Chapter 3

EVALUATING LAKE AND RESERVOIR WATER QUALITY

Water quality monitoring is ideally a systematically organized activity with the goal of obtaining information on the quality of water in particular localities or regions. Water quality monitoring has often been characterized by the "data-rich, but information-poor syndrome" (Ward et al., 1986). To avoid such constraints, this chapter concentrates on deriving information from the data collected, rather than the collection process itself. There are other books that deal with this latter topic (e.g., Bartram and Ballance, 1996; Straškraba and Tundisi). The use of data represents a good basis for obtaining needed information. This is far from guaranteed however, since collecting the data is only the first step. The next, and most important, step is to derive the needed information from the data. If data are inadequately collected, it may not be possible to derive accurate information, even from extensive data sets. To obtain relevant information, the goal of the monitoring effort must first be clarified, and the monitoring needs defined.

Monitoring must be treated as a system of activities, not just as one. It is necessary to distinguish several steps, which must be strictly coordinated to obtain maximum information with the minimum possible effort. These steps are as follows:

(1) *Monitoring goals.* What conclusions are expected to be drawn from a monitoring program depends on human perceptions of water quality (Section 3.1). The most common goals are (i) to judge the adequacy of the water for the intended use(s); (ii) to identify causative agents for management actions; (iii) to document trends over time; and (iv) to anticipate adverse impacts to waterbodies, in order to prevent them and to save the waterbody's ecological integrity. The last goal is the broadest one, and can be expressed as maintaining environmental quality for human use and protection of all aquatic life. In a more restricted sense, it can be directed to activities such as protecting particular types of fisheries.

(2) *Relevant indicators.* Which water quality and environmental indicators are relevant for achieving the monitoring goals? Which classification system should be used? What degree of sensitivity is necessary for each water quality parameter of concern? It is pointless, for example, to use sensitive, expensive methods for measuring variables which vary within minutes, or from one surface station in a lake to stations a few meters apart. The variables must be sensitive, and their range suitable for the application, reproducible and precise. Decisive for selecting the variables is the monitoring goal, the legislature and the evaluation system to be used (Section 3.2).

(3) *Sampling schedule.* Which localities and depths must be sampled and at what sampling frequency? If the goal is to obtain information on the water used for power generation or drinking water, a sampling locality close to the outlet or uptake place is important

if sources of pollution near the inflow and near-shore localities along the waterbody must be detected (Section 3.3.2).

(4) *Type of sampling.* Will manual sampling be appropriate, or should automatic monitoring or remote sensing be used? What sampling gear should be used? For some determinations, only a small quantity of water is needed. However, others may require large quantities for reliable measurement. The samples should not be contaminated from other sampling points and depths. If organisms are being sampled, another important consideration is whether or not the organisms are capable of avoiding the sampling gear (see Section 3.3).

(5) *Sample elaboration.* How will the samples be stored and elaborated? Samples must either be preserved or protected against heating or cooling, in order that major changes not occur for some variables (Section 3.4.1).

(6) *Information extraction.* What methods will be used to extract information and draw conclusions? If comparisons are to be made with other data, are they obtained using comparable methods? If models are to be used, are all the necessary input data available or obtainable (Section 3.5)?

A very important factor in developing a monitoring program is its cost. A methodology for estimating the costs of monitoring programs was previously presented by Cale and McKown (1986). Millard and Lettenmaier (1986) also discuss a formalized mathematical procedure for the optimal design of sampling programs in two forms, including (i) minimization of sampling costs for a given detection of changes, and (ii) maximization of detection for a given cost.

The goal of monitoring should be to avoid the data-rich, but information-poor syndrome by careful consideration of the above-noted steps. After completion of monitoring efforts, there are only very limited and time-consuming possibilities that exist for attempting to correct errors made in preparation of the full monitoring program.

The general aspects of water quality monitoring are treated in several recent volumes of a series of books produced under the auspices of the UNEP/WHO Global Environmental Monitoring System (GEMS/Water). One is a practical guide for the design and implementation of freshwater quality studies and monitoring programs (Bartram and Ballance, 1996), two are devoted to the use of biota, sediments and water in environmental monitoring (Chapman, 1992, 1996), and the most recent to monitoring of bathing water (Bartram and Rees, 2000). Accordingly, this chapter focuses on the means of deriving information from completed monitoring programs; the other topic to be mentioned is related to retrieval of the information. More details are to be found in the above-noted book series. A computerized expert system to support environmental sampling may also prove helpful in such efforts (e.g., Olivero and Bottrel, 1990).

3.1 HUMAN PERCEPTIONS OF WATER QUALITY

The general public perceptions of lakes and reservoirs was discussed in Section 1.2.1. This chapter discusses perceptions of water quality in more detail. Humans perceive lake and reservoir quality on the basis of its intended uses. The perceptions will differ, depending on whether or not an individual is swimming in a lake, boating or fishing in it, or if a lake is being used as a drinking water supply. The perceptions can arise solely on the basis of human senses or, alternatively, from specific technical determinations.

Vision, smell and taste are the natural senses that humans primarily use to form perceptions about water quality. For swimming, water clarity is usually the most important variable, and is dictated by water color (blue is preferred over brown), by mineral turbidity, and by the quantity of algae present in the water. The presence of blue–green algae (cyanobacteria) is not only unpleasant, as it can accumulate in swimming suits, but it can also cause skin and eye irritation in sensitive people if some toxic strains are present in the waterbody. When drinking water is taken from waterbodies with large quantities of cyanobacteria, the health consequences can ultimately be very serious (Section 2.2.2). The presence of dense, submerged vegetation can reach close to the water surface, preventing easy and pleasant swimming and fishing. Boating is a far less sensitive activity in regard to what is present in a waterbody, being instead more directly related to the beauty of the surrounding scenery. For the purpose of drinking water, visibly clean, nonturbid water, with no tastes and odors, is preferred. Even hygienically safe water will not necessarily be considered good for drinking if it does not possess the previously mentioned qualities. The only chemical water characteristic that humans can readily recognize (when extreme) is water hardness. The latter is obvious from taste, but is particularly obvious when the water is used to prepare drinks and food, or for washing. Tea is ugly when prepared with very hard water. For preparing coffee, however, hard water is often preferred. The ability to use soap in these two extremes of water hardness differs considerably.

A number of water quality characteristics are not detectable via our senses. Humans cannot sense the quantity of nutrients, organic matter, heavy metals, toxics and other materials in water in a direct way unless they are reflected in some way in the lake biota (e.g., nutrients via algal blooms, high concentrations of toxics via the absence of fish or if they create human health problems).

The most generally recognized variable indicative of the general status of lake water is water transparency, which can be easily measured with a Secchi disk. The systematic and continuous monitoring of this variable by volunteers, for example, can give very indicative data of lake and reservoir water quality conditions over the long term.

