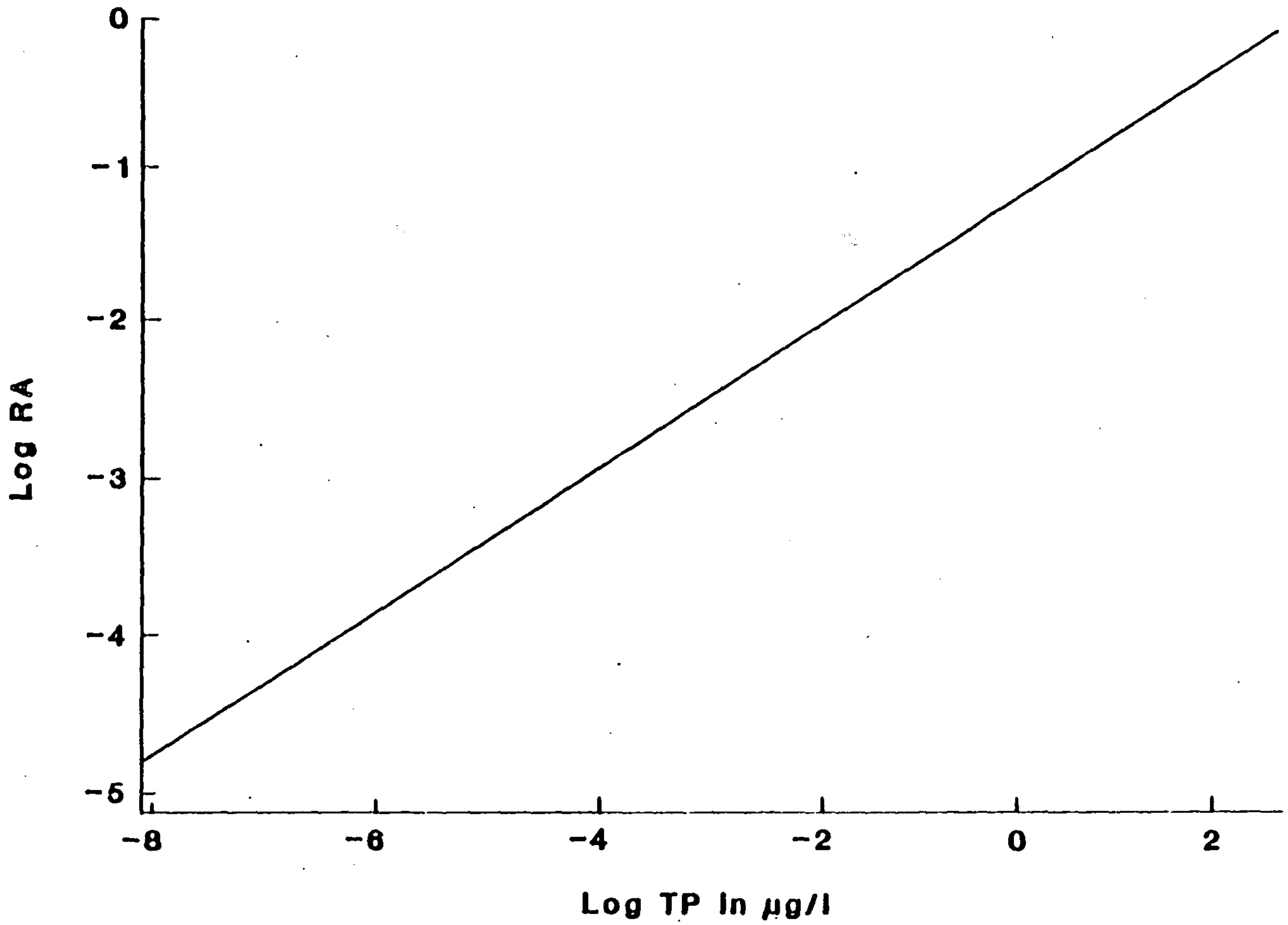


Figure II-15. The relationship between summer log RA and log TP based on Jones and Bachmann's (1976) regression equation (from Hern, et al., 1981).

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:

$$\overline{TP}_{AE} = \frac{ETP(ERA/AERA)}{1-R} \tag{11}$$

$$\overline{TP}_{AM} = \frac{MTP(MRA/AMRA)}{1-R} \tag{12}$$

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:

$$\log \text{ug/l CHLA} = -1.09 + 1.46 \log \text{ug/l TP} \tag{13}$$

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:

$$\log AERA = \left[\frac{\log ORA - A}{\log OTP - B}\right]\left[\log ETP - B\right] + A \qquad (14)$$

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:

$$\log AERA = \left[\frac{\log ORA + 4.77}{\log OTP + 8}\right]\left[9.30\right] - 4.77 \qquad (15)$$

The AMRA is determined from the following equation:

$$\log AMRA = \left[\frac{\log ORA - A}{\log OTP - B}\right]\left[\log MTP - B\right] + A \qquad (16)$$

Substituting into Equation 16:

$$\log AMRA = \left[\frac{\log ORA + 4.77}{\log OTP + 8}\right](9) - 4.77 \qquad (17)$$

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 requirements presented later in this manual in conjunction with predicted water 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.

Numerical lake circulation models have been investigated in detail by Wilbert Lick of the Case Western Reserve University. In a report for the U.S. Environmental Protection Agency, Lick (1976b) describes his work on three-dimensional models. The three-dimensional models developed by Lick include: (1) a steady-state, constant-density model; (2) a time-dependent, constant-density model; and (3) a time-dependent, variable-density model. Vertically averaged 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)

<u>Support</u>:	<u>User's Manual</u> WASP and MVP documentation is available from USEPA (Grosse Isle Laboratory). AESOP documentation is available from HydroQual. <u>Technical Assistance</u> 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.

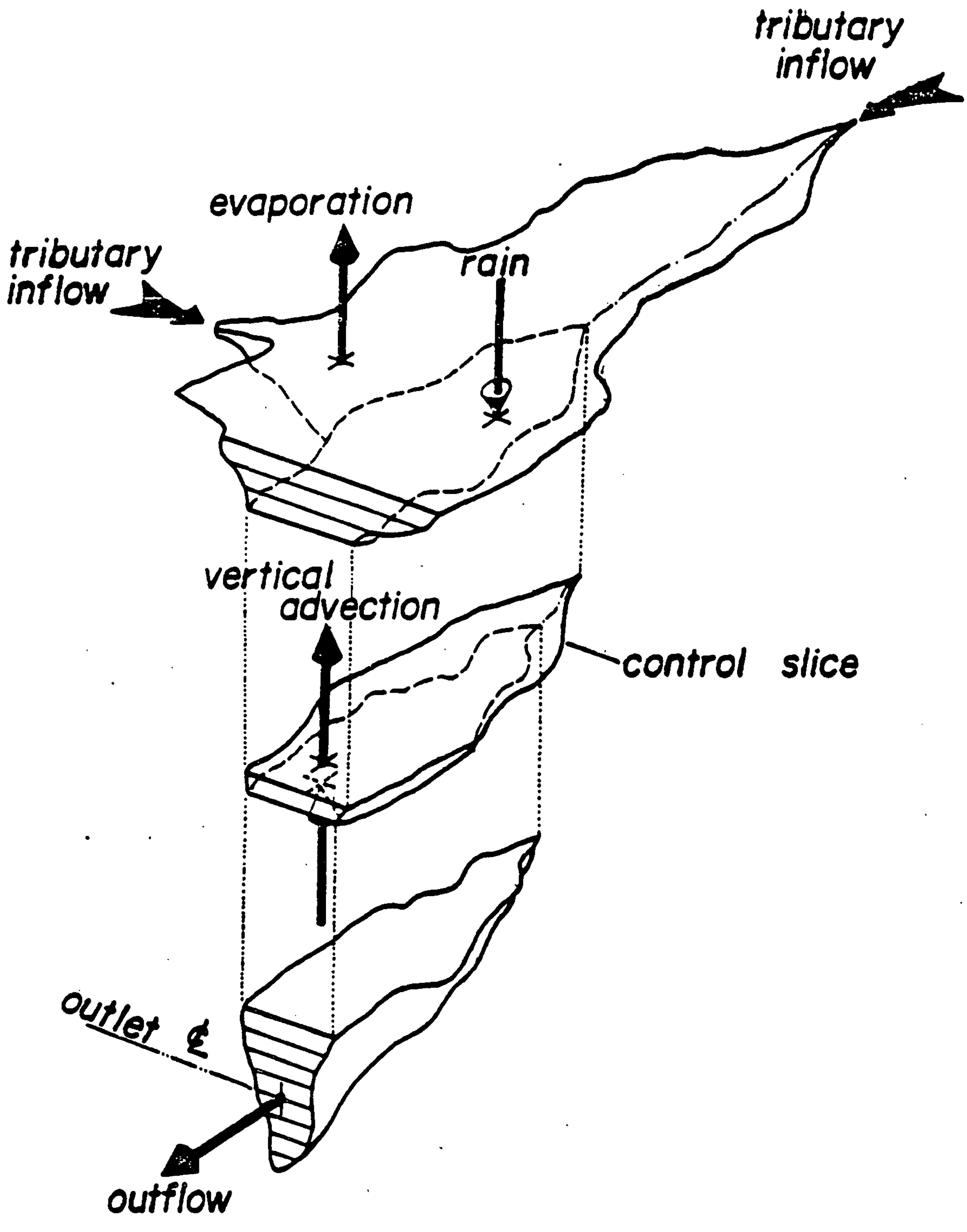


Figure II-16. Geometric Representation of a Stratified Lake
(from Gaume and Duke, 1975).

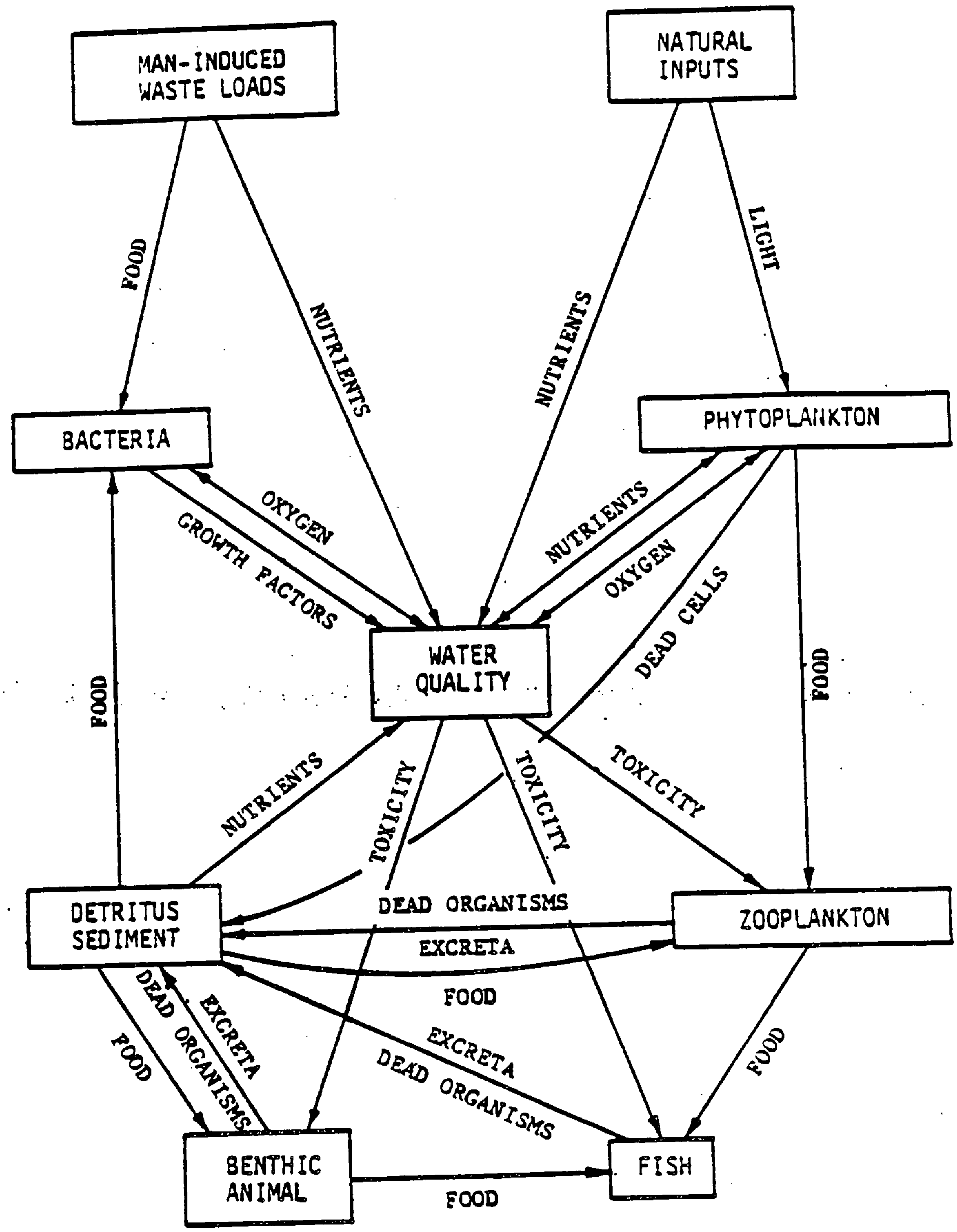


Figure II-17. Conceptual Model of an Aquatic Ecosystem (from Chen and Orlob, 1972).

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

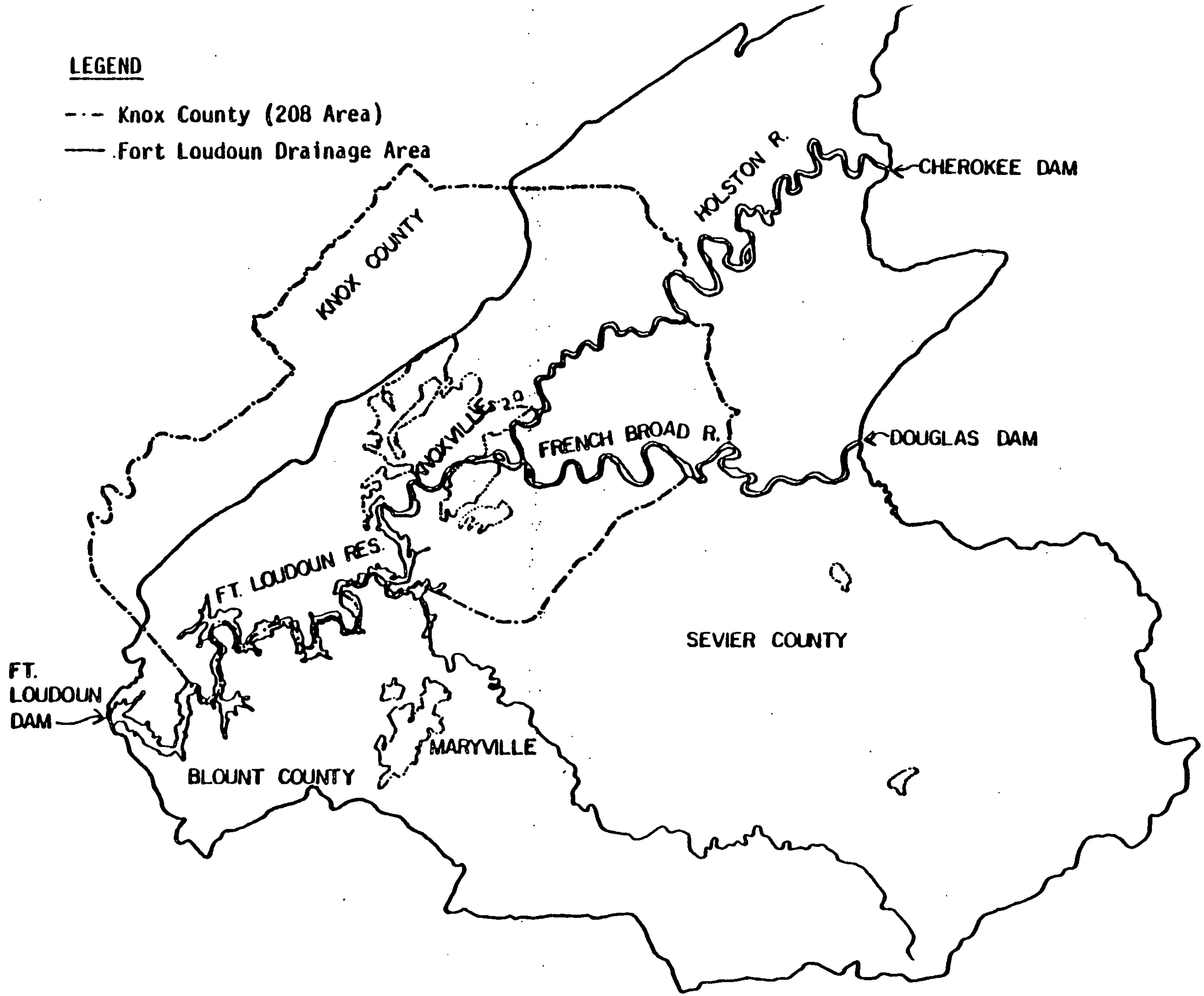


Figure II-18. 208 Study Area (from Hall et al, 1976)