3.2 INDICATORS AND CLASSIFICATION OF WATER QUALITY

The indicated geographic differences of lakes discussed in Section 1.3.2 relate to both unpolluted and polluted lakes. However, the distinctions above do not specifically relate to pollution. This section deals with water quality indicators and the corresponding water quality classifications. The primary problem is that natural conditions differ in different regions, and accurate water quality classification must consider these differences. This is why different water quality classification systems have been developed in different countries. In some instances, they differ only in the value ranges of the same criteria used to distinguish different classes. A review of the ecological classification of surface waters, with an emphasis on lakes, is given by Premazzi and Chiaudani (1992). Because it is not possible to enumerate all of them, this chapter will highlight representative examples.

Water quality is estimated on the basis of different criteria related to water physics, chemistry and biology. The physical and chemical indicators are subject to significant short-term and spatial variability. The criteria related to the structure of biological communities, however, are more conservative. Because the systems of water quality indicators used for flowing water usually are not very useful for lakes, and the tradition of using water quality indicators is different for these two types of water systems, lakes and reservoirs are usually classified on the basis of chemical composition and trophic state indicators of a biological nature, and to the nutrient concentrations causing their respective trophy (Busch and Sly, 1992).

Two basic approaches to lake and reservoir water quality evaluation are possible, namely:

- Evaluation of individual criteria,
- Use of combined, complex, multimetric, multicriteria systems.

The advantage of the first approach is that maximum information can be extracted from the data collected, and specific management conclusions can usually be derived, including the specific causes of the state of the waterbody and what management options can be taken to solve the issues at hand. The second approach is more useful for general characterization of the overall state of the waters in a country or region, for assessment of the development of the situation over time, or for general characterization of the possible use of the waterbodies. Its usefulness for making specific management decisions, however, is restricted.

Only a few of the complex lake classification systems are mentioned here as possibilities, rather than suggesting which one should be followed in a specific case. In fact, it is almost useless to use a classification system outside the geographical locality and character of the waterbody from which the system was originally elaborated. The systems are based largely on statistical elaboration of extensive sets of measurements, another feature that makes the results regionally specific. The systems used in the United States, for example, explicitly relate the water quality values to baseline standards characterizing the "pristine" local conditions.

This book stresses the use of classification schemes for water quality management purposes, rather than for evaluation based on individual variables. The reason is twofold: first it provides a sound basis for specific management decisions and, second, it is easier to consider geographical differences if the latter are important.

3.2.1 Water Quality Consequences of Lake Morphometry

The dominant effect of lake morphometry is related to lake depth, with a distinction made between shallow and deep lakes. More subtle effects occur because of the depth distribution of water layer areas and volumes at different depths.

Common sense dictates establishing an absolute depth limit for waterbodies to be considered shallow. From the perspective of water quality, however, the absolute depth is not a good depth classification criterion, primarily because it is not decisive for the development of anoxic conditions near the lake bottom as a major threat to water quality. Hydrologically, deep waterbodies are distinguished as those which can thermally or chemically stratify for longer periods, whereas shallow waterbodies do not stratify, or only do so intermittently for a few days at a time, during particularly hot and calm weather conditions. The wind fetch is a dominant factor in the development of stratification in lakes. Intensive wind fetch mixes lakes and reservoirs to greater depths. The wind fetch depends mainly on two variables; namely, the wind speed in the area and the distance it travels over the lake surface. The wind gains speed when moving over smooth surfaces, such as a lake. In the same area and wind exposure, a waterbody with a larger surface area will be mixed to a deeper depth than a waterbody with a smaller area, or a lake less sheltered from wind effects. It is generally the ratio of the waterbody surface area to its depth that dictates whether a waterbody is shallow and, hence, fully mixed by wind, or is deep and stratified, thereby only mixed to a certain depth. A waterbody with an area of only a few hundreds of square meters can be stratified and, therefore, hydrologically deep with a maximum depth of only 3–4 meters. In contrast, a large waterbody with a depth of 20 meters can still be hydrologically shallow because it can be completely mixed on a daily basis. Gorham and Boyce and Hanna provide more detailed estimates of the mixing depth. Once the mixing depth (h_{mix}) is known, an estimate can be made from the area curve of the waterbody (the curve expressing the water areas at particular depths), the surface area of the lake (A_0) and the surface area at the mixing depth (A_h). A lake will be considered shallow if $A_h < 1/2A_0$, and deep if $A_h > 1/2A_0$.

Shallow waterbodies are often characterized by extensive areas covered by emergent and submerged vegetation, which may inhibit or prevent the utilization and management of the water resources. Three types of vegetation can be distinguished, including (i) *submerged*—rooted on the bottom of the lake, but not reaching out the surface; (ii) *floating*—not rooted on the bottom, but floating freely on the water surface, and (iii) *emergent*—rooted in the shallower parts of the bottom and extending above the water surface. Techniques to control vegetation should be used in such cases (Section 4.3.4). Two steady state conditions exist in shallow waterbodies, one with increased turbidity due to increased mineral content or phytoplankton concentrations and low macrophyte vegetation, and one with dense vegetation and lower phytoplankton concentrations. These two steady states can switch in temperate regions, where it has been sufficiently studied, due to such phenomenon as changes in fish populations, which can increase turbidity because of lake bottom disturbances. This switch also can be related to increasing phytoplankton concentrations, which prevent light from penetrating to the lake bottom, thereby limiting the development of rooted vegetation. On the other hand, suppression of phytoplankton growth (e.g., by biomanipulation) can enhance macrophyte development (Scheffer, 1998; Perrow et al., 1997).

Typical differences between the water quality in shallow and deep waterbodies are shown in Figure 3.1. A deep reservoir is characterized by vertical differentiation of water masses, while the shallow reservoir is vertically more or less homogeneous, but with significant horizontal differences at small distances caused by wind and macrophytes. Anoxic