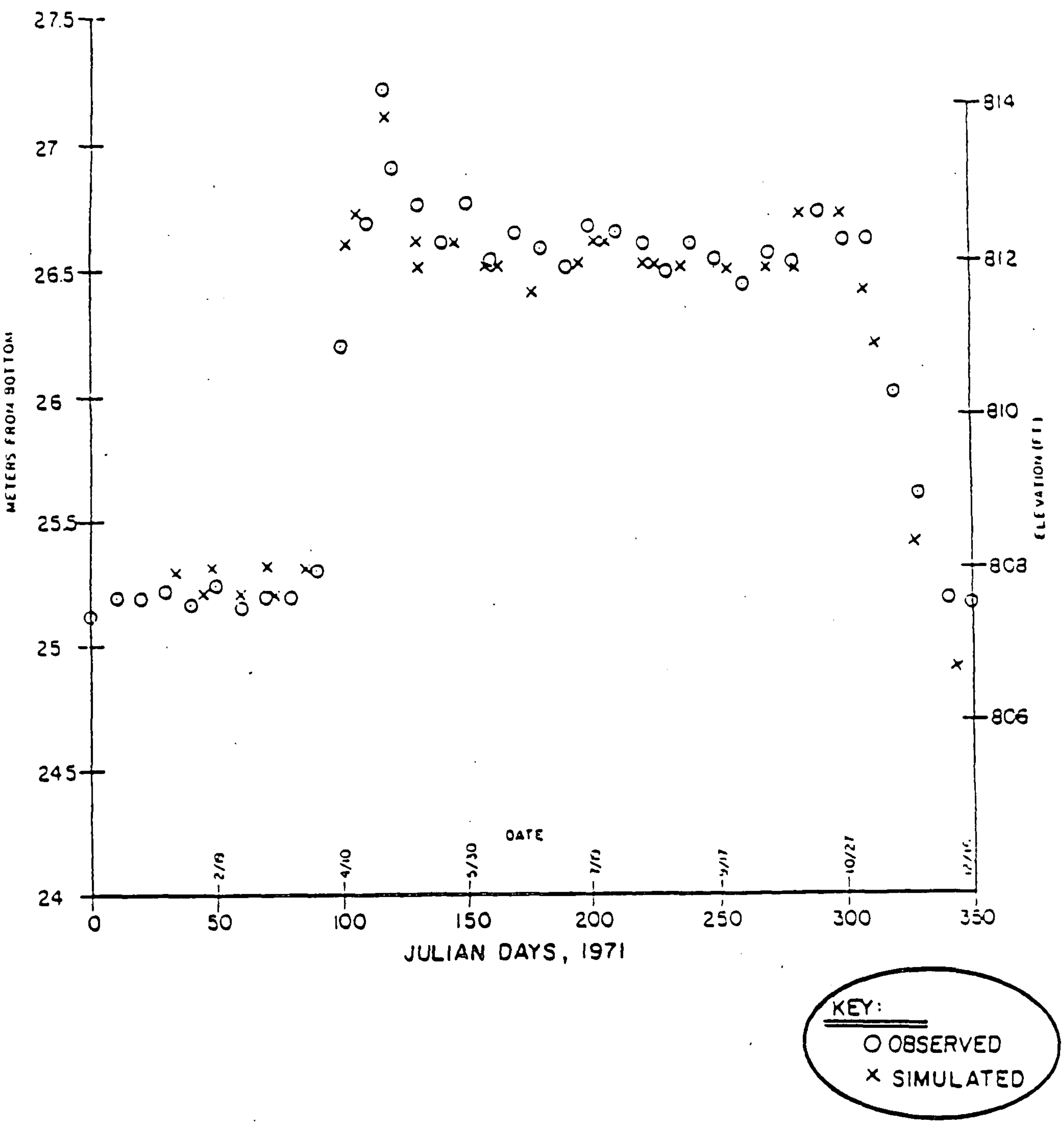


Figure II-19. Fort Loudoun Reservoir Elevations 1971 Observed vs. Simulated (from Hall et al, 1976)

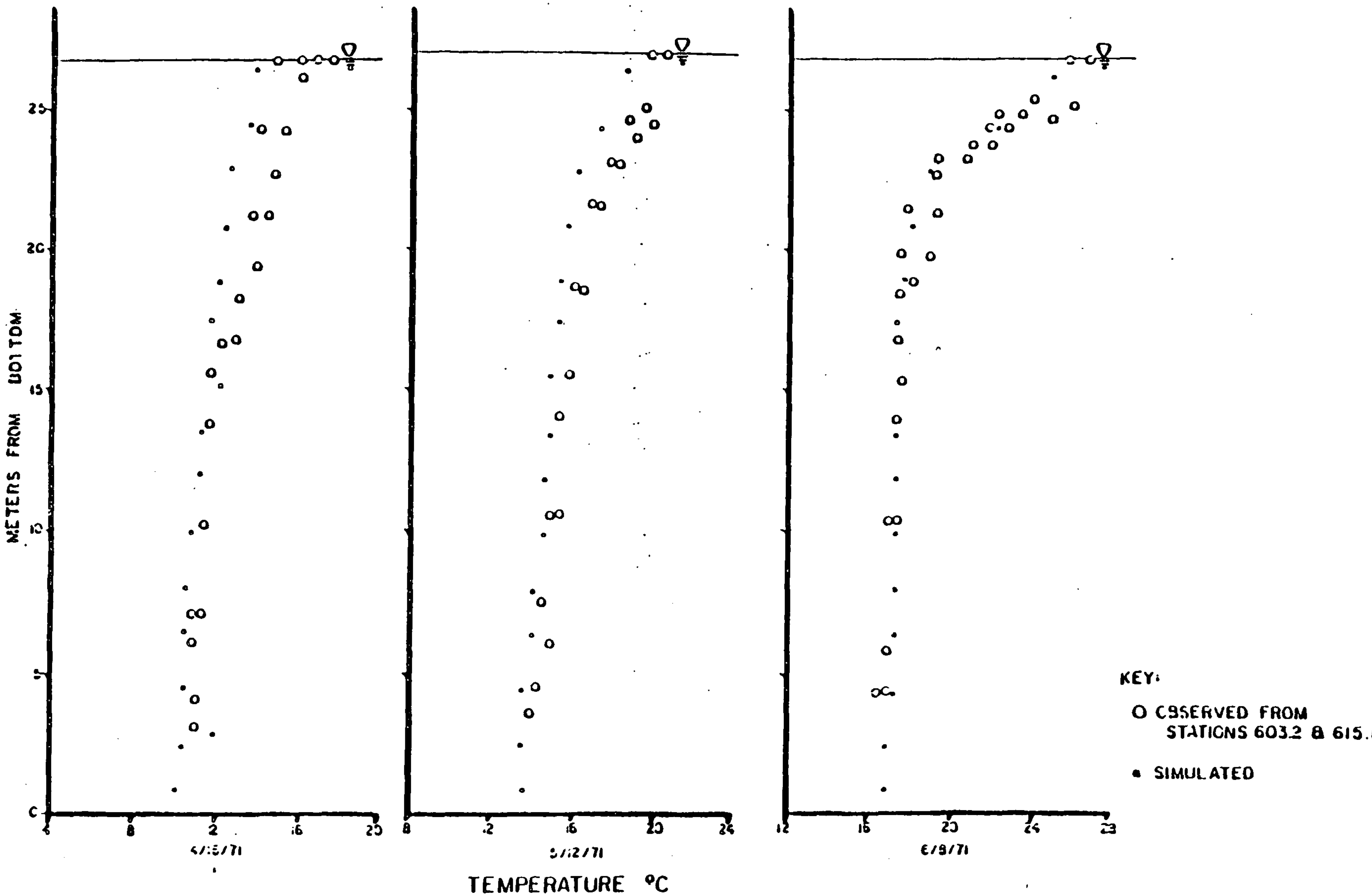


Figure II-20 Temperature Profile for Fort Loudoun (from Hall et al. 1976)