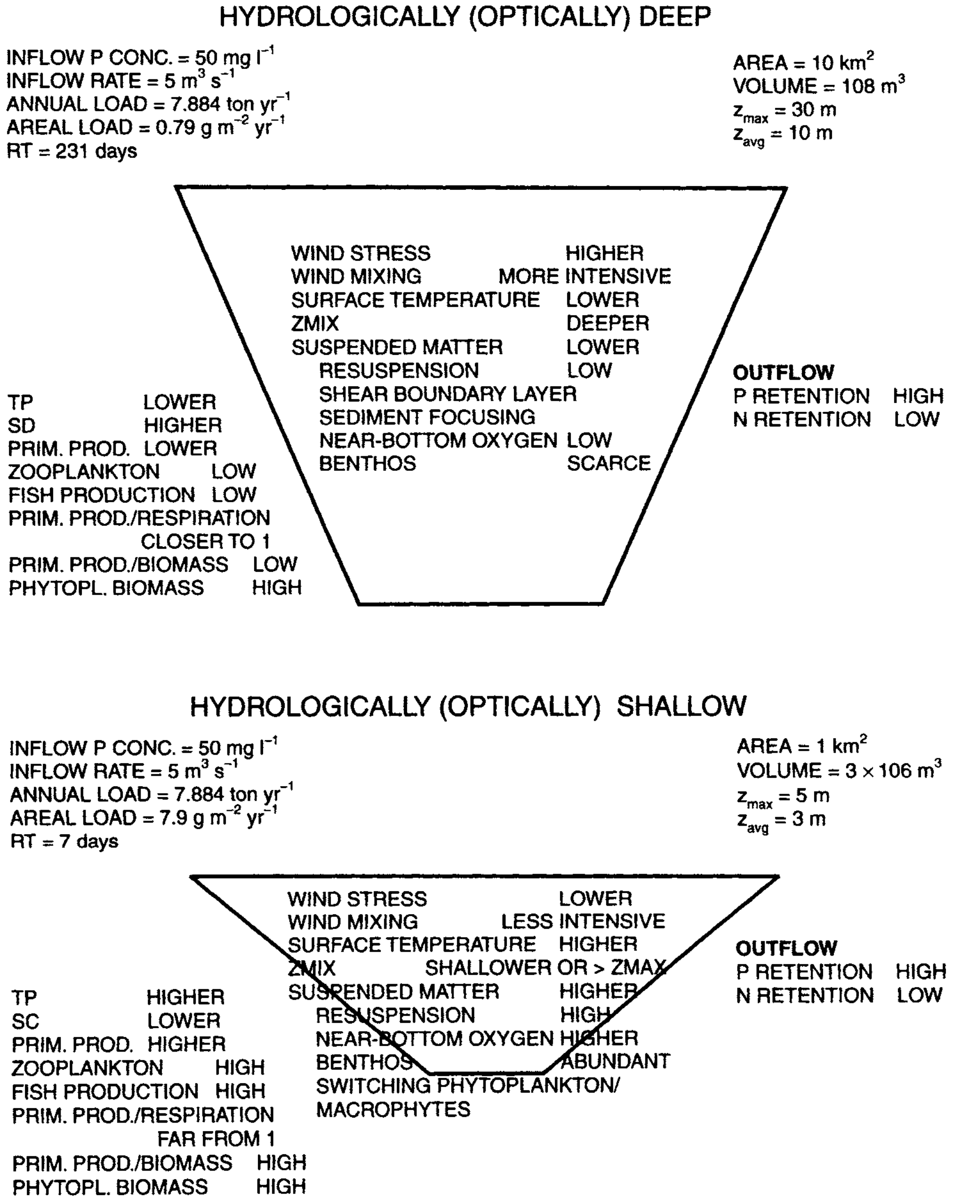
HYDROLOGICALLY (OPTICALLY) DEEP
INFLOW P CONC. = 50 mg l^{-1}
INFLOW RATE = 5 m^3 s^{-1}
ANNUAL LOAD = 7.884 ton yr^{-1}
AREAL LOAD = 0.79 g m^{-2} yr^{-1}
RT = 231 days
AREA = 10 km^2
VOLUME = 108 m^3
z_{max} = 30 m
z_{avg} = 10 m
WIND STRESS HIGHER
WIND MIXING MORE INTENSIVE
SURFACE TEMPERATURE LOWER
ZMIX DEEPER
SUSPENDED MATTER LOWER
RESUSPENSION LOW
SHEAR BOUNDARY LAYER
SEDIMENT FOCUSING
NEAR-BOTTOM OXYGEN LOW
BENTHOS SCARCE
TP LOWER
SD HIGHER
PRIM. PROD. LOWER
ZOOPLANKTON LOW
FISH PRODUCTION LOW
PRIM. PROD./RESPIRATION CLOSER TO 1
PRIM. PROD./BIOMASS LOW
PHYTOPL. BIOMASS HIGH
OUTFLOW
P RETENTION HIGH
N RETENTION LOW
HYDROLOGICALLY (OPTICALLY) SHALLOW
INFLOW P CONC. = 50 mg l^{-1}
INFLOW RATE = 5 m^3 s^{-1}
ANNUAL LOAD = 7.884 ton yr^{-1}
AREAL LOAD = 7.9 g m^{-2} yr^{-1}
RT = 7 days
AREA = 1 km^2
VOLUME = 3 × 106 m^3
z_{max} = 5 m
z_{avg} = 3 m
WIND STRESS LOWER
WIND MIXING LESS INTENSIVE
SURFACE TEMPERATURE HIGHER
ZMIX SHALLOWER OR > ZMAX
SUSPENDED MATTER HIGHER
RESUSPENSION HIGH
NEAR-BOTTOM OXYGEN HIGHER
BENTHOS ABUNDANT
SWITCHING PHYTOPLANKTON/ MACROPHYTES
TP HIGHER
SC LOWER
PRIM. PROD. HIGHER
ZOOPLANKTON HIGH
FISH PRODUCTION HIGH
PRIM. PROD./RESPIRATION FAR FROM 1
PRIM. PROD./BIOMASS HIGH
PHYTOPL. BIOMASS HIGH
OUTFLOW
P RETENTION HIGH
N RETENTION LOW

Fig. 3.1.

Table 3.1. Environmental variables as they affect a deep and shallow lake (modified from Dokulil; and Istvanovics and Somlyódy)

Variable	Deep lake	Shallow lake
Short-term weather effects	Moderate	Very strong
Morphometry	Minor	Strong
Water-level fluctuations	Minor	Considerable
Flushing	Moderate–strong	Strong
Temperature stratification	Strong	Lacking or transient
Light limitation	Moderate	Moderate–strong
Diurnal variations	Weak	Moderate–strong
Littoral zone	Weak	Moderate–strong
Higher aquatic vegetation	Weak–moderate	Moderate–strong
Importance of sediments	Low	High
Resiliance	Moderate	Moderate–strong

conditions, a consequence of lake stratification, may occur in a shallow lake for a short period of a few days when the weather is calm and the days and nights are hot. This often happens in tropical regions. When the nights get colder and/or the wind blows, the water is again mixed. In a deep lake, the forces of wind and the heating of the lake water by sunlight are not sufficient to intensively mix the whole lake, with water layering occurring as a result. However, because of larger water volumes and lower phytoplankton production, a deep waterbody is less prone to anoxic conditions than shallow waterbodies. Thus, better quality drinking water supply can be obtained from a deep waterbody than from a shallow one. In shallow waterbodies, water quality interactions with bottom sediments is more intensive than in deep waterbodies. Nutrients released from the sediments, for example, are immediately available to phytoplankton, and the oxygen in the bottom water layer can be consumed intensively by the sediments.

There are several differences between shallow and deep lakes that have consequences for water quality:

- Shallow lakes have greater primary production, due to more light being available on average to phytoplankton. For the same nutrient load, therefore, eutrophication is more pronounced in shallow lakes than in deep lakes. Another important variable modifying

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Fig. 3.1. Schematic representation of water quality differences of a hydrologically-deep and hydrologically-shallow lake. The specific example also shows differences in the surface area of the two waterbodies, the larger having higher wind stress, more intensive wind-induced mixing and deeper z_{mix}. The lakes are throughflowing, as indicated by the characterization of the outflow, but the theoretical water retention time (*RT*) is highly different in spite of identical inflow rates. Inflow phosphorus concentration and flow are identical; thus, the total phosphorus load to both is identical. However, the areal phosphorus load is very different due to the difference in surface areas of the two lakes. *Prim. prod.* = primary production, *phytopl.* = phytoplankton. *Switching phytopl./macrophytes*—see the text. *Prim. prod./respiration* expresses the ratio of phytoplankton photosynthesis to respiration.

primary production, however, is turbidity and organic water color, which can decrease the availability of light for phytoplankton growth (Section 2.2.2—*Eutrophication*).

- In the usual absence of regular vertical differentiation, the horizontal distribution of water quality variables is determined largely by the effects of wind, which move and consequently accumulate particles (seston, plankton, debris, etc.) in the wind direction. The degree of exposure of a waterbody to wind is important. Chemical variables also can be unevenly distributed as a consequence of biological activities, including the microbial decomposition of accumulated materials. Attention should be given to this fact, therefore, in taking lake water samples. Further, the oxygen conditions and derived classifications for deep, stratified lakes cannot be transferred to shallow waterbodies.
- The contact of water masses with bottom sediments is much more intensive, and the effects on their water quality greater, in shallow lakes. The effects of diffusion from sediment to the water column might be increased. Resuspension of sediments also can contribute to an increased transfer of the matter accumulated in the sediments back into the water column.

In deep waterbodies, vertical gradients dominate, depending on the waterbody mixing type (see Section 1.3.2). Although horizontal gradients resulting from wind action are less developed, flow-generated gradients are more typical for deep reservoirs (Chapter 6).

3.2.2 Chemical Water Quality Classification of Lakes

In addition to the geologically-based, natural differences in aquatic chemistry, several major and many minor compounds produced as a result of human activities are reaching waterbodies in the form of water pollution. Lake classification schemes have been developed in respect to several of them, including organic matter, water hardness, acidity and nutrient contents. The first two categories are discussed in this section, while the third is treated in the next section in relation to the trophic classification of lakes.

Organic matter

With respect to its water quality effects, organic matter can be basically classified into two gross categories; namely, resistant and easily decomposable organic matter. Organic matter of natural (autochthonous) origin can be of variable proportions between these two categories. The autochthonous organic matter produced within a waterbody by organisms belongs largely in the second category. The two categories are considered can be expressed more or less adequately by two common arbitrary methods; namely, chemical oxygen demand (COD) and biochemical oxygen demand (BOD). In fact, the resistant organic matter can be approximated as the difference between COD and BOD. The importance of BOD, therefore, is independent on the origin of organic matter, as it is related to both the external pollution of a waterbody and to the consequence of eutrophication. It is a useful criterion for water quality, therefore, with respect to the possible utilization of a lake for various purposes. A lake classification based on BOD, to which can also be added electrical conductivity because of differences in soft and hard water for different purposes, is shown in

Table 3.2. Classification of waters on the basis of their possible utilization, based on conductivity and biochemical oxygen demand (BOD)

Conductivity	BOD	
	Low	High
Low	Class 1	Class 4
Intermediate	Class 2	
High	Class 3	Class 5

Class 1.	Probably suitable as a source of municipal water supply and for most other uses
Class 2.	Probably suitable as a source of municipal water supply, provided it is abstracted by means of a suitably designated dam. Probably suitable for drinking water by private consumers and probably for most other uses, but not for irrigation, except in special circumstances
Class 3.	Not suitable as a source of municipal water supply, nor for industrial use, nor ordinarily for irrigation, but in many instances suitable for drinking by private consumers and for watering cattle if the conductivity is not excessive
Class 4.	Probably suitable for irrigation, but not for drinking, stock watering or industrial purposes
Class 5.	Unsuitable for almost every use except perhaps irrigation under special circumstances

Explanation:
Low conductivity = a value below 750 μmho cm^{-1} at least 95% of the time;
High conductivity = a value above 2,250 μmho cm^{-1} at least 95% of the time;
Low BOD = a value less than 4 ppm at least 95% of the time;
High BOD = a value greater than 4 ppm at least 95% of the time.
A water described as probably suitable for some specific use must not, in fact, be accepted for that use until further details of the relevant specification have been determined and other matters considered.

Table 3.2. The differences between reservoirs containing soft and hard water are treated in Section 5.2.4.

The ratio between the autochthonous load of easily decomposable organic matter (AUEDOM) to the allochthonous load (ALEDOM) is an indicator of the ratio between organic pollution and eutrophication. The load of AUEDOM is the organic matter reaching a lake in its inflow, expressed as BOD. The ALEDOM is the amount of easily decomposable organic matter, equivalent to the phytoplankton production within a waterbody, also expressed as BOD.

Acidity

The origin of water acidity can be natural and man-made. Human-induced acidification is discussed in Section 2.2.2. Dixon (1998) discusses quality control for acid waters. The distinction between strong and weak acids is discussed by Henriksen and Seip (1980).

The natural origin of acid waters is either the direct dissolution of acidic rocks or the release of fulvic acids and related humic compounds from decomposing vegetation. Humic compounds affect a number of aquatic processes, including the binding of nutrients and the formation of chemical complexes. They also decrease sunlight penetration into the water column, directly affecting invertebrates and fish distribution (Hessen and Travnik, 1998).

Hardness

Soft and hard-water lakes exhibit significant differences in water quality reactions. In addition to differences in the use of hard and soft water for cooking and washing, there is a significant difference in eutrophication processes as well. The quantity of algae in hard water lakes receiving the same phosphorus load as soft water lakes is far less than the latter, due to the co-precipitation of phosphorus with calcite (Koschel, 1997; see also Section 3.4.2—*Phosphorus* and Section 3.4.2—*Chlorophyll-a*).

Differences exist mainly between lakes containing soft and hard water, and between clear-water and humic (brownwater) lakes. Softwater lakes are less buffered and, therefore, more prone to acidification. In an area affected to the same degree by air pollution and acid precipitation, lakes located on granitic rock substrates will have very low pH values (i.e., be very acidic) with the appearance of high concentrations of aluminium and the consequent absence of fish and mortality of invertebrates. In contrast, nearby lakes located on calcareous rock substrates will be unaffected by acidification. The degree of eutrophication also is different. Hard-water waterbodies react differently to phosphorus limitation than softwater ones. Because of phosphorus co-precipitation with calcite, thereby removing the phosphorus from use by algae, less phytoplankton will be produced in hard-water lakes for similar phosphorus loads (see Section 4.2.5—*Calcite Precipitation*). The trophic state of a waterbody in highly alkaline regions will be significantly lower than one with soft water. There also is a major difference in the preparation of coffee and tea between the two types of water. Hard water will produce bad-tasting tea, while coffee tastes better when produced with hard water. The transparency of a hard-water lake can be lower than that of a soft-water lake, due to coagulation and increased sedimentation of organic matter.