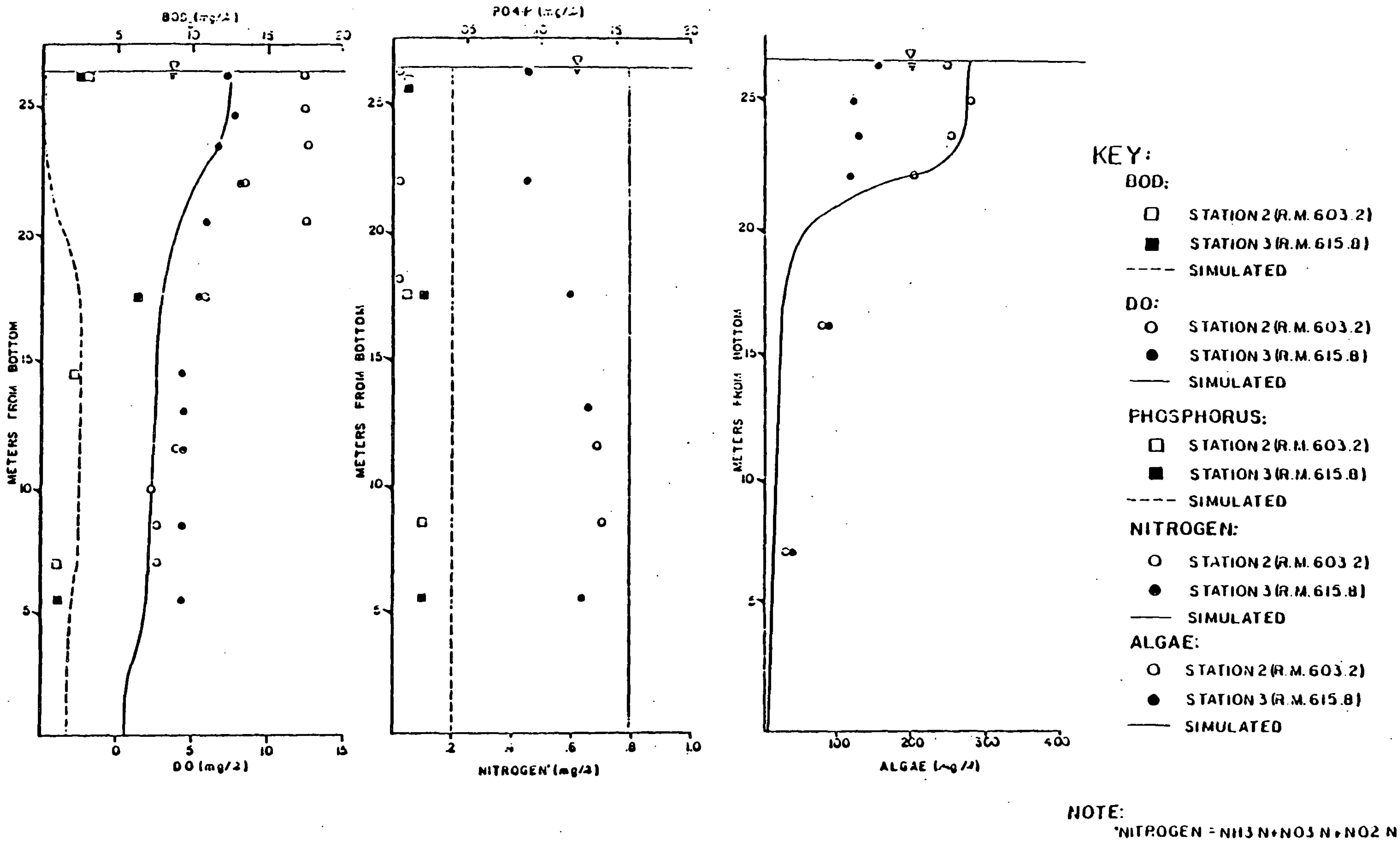


Figure II-21. DO and BOD_5, Inorganic Phosphorus and Nitrogen, and Algae September 10, 1971 Fort Loudoun (from Hall et al, 1976) organic matter in the lake deplete the oxygen supply, which in turn results in a loss of some types of fish. Excessive aquatic weed growth is also detrimental to swimming, 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.

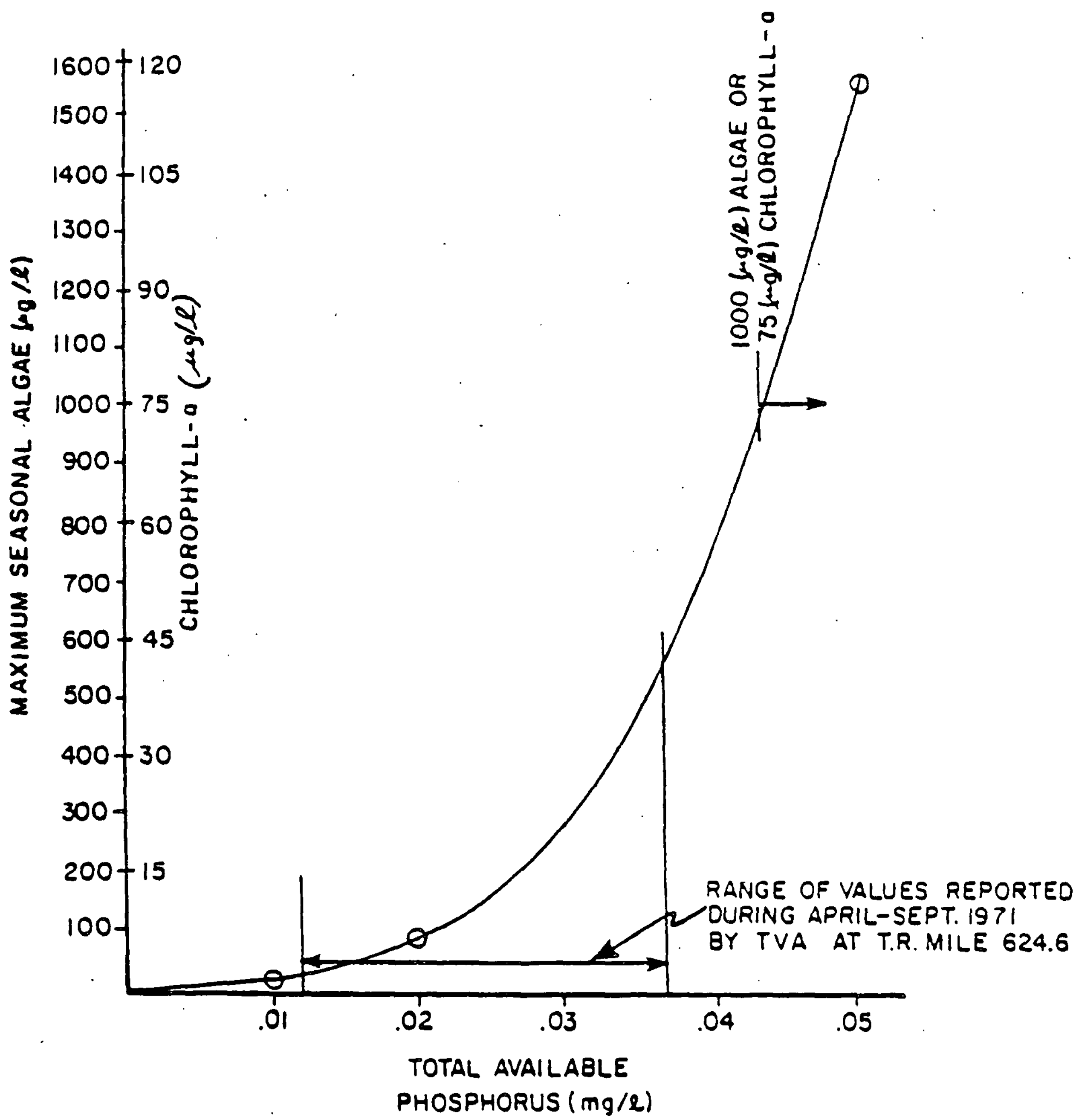


Figure II- 22 Maximum Seasonal Algae vs. Total Available Phosphorus
Lake Model Sensitivity Study - Fort Loudoun
(from Hall et al, 1976)

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)

---

*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.