3.2.3 Indicators of Lake Trophy and Trophic Classification

The classical system of distinguishing lake water quality groups can be traced to August Thienemann (1882–1960). He classified lakes according to their trophic conditions into oligotrophic (low trophy), eutrophic (high trophy), and dystrophic (lakes of boggy character, with highly-colored water due to the presence of organic matter from decaying vegetation). The lakes were differentiated initially on the basis of the composition of their bottom sediments and the associated benthic fauna.

More recent indicators of lake trophic state are based on work initiated by Vollenweider (e.g., Vollenweider and Kerekes, 1982), which reflects the situation outlined in Section 2.2.2—*Eutrophication*, and Figure 2.4, which remains a major threat to lake water quality and trophic condition. Because of the combined effects of organic pollution, however, the situation in developing countries with rapid population growth and industrialization is more complicated (see also Section 2.2.2—*Eutrophication*). Trophic state indices are generally based on a few variables, representing both the causes and responses of trophic conditions. The primary cause of increasing trophy is the in-lake concentration of the critical limiting nutrient, which most often is phosphorus. Total phosphorus was selected as the typical measure of phosphorus concentrations in lakes because of its relatively easy measurement. Reactive (inorganic) phosphorus, which is more readily accessible to uptake by algae, is more difficult to measure accurately because of its rapid incorporation and release by aquatic organisms. The consequence of increased critical nutrient loads to a lake, and the resultant increased in-lake concentrations, is the increased production of organic matter within the lake, mainly in the form of algae.

The quantities of algae in a lake can be determined in three primary ways:

- The classic method is to determine the number of individuals of different species of algae per unit of water volume. Individuals of different algal species, however, can be very different in size.
- A more appropriate, but more time consuming method, is to determine the algal biomass by counting individual species and summarizing their volumes (considered equal to their fresh weight, assuming the specific weight of all species to be equal to 1). This approach is subject to large error if unit volumes are ascribed to each algal species, primarily because the fresh weight of individual algal species is far from constant. It depends on the growth conditions for the algae, and can vary by a factor of 10 or even 100. More exact results are obtained when the size of algae from a given locality and time period are measured with a microscope. The algal size classes must be distinguished and counted separately to summarize the multiples of the number of individuals in each size class, and the respective average weights for each size class. Modern particle counters can be used when there is not much interference of abiotic particles with the various phytoplankton sizes.
- The quantity of algae present can be much more rapidly and conveniently measured as the concentration of chlorophyll-a in the water samples. This is done either by spectrophotometric examination of the material collected on filters through which a known volume of lake water is passed, or by fluorimetric measurement of water samples containing live algae. Measurements can even be made in a lake, using modern submersible throughflow flourimeters. The values may be biased, however, because of major differences in the chlorophyll-a content in the biomass of different taxonomic groups of phytoplankton. Cyanobacteria, for example, contain less chlorophyll-a than other groups.

There are no unequivocal critical boundary levels for different variables and trophic states (Table 3.3). A recent evaluation for North American lakes by Nürnberg (1996) identified the values of the average summer total phosphorus concentrations in the epilimnion of lakes that define the boundaries between oligotrophic, mesotrophic, eutrophic and hypereutrophic lakes as, respectively, 10, 30 and 100 micrograms/liter ($\mu g\ l^{-1}$); the corresponding boundary values for total nitrogen are 350, 650 and 1200 $\mu g\ l^{-1}$. The respective boundary values for the summer chlorophyll-a concentrations were identified as 3.5, 9 and 25 $\mu g\ l^{-1}$, the summer transparency (measured with a Secchi disc) values were 4, 2 and 1 meter, and the values for the areal hypolimnetic oxygen depletion rate were 250, 400 and 550 mg m^{-2} per day. In identifying these boundary values for the different variables used to classify lakes on their basis of their trophic state, it is noted that in regions that are generally cleaner

Table 3.3. Ranges of Secchi disc transparency (SD), total phosphorus (TP), chlorophyll-a (CHA) and primary production (PP) considered indicative of particular trophic states by different authors (Busch and Sly, 1992)

Trophic degree	Variable			
	Secchi disc	Total P	Chlorophyll	Primary production
Oligotrophic	> 4	< 8	< 2.9	< 25
	> 5	< 10	< 3	< 30
	> 6	< 11	< 3.7	< 100
		< 15	< 4.3	< 145
		< 15		
Mesotrophic	2.5–4.00	8–23	2.9–5.6	25–75
	5–3	10–20	3–7	30–200
	6–3	11–21.7	3.7–10	100–200
		15–25	4.3–8.8	145–240
		15–30		
Eutrophic	< 2.5	< 23	< 5.6	< 75
	< 3	< 20	< 7	< 200
	< 3	< 21	< 10	< 200
		< 25	< 8.8	< 240
		< 30		

Explanation:
Secchi disc transparency expressed in meters (m);
Total phosphorus and chlorophyll concentrations expressed in micrograms/liter ($\mu g\ l^{-1}$);
Primary production expressed in milligrams/meter^2 · day ($mg\ m^{-2}\ day^{-1}$).

and less productive, lower margins are considered adequate, while in more productive and polluted regions the boundaries shift to higher values.

Based on the results of evaluations of a number of measurements from around the world, Vollenweider (1968, 1987) developed indicators of lake trophic state based on the relations between their phosphorus and chlorophyll-a concentrations. The summer phosphorus concentrations in his study lakes were lower in the winter and early spring. These conditions are the result of the magnitude of the input of phosphorus to a lake from its inflows and in-lake sediments on the one hand, and its removal via algal uptake in the lake on the other hand. The summer concentrations of both variables depend on the phytoplankton species composition, because different algal species have differing capabilities to utilize dissolved phosphorus in a lake. A more suitable comparison for lakes, therefore is to use the concentrations of the phytoplankton existing in the spring, before they begin to significantly utilize the phosphorus and grow rapidly. An even better indicator of lake trophic state than the in-lake phosphorus concentration is the phosphorus load to the lake (i.e., the total mass of phosphorus entering a lake over some (usually annual) period). The phosphorus load is often calculated as the weighted sum of the water flowing into the lake and the phosphorus concentration in the inflowing water. Figure 3.2 provides a graphic representation of the

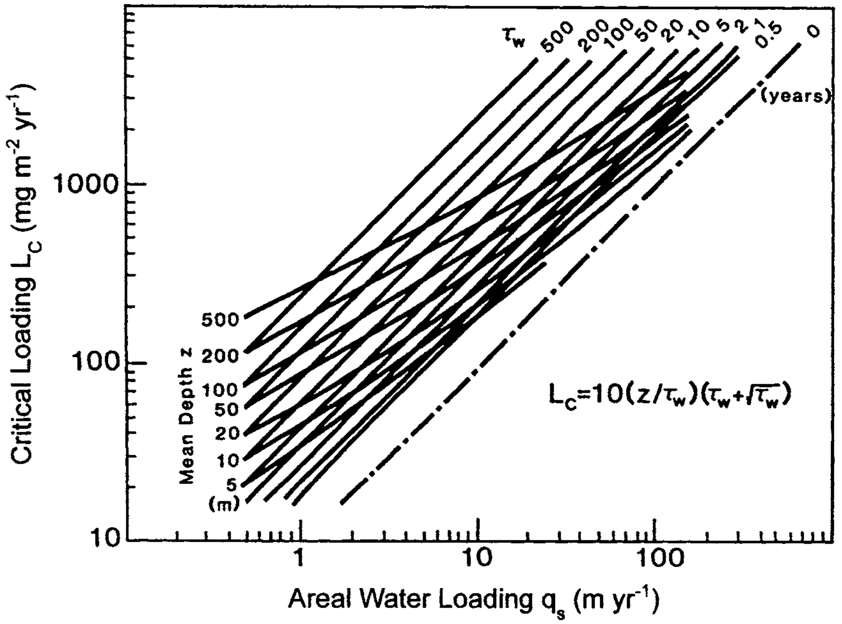

Fig. 3.2. Vollenweider diagram for critical phosphorus loading of lakes (to remain in an oligotrophic state), based on the areal water loading, mean depth and theoretical retention time (τ_W, expressed in years). The formula on which the graph is based is shown (Vollenweider, 1975).

estimated trophic condition of a lake on the basis of these relationships. The Vollenweider relationships indicating that the chlorophyll-a concentrations will be higher when the phosphorus load gets higher, however, appears to be biased when the phosphorus concentrations exceed about 50 µg l^{-1} of reactive phosphorus and about 100 µg l^{-1} of total phosphorus. Straškraba (1976) pointed out, on the basis of both theory and observations, that the quantity of chlorophyll-a cannot increase indefinitely, as lakes have an upper limit in their ability to produce new algal biomass. When algal populations get dense, the shading of the water column by algae (self-shading) becomes intensive, and the resulting limitation of light begins to limit algal growth as well. For a large range of phosphorus and algal concentrations, therefore, the phosphorus–chlorophyll-a relationship cannot follow the continuous increase deducible from the relationships derived by Vollenweider and others (e.g., see Rast and Thornton, 1996). Rather, the relationship should be asymptotic in nature. This observation was subsequently supported by additional data analyzed by Straškraba (1985), Prairie et al. (1989), McCauley et al. (1989) and Rast and Thornton (1996). The shape of the dependence, and a procedure to determine the trophic state of a lake and its possible reactions to increasing or decreasing algal concentrations, is depicted in Figure 3.3. The asymptotic relationship between the total phosphorus–chlorophyll-a concentrations in a lake is due to light limitation. The degree of light limitation of algal growth in a given lake

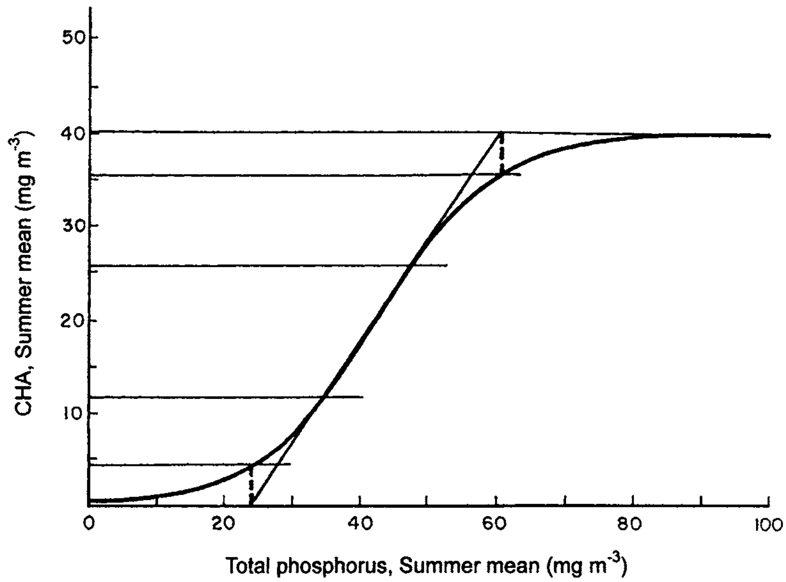

Fig. 3.3. Schematic representation of the average relationship of the summer mean chlorophyll-a (CHA) concentration to the summer mean total phosphorus concentration in phosphorus-limited, transparent, temperate, dimictic lakes. The lines indicate: 1—the critical point of chlorophyll rise; 2—the onset of linear increase; 3—the termination of the linear increase; 4—the termination of the steep rise; and 5—the saturation limit (for original data see Straškraba, 1976, 1985; Prairie et al., 1989; McCauley et al., 1989).

depends on the attenuation of light by the water and the materials contained in it, and on the depth to which the algae are mixed. Distilled water obviously has a very low light attenuation. High attenuation is caused by colored organic material in the water, described less precisely as "humic acids", and also by mineral particles suspended in the water. Algae are usually mixed within a certain water layer in a lake, due to wind activity and/or convective mixing. If the mixed layer is shallow, as in a shallow or small lake, and a lake is protected from wind by its surrounding landscape, the asymptotic concentration of chlorophyll-a in the lake is higher. In a larger, less protected deeper lake, the water is mixed intensively to a greater depth and the algae are periodically moved to less-illuminated deeper water layers, thereby receiving less sunlight energy on average. In such cases, the asymptotic algal concentration is lower. A similar difference exists between lakes with low color and low mineral turbidity, and lakes where light attenuation is high as a consequence of high color and turbidity. Thus, the asymptotic limit of the chlorophyll-a concentration (the concentration not limited by nutrients), and how it is reached in lakes limited by the critical nutrient, can be represented by a family of curves given in Figure 3.4. Quantitative relations are given in Section 3.4.2—*Chlorophyll-a.*