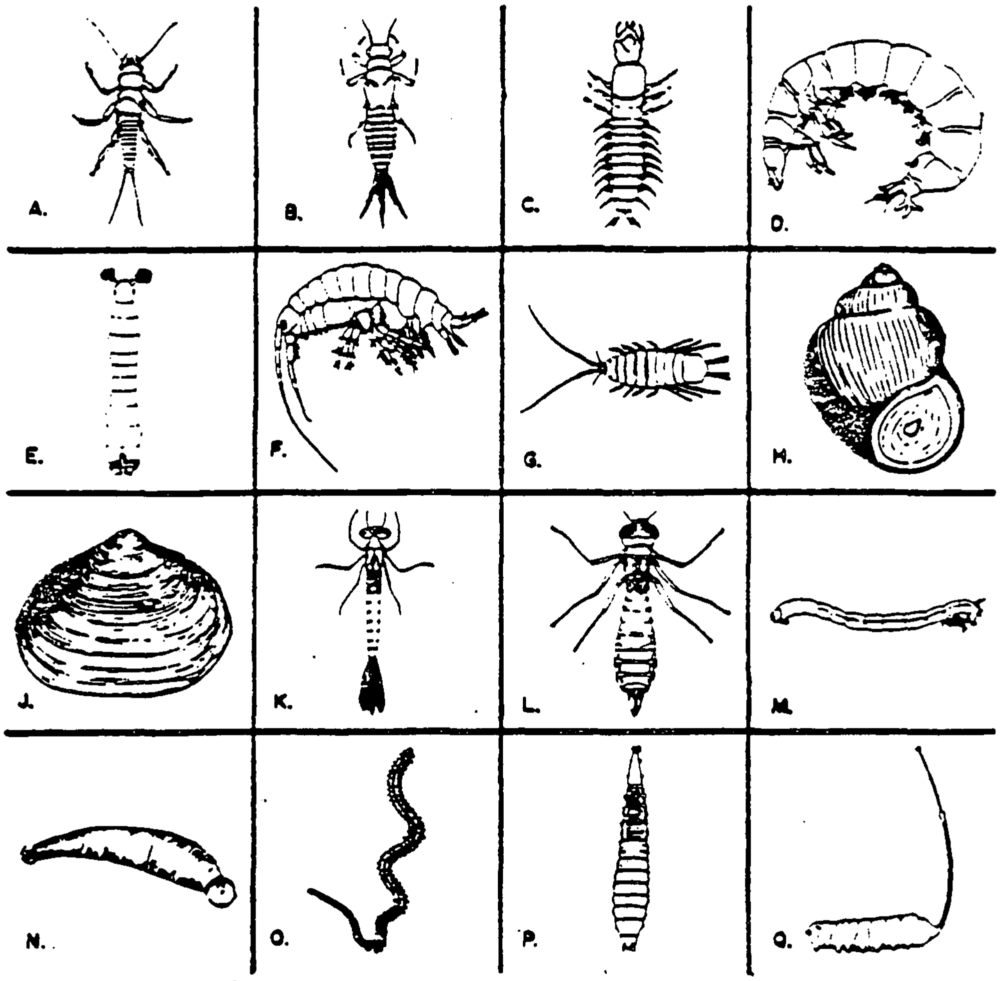
Sponges are affected by pollution although they are not usually considered indicator organisms. Of the freshwater sponges, *Ephydatia fluviatilis*, *E. muelleri*, *Heteromeyenia tubisperma*, and *Eunaius fragilis* may be found in eutrophic waters. Also, *Ephydatia robusta* can survive very low dissolved 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.

- 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);

- 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.

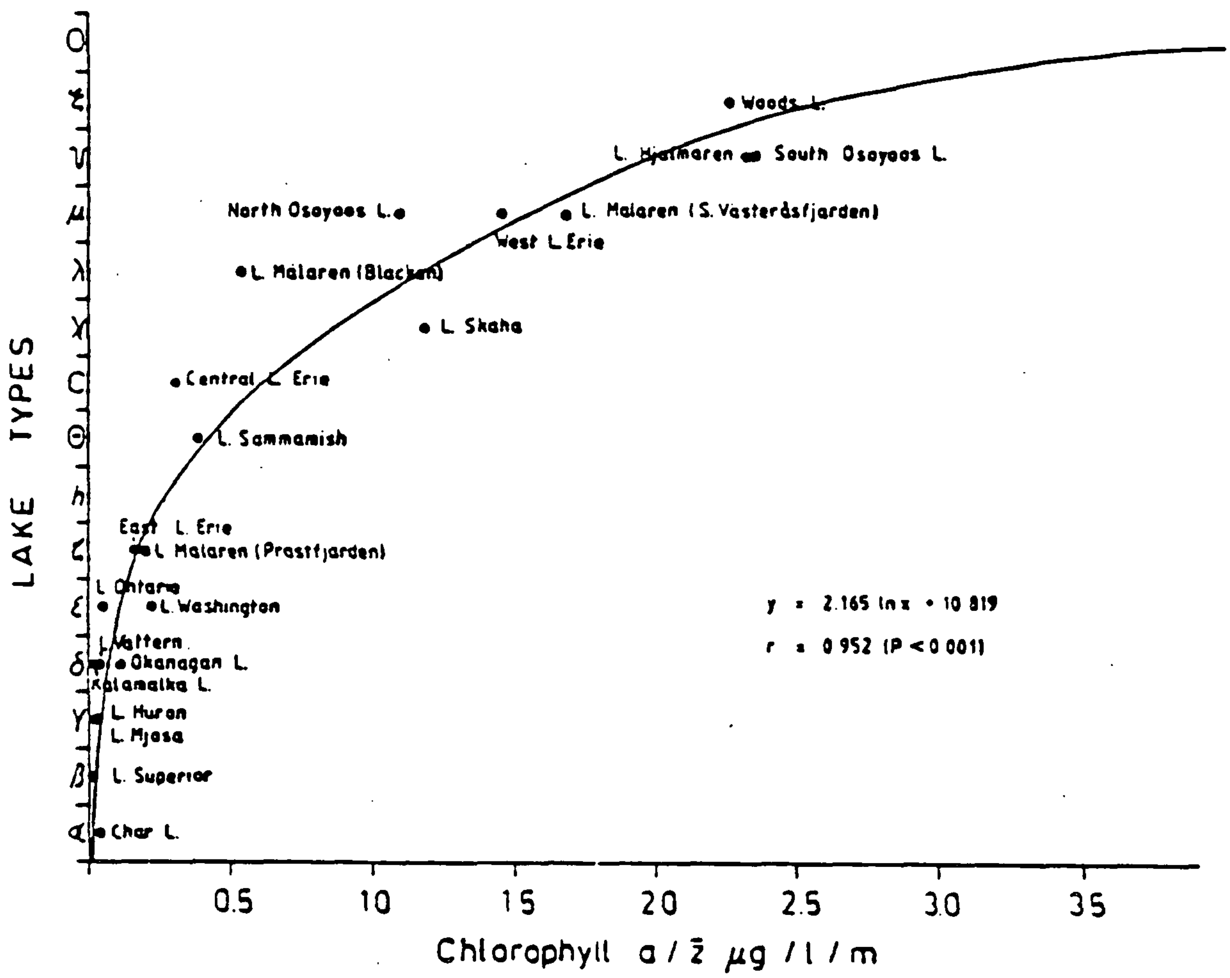


Figure III-2. Chlorophyll-a/ Mean Lake Depth in relation to 15 lake types based on Chironomid Communities (From Saether, 1979).

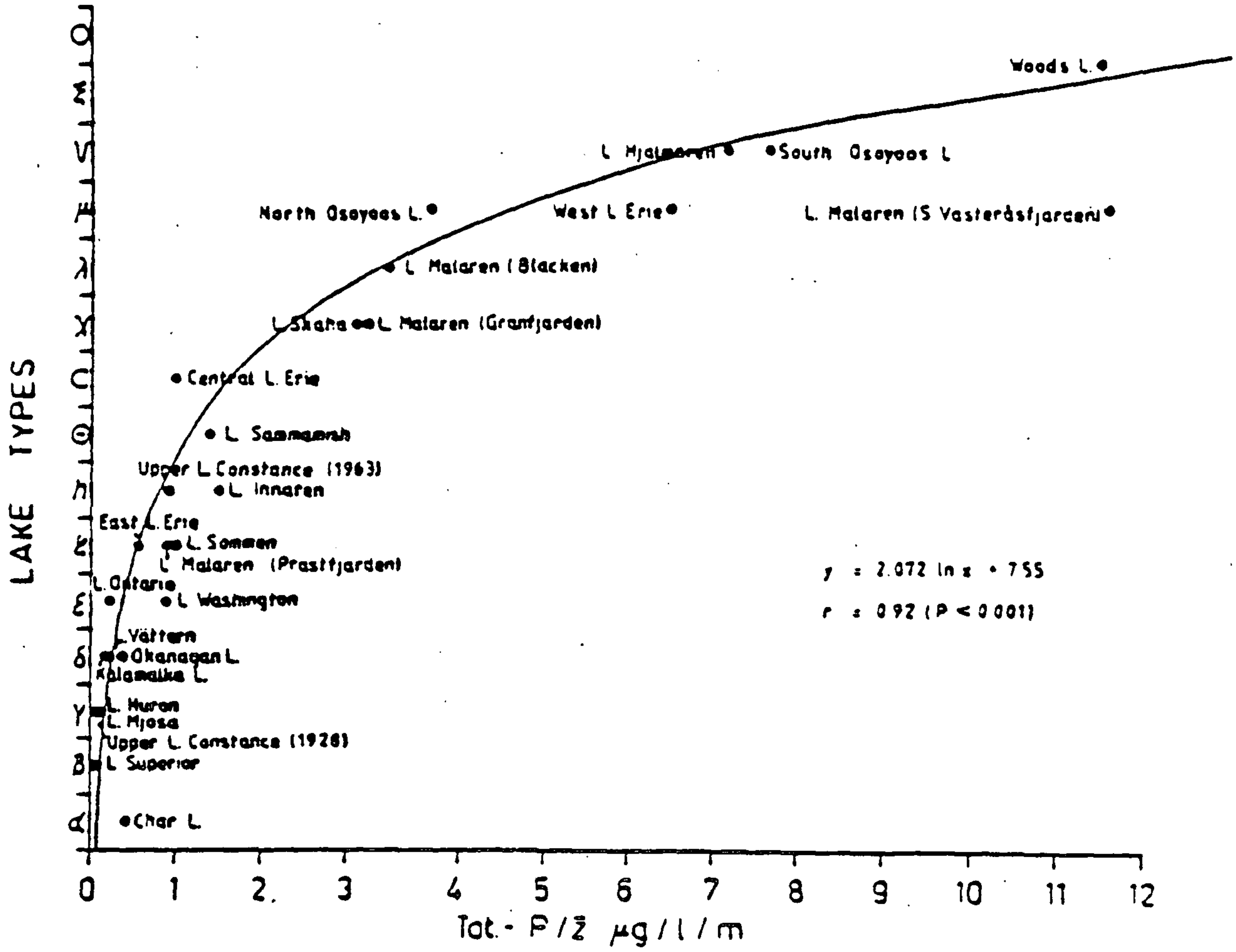


Figure III-3. Total Phosphorus/Mean Lake Depth in relation to 15 lake types based on Chironomid communities (From Saether, 1979).

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
Chironomus & 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

$$S = \frac{\Sigma\, s \cdot h}{\Sigma\, h}$$

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

$$BQI = \sum_{i=0}^{5} \frac{N_i(k_i - 1 + C_i)}{N}$$	Saether, 1979

C_i = the constancy of the respective groups within a sample

Trophic Condition Index	Howmiller and Scott, 1977 Saether, 1979

$$TCI = \frac{\Sigma N_i + 2\,\Sigma N_2}{\Sigma N_0 + \Sigma N_i + \Sigma N_2}$$

ΣN_0 = total number of oligochaete worms intolerant of eutrophic conditions (see Table C)
ΣN_i = total number of organisms characteristics of mesotrophic areas
ΣN_2 = total number belonging to species tolerant of extreme eutrophy

- Mesosaprobic: 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.

- Oligosaprobic: 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, *Heterotrissocladius subpilosus* (Kieff.); 4, *Micropsectra* spp. and *Paracladopelma* spp., specifically *P. nigritula* (Goetgh.); 3, *Phaenospectra coracina* (Zett.) and *Stictochironomus rosenschoeldi* (Zett.); 2, *Chironomus anthracinus* (Zett.); 1, *Chironomus plumosus* 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 *L. hoffmesiteri* (%). 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.

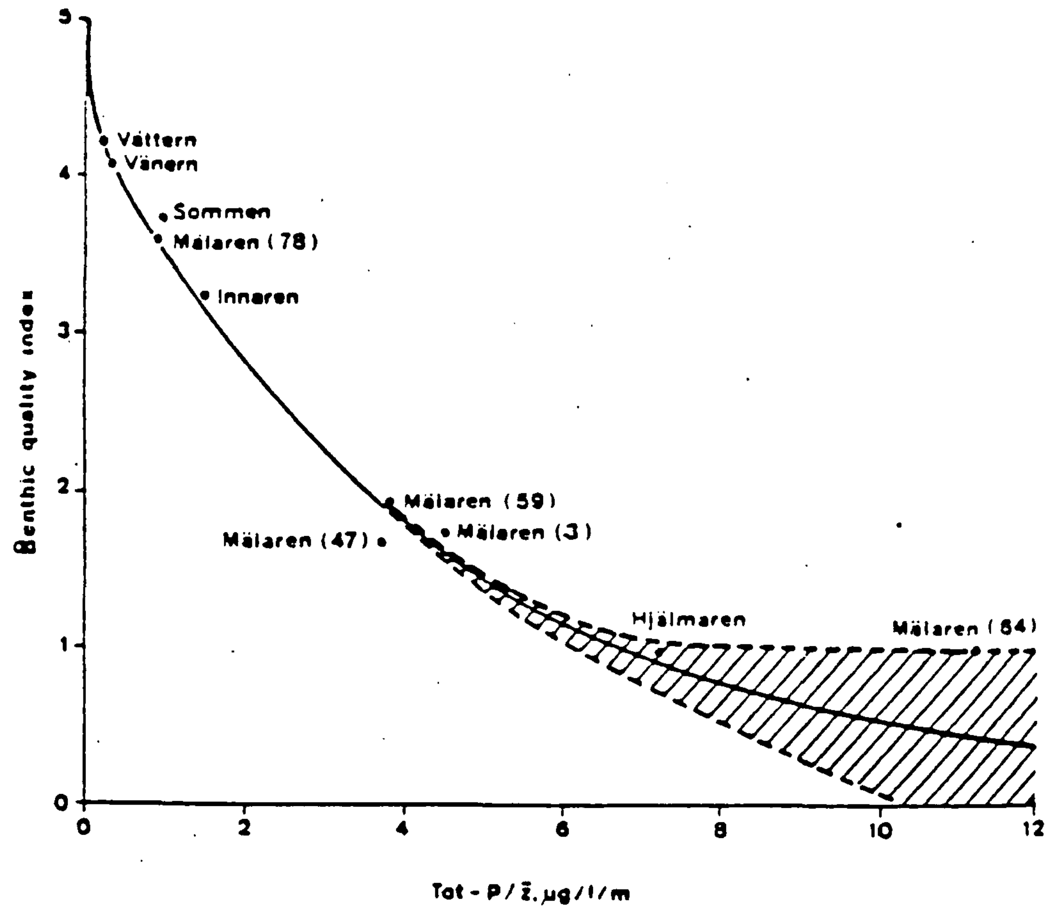


Figure III-4. Total phosphorus/mean lake depth in relation to a benthic quality index (BQI) based on indicator species of chironomids (From Wiederholm, 1980).