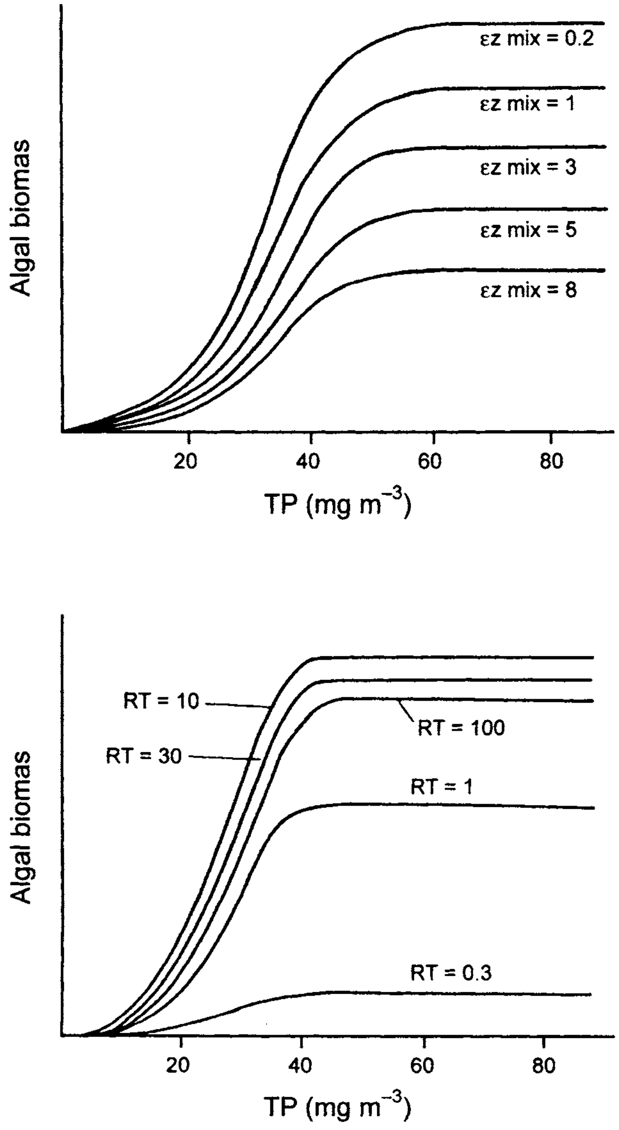

Fig. 3.4. The chlorophyll-a concentration and primary production in lakes and reservoirs depends not only on the critical nutrient (in this case phosphorus, *TP*), but simultaneously on other variables as well. Schematically, the effect of the optical depth (multiple of the extinction coefficient and mixing depth (εz_{mix}, for explanation see Fig. 4.17) and water retention time (*RT*, days) is shown.

Because algae produce biotic turbidity and the natural turbidity of lakes is generally low, a very simple measure of the trophic state of the lake is often made on the basis of its clarity, determined by a Secchi disc. The Secchi disc depth, or transparency, is determined as the depth at which the contours of a white–black disc cease to be distinguishable. Quantitative relations are given in Section 3.4.2—*Evaluation of Water Transparency.*

Morphoedaphic index (MEI)

The index first developed by Ryder (1965) was intended as a means of easy estimation of the potential fish yield of lakes. It is based on mean lake depth as a morphological variable, and total dissolved solids (TDS) as the edaphic variable. It was extensively evaluated in fishery literature, which illustrated its limits and weaknesses (Ryder et al., 1974; Henderson and Welcome, 1974). Geographical variables play major differentiating roles, as do the types of fishery and other variables. The use of the MEI in a management perspective was the goal of the investigation by Vigni and Chiaudani (1985). They found a highly significant correlation between the MEI, calculated as the ratio between the mean depth and the alkalinity (or the conductivity), and the phosphorus concentration in lakes not significantly affected by the human-induced phosphorus input. This allows for the estimation of the trophic condition of a lake before it is extensively settled and, hence, indicates the maximum possible improvement attainable with management options. The regressions, equations based on analysis of numerous European and North American lakes, were as follows:

$$\log P\ [\mathrm{mg\ m^{-3}}] = 1.48 + 0.33(\pm 0.09) \log MEI_{\mathrm{alk}},$$

$$\log P\ [\mathrm{mg\ m^{-3}}] = 0.75 + 0.27(\pm 0.11) \log MEI_{\mathrm{cond}}.$$

The present difficulty is that widespread artificial fertilization results in the continuous increase not only in phosphorus and nitrogen concentrations, but also in alkalinity and conductivity as well.

Carlson trophic state index (TSI)

Although a more complicated and locally more limited trophic state index (TSI) was elaborated earlier by Shannon and Brezonik (1972), the TSI attributed to Carlson (1977) has been widely used in North America. It is based on relative values for three in-lake variables, including average summer chlorophyll concentration (CHA), summer total phosphorus concentration (TP), and summer Secchi disc (SD), with the values ranging from 0 to 100. The primary assumption with this TSI is that CHA is a good measure of the phytoplankton biomass in a lake, that the in-lake total phosphorus concentration is the growth-limiting factor for phytoplankton, and that water transparency measured as SD is an additional decisive variable related mainly to CHA. Each variable can be evaluated separately as *TSI*(CHA), *TSI*(TP) and *TSI*(SD), or the three can be averaged (using different weights). The relative values are based on regression equations derived from local data and, therefore, subject to local differences in the relations and effects of these three variables. Because each of these three variables has some drawbacks in uniquely defining lake trophic state, the advantage of this approach is the averaging of these variables with the regression character of the relations. Averaging the values also helps reduce the effects of random data errors in each type of measurement. The relations between the TSI and these three variables are as follows:

$$TSI(\mathrm{TP}) = 10\left(6 - \frac{\ln(48/TP)}{\ln 2}\right),$$

$$TSI(\mathrm{CHA}) = 10\left(6 - \frac{2.04 - 0.68 \ln CHA}{\ln 2}\right),$$

$$TSI(\mathrm{SD}) = 10\left(6 - \frac{\ln SD}{\ln 2}\right).$$

Re-examination of other data, however, always results in different TSI values (e.g., Lambou et al., 1982), which is a characteristic of empirically-derived relations. Particularly significant differences are found for lake in which phosphorus is not the algal growth-limiting nutrient, or the non-algal turbidity or color of the water is high. An attempt to increase the precision of the estimates by incorporating the ratio of the volatile suspended solids to the total suspended solids, as a measure of the sediment effects on transparency and light availability, was suggested by Swanson (1998).

The importance of nitrogen limitation of algal growth was first recognized in regard to lake trophic state for shallow, subtropical lakes of Florida, for which Kratzer and Brezonik (1981) developed TSI relations based on nitrogen, rather than phosphorus. Shannon and Brezonik (1972) elaborated earlier multiple regression-based TSI values for the same sets of data. The validity of this index reflects only the region for which it was elaborated. Non-algal turbidity is more important in reservoirs than in natural lakes, and also in lakes in dry regions and in the tropics. For such reservoirs, Walker (1980, 1983, 1984) reported that the index has to be modified and elaborated in order to be useful for reservoirs in the United States. Instead of total phosphorus concentrations, Walker used a composite nutrient concentration (X_{PN}), using the ratio of nitrogen and phosphorus based on their assumed limitation: X_{PN} [mg m^{-3}] $= [TP^{-2} + ((N - 150)/12)^{-2}]^{-0.5}$, where TP is the total phosphorus concentration and N is the total nitrogen concentration, both expressed in mg m^{-3}. Using multivariate analysis, he detected two composite axes, one corresponding to the total quantities of nutrients and light extinction, and the second to the partitioning of nutrients and light extinction between organic and inorganic forms. In an effort to avoid the error in using Secchi disc values (SD) for waterbodies with high levels of non-algal turbidity, Aizaki et al. (1981) substituted particulate organic carbon and particulate organic nitrogen for SD in classifying Japanese lakes.

An interesting evaluation using the TSI methodology was done by Havens (1994) for a shallow subtropical lake Okkechobee in Florida. He followed the seasonal changes of deviations of TSI(CHA—TP), the difference between TSI(CHA) and TSI(TP), and TSI(CHA—SD), finding that deviations among trophic state index variables indicated both the degree of nutrient limitation and the composition of seston in the lake. The degree of nutrient limitation was positively correlated with the TSI deviation (CHA–NUTRIENT), negative values indicating limitation by the nutrient under consideration. Seston particle size is positively correlated with the TSI deviation (CHA–SD), with positive deviations indicating presence of large algal particles which attenuate less light than the same mass of small particles, and vice versa for negative deviations (Fig. 3.5).