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.

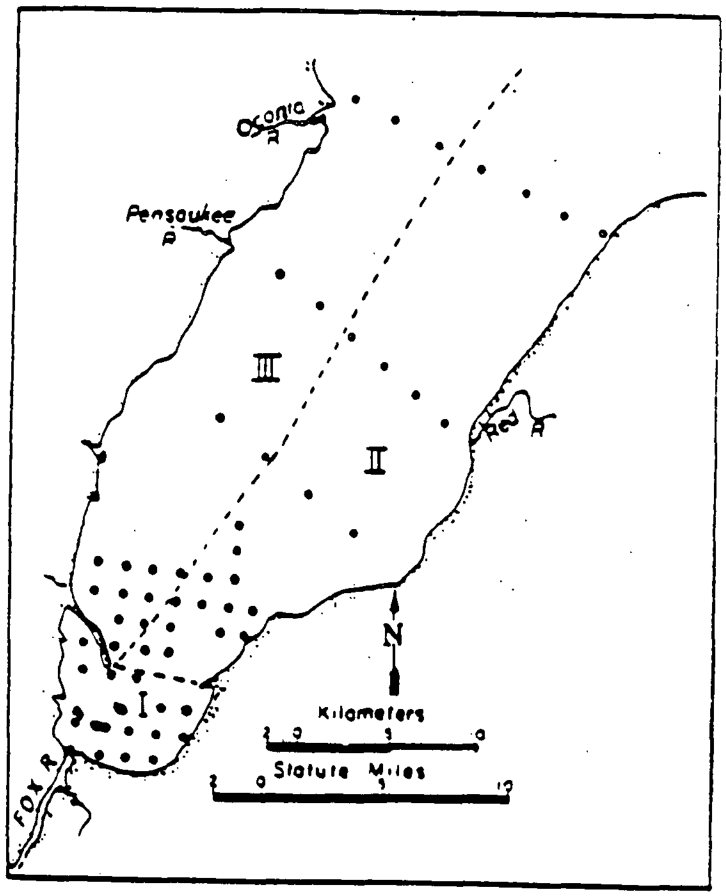


Figure III-5. Map of Lower and Middle Green Bay showing location of benthos sampling stations and areas designated I, II, and III (from Howmiller and Scott, 1977).

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, chub-suckers, 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 a 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.

Comparison

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 Technical Support Manual (U.S. EPA, 1983b).

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.

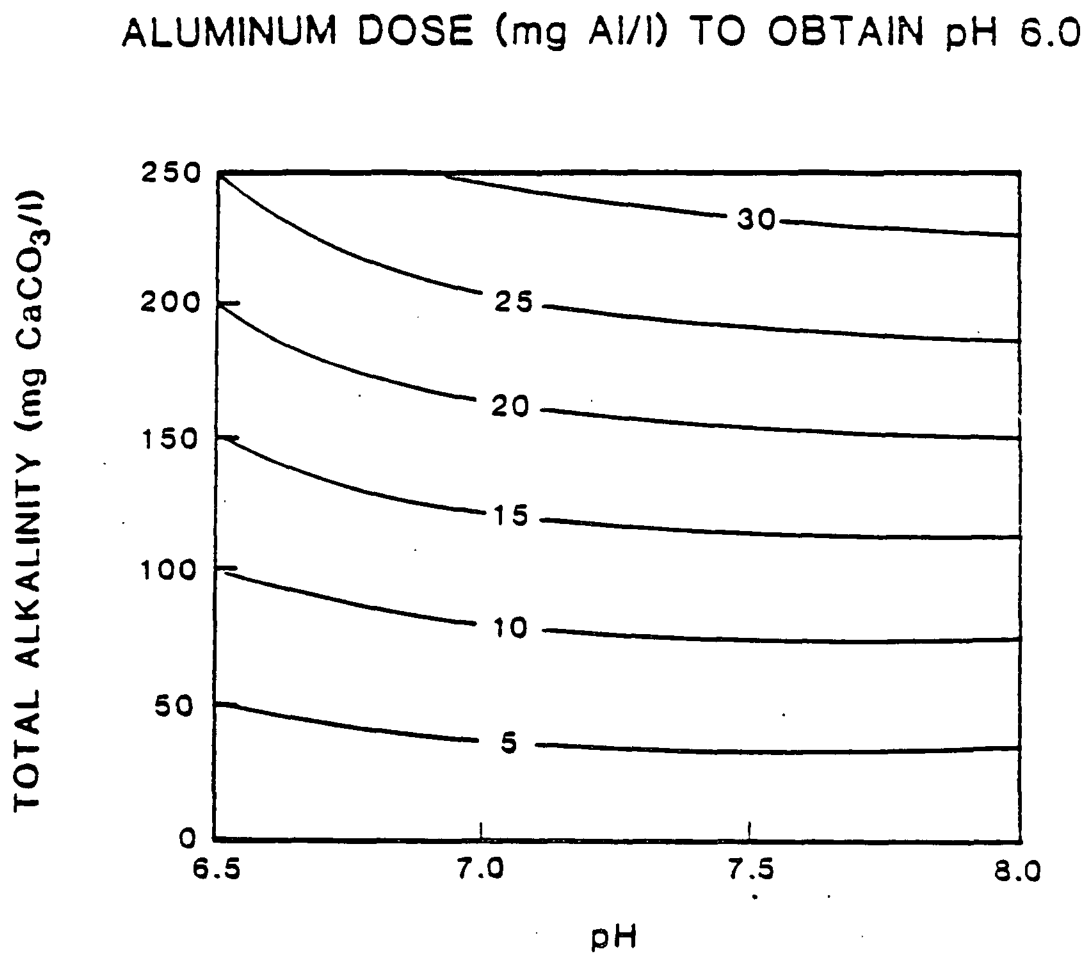


Figure IV-2. Estimated aluminum sulfate dose (mg/l) required to obtain pH 6 in treated water of varying initial alkalinity and pH (from Cooke and Kennedy, 1980).

(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.

<u>Artificial Circulation</u>. 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 manage 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 (1980*a*) 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, 1980*a*). 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 (1980*a*) compiled the following list of responses of some common nuisance aquatic macrophytes to drawdown:

- Increased: *Alternanthera philoxeroides* (alligatorweed)
  Najas flexilis (naiad)
  Potamogeton spp. (pondweed)

- Decreased: *Chara vulgaris* (muskgrass)
  Eichornia crassipes (water hyacinth)
  Nuphar spp. (water lily)

- 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, 1981*a*, 1981*b*).

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 measurement was made. The data available showed that Secchi disc depth increased from 1 m to 4 m and chlorophyll-*a* decreased over 90 percent (from 45 ug/l to 20 ug/l). Total phosphorus in the lake water declined from a summer mean of 65 ug/l to 20 ug/l (Welch, 1979, 1981*a*, 1981*b*; U.S. EPA, 1982).

Control of Nuisance Aquatic Vegetation

Management practices for the control of aquatic weeds include chemical control, mechanical control (dredging and harvesting), habitat manipulation (use of shades, dyes, bottom coverings, lake drawdown) and biological control (fish, shellfish, insects, disease, competitive plants).

Chemical Control. Aquatic weeds can be controlled by a variety of chemicals, including 2,4-D, Diquat, Endothal, Simazine, Fenac, Dichlobenil, Floridone, acrolein, and copper compounds. Combinations of Diquat and copper sulfate ($CuSO_4$) and Endothal and copper sulfate have been shown to be effective for weed control, using lower concentrations of herbicide than that required for the herbicide alone (Nichols and Shaw, 1983). Herbicides are most effective in water with low turbidity, at water temperatures of 15°C to 18°C, and on young plants. Effectiveness is also increased in waters with high calcium concentrations, and when herbicides are applied before weeds develop seeds.

Harvesting. Harvesting is commonly practiced in the Northeast, Upper Midwest, and West coastal regions to control aquatic weeds. The efficacy of harvesting depends upon the biology of the particular species. For example, more than one harvest is needed to control milfoil regrowth over the growing season. The major positive effects of harvesting are (Nichols and Shaw, 1983):

- Organic material removed by harvesting is no longer available to deplete oxygen supplies upon decay;
- Nutrients are not available for recycling upon decay of the plant; and
- Foreign material of a chemical or biological nature is not being introduced into the system.

The negative impacts include (Nichols and Shaw, 1983):

- Temporary increase in turbidity;
- Loss of animal habitat;
- Potential of plant spread by vegetative means;
- Increased growth following removal of canopy;
- Harvesting of animal material;
- Release of nutrients from cut stumps.

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:

- Decreases in acidophilic algae and mosses, with concurrent increases in diversity of planktonic algae;
- Predominance of cladocerans shifts to a predominance of copepods after neutralization;
- Reduction in benthic biomass after liming, but eventual recovery with repopulation of less acid tolerant species;

- 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 re-acidify 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:

$$MV = \text{Log TOTALP} + \text{Log CHLA} + \text{Log KJEL} - \text{Log SECCHI}$$

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	1488	.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 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
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 8-3
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 fragilis		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	
Gymnolaemata			
Ctenostomata			
Paludicellidae			
Paludicella ehrenbergi		11	
Coelenterata			
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		
Prosopora			
Lumbriculidae	14		
Hirudinea			
Rhynchobdellida			
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			
Hyalella azteca		4,2 3,9	
H. knickerbockeri	11		
Gammaridae			
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. conasaugaensis			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. litosternum		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 richardson			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	[illegible]
Psychodidae	3		
Psychoda alternata	10		
P. schizura	10		
Psychoda	9		
Telmatoscopus albipunctatus	14		
Telmatoscopus			10
Simulidae	9	10	4,3
Simulium vittatum		6,10	
S. venustrum			10
Simulium			2
Prosimulium johannseni			10
Cnephia pecuarum			10
Stratiomyiidae	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. aenaus	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 crepuseularis			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 vittgera			9
Baetidae			
Baetis vagans			9
Callibaetis floridanus	3		
Callibaetis		6	
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			
Corydalis cornutus		9	4,2,
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)

Macroinvertebrate	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
Anacyronyx 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. perexra		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			8
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			11
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
Trigonia 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	4,8		
Sphaerium notatum	8		
S. corneum		8	
S. rhomboideum		8	
S. striatinum		8	
S. s. var. corpulentum		11	
S. s. var. lilycashense		11	
S. sulcatum		8	
S. stamineum		11,8	
S. moenanum		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. idahoensis	8		
P. complanatum	11,8	11,8	
P. subtruncatum		8	
Pisidium		11	
Dresisseniidae			
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-oligotrophic
   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 Sæth.		
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 Sæth.		
Micropsectra lindebergi Säw.		x
Tanytarsus lugens Kieff.	x	x
Heterotrissocladius sp. *A* near *subpilosus*		
Heterotrissocladius sp. *B* near *maeaeri*		
Protanypus ramosus Sæth.		
Micropsectra contracta Reiss	x	
Micropsectra insignilobus Kieff.		x
Paracladopelma galaptera (Town.)		
Paracladopelma nigritula (Goetgh.)	x	x
Monodiamesa tuberculata Sæth.		
Macropelopia fehlmanni (Kieff.)	x	
Tanytarsus bathophilus Kieff.	x	x
Protanypus morio Zett.	x	x
Heterotrissocladius changi Sæth.		
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* Sæth.		
Monodiamesa bathyphila (Kieff.)	x	x
Stictochironomus rosenschoeldi (Zett)	x	x
Phaenopsectra coracina (Zett.)	x	x
Tanytarsus n. sp. *lestagei* - aggl.		
Monodiamesa depectinata Sæth.		
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

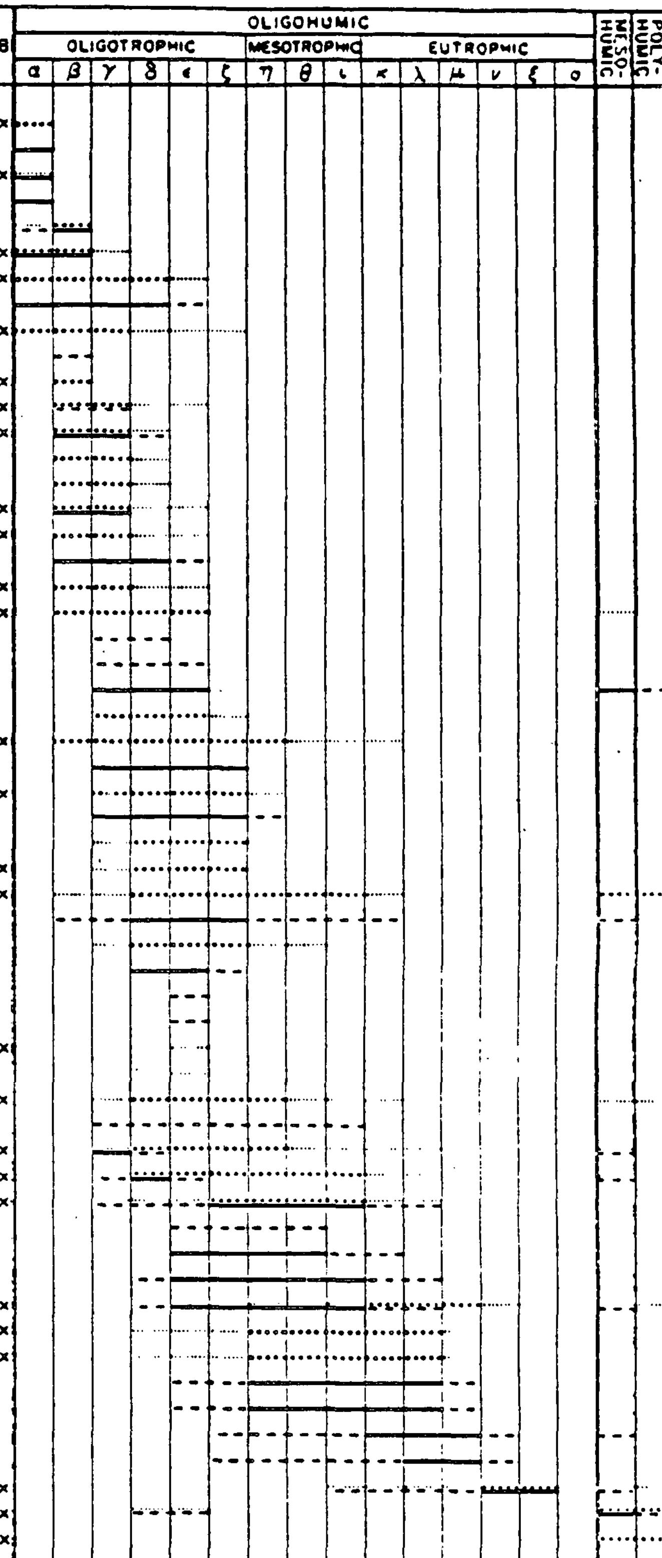

TABLE D-2
CHARACTERISTIC SUBLITTORAL AND LITTORAL CHIRONOMID HABITATS IN NEARACRTIC AND PALEARCTIC LAKES.

OLIGOHUMIC: OLIGOTROPHIC, MESOTROPHIC, EUTROPHIC — α β γ δ ε ζ η θ ι κ λ μ ν ξ ο; MESO-HUMIC; POLY-HUMIC

SPECIES	A	B
Heterotrissocladius subpilosus (Kieff.)		x
Heterotrissocladius oliveri Sæth.		
Hydrobaenus fusistylus (Goetgh.)		
Zalutschia trigonacies Sæth.		
Abiskomyia virga 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 exmani 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 absepta (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 natvigi (Goetgh.)		x
Heterotanytarsus apicalis (Kieff.)	x	x
Paratanytarsus penicillatus Goetgh.		x
Stempellina n. sp. near *almi* Brund.		
Nanocladius (N.) anderseni 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 Sæth.		
Saetheria tylus (Town.)		
Heterotrissocladius latilaminus Sæth.		
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 Sæth.		
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 (Po.) 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.) platypes Edw.	x	x
Zalutschia mucronata (Brund.)		x