Thornton and Rast (1993), Lind et al. (1993) and Rast and Straškraba (2000), among others, demonstrated that application of common trophic classifications to reservoir ecosystems is inappropriate and misleading. In addition to higher nutrient loads from their typically-larger drainage basins and non-algal turbidity, a primary cause is the short retention time of some reservoirs, causing more rapid flushing of the algae from the lake, and strong longitudinal gradients resulting in longitudinal differences in non-algal turbidity, nutrient and algal concentrations.

The examples in Table 3.2, all from the temperate region, are illustrative of local differences in trophic classifications. These differences are due mainly to regional differences in nutrients and lake depth, as well as the subjective nature of estimations of lake trophic state. In naturally nutrient-poor regions with deep lakes, where most lakes are very clean, for example, a lake with slightly elevated quantities of algae and slightly reduced water transparency will be considered mesotrophic, while a similar lake located in a region that is naturally richer in nutrients, or is a shallower lake, would be considered oligotrophic. Other specific conditions are found in hard water waterbodies (Section 5.2.4).

With respect to trophic conditions, tropical lakes seem to be systematically different from lakes in the northern temperate regions of the world. Some authors claim there also are differences between conditions in the temperate lakes of the Northern and Southern Hemispheres. Salas and Martino (1991) summarize investigations of methodologies for simplified evaluation of eutrophication in warm water, tropical lakes. Their elaboration, based on 44 data sets, is strongly biased toward reservoirs and lakes with short retention time, with the geometric mean of the water retention time for their entire lake data set being 100 days. For reservoirs, the simple mean value of the water retention time was 87 days.

Areal hypolimnetic oxygen deficit (AHOD)

Hutchinson (1938) derived a measure of the rate of oxygen depletion in the water column of deep lakes, expressing the magnitude of the oxygen deficit as the extent of the anoxic

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Fig. 3.5. Evaluation of seasonal trends in a shallow subtropical lake by means of deviations between two forms of the Carlson's TSI index—based on nutrient concentrations and based on Secchi disc readings. A—Seasonal trend for monthly means during 1980–1992; B—Evaluation of the deviations between the index based on chlorophyll-a versus that based on the nutrients and that based on Secchi disc readings. Observations are for a shallow subtropical lake (Lake Okkechobee, Florida), (Havens, 1994).

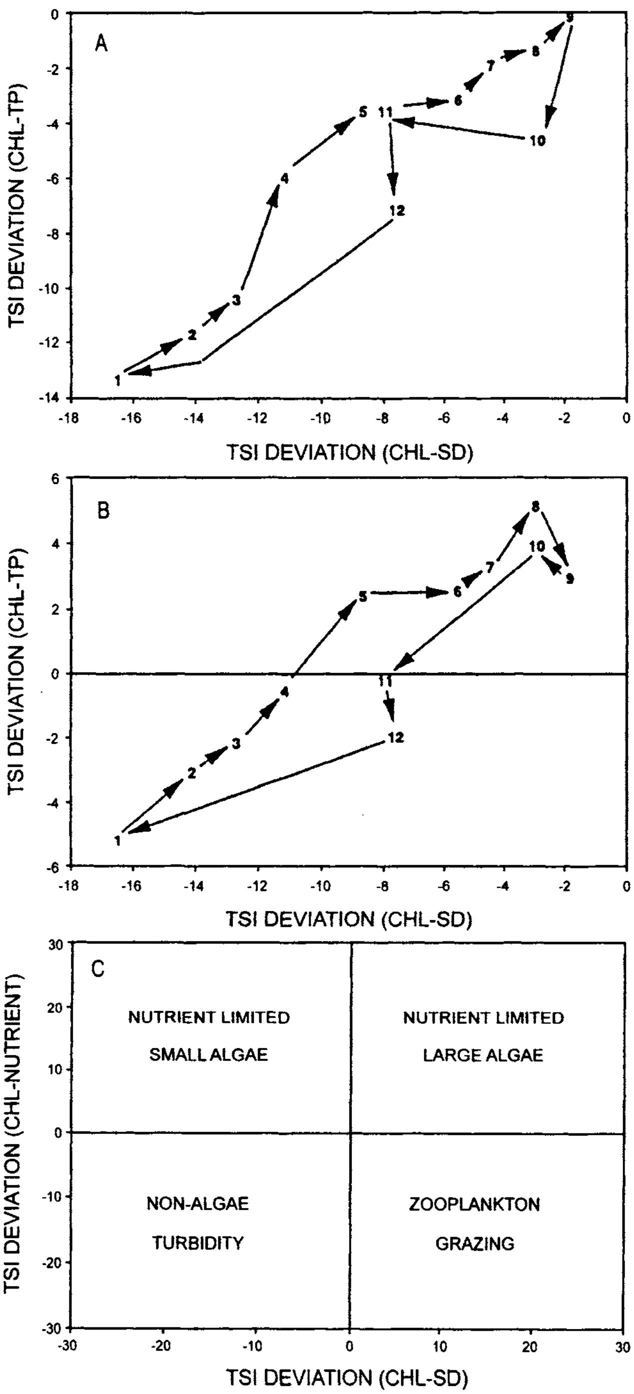
A
TSI DEVIATION (CHL-TP)
TSI DEVIATION (CHL-SD)
B
TSI DEVIATION (CHL-TP)
TSI DEVIATION (CHL-SD)
C
NUTRIENT LIMITED
SMALL ALGAE
NUTRIENT LIMITED
LARGE ALGAE
NON-ALGAE
TURBIDITY
ZOOPLANKTON
GRAZING
TSI DEVIATION (CHL-NUTRIENT)
TSI DEVIATION (CHL-SD)

Fig. 3.5.

(oxygen-depleted) area. It is determined by measuring the development of oxygen stratification in the hypolimnion of a lake or reservoir in the late summer (Wetzel, 1975), and represents a measure of the cumulative effects of all in-lake, oxygen-depleting processes. It is based on the fact that the bottom water layer (hypolimnion) of a lake is isolated from the oxygen supply in the epilimnion as a consequence of thermal (and density) stratification.

Prat et al. (1992) found in Spanish lakes, and generally in southern European lakes at altitudes lower than 1500 meters, that because of long stratification periods, coupled in many cases with high in-lake sulfate concentrations, the oxygen depletion close to the lake bottom occurs at lower levels of eutrophication than in northern and central European lakes. Mitchell and Burns (1979) investigated this phenomenon in warm, monomictic lakes in New Zealand. Conditions different from the temperate region are expected in tropical regions, with more rapid oxygen consumption accompanied by more frequent and often-irregular mixing. The consequences of anoxia resulting from internal phosphorus loads were clarified by Nürnberg (1987).

The determination of hypolimnetic oxygen depletion (HOD) in reservoirs is frequently precluded by deviations in the assumptions of the calculation method. These deviations include significant heat gain and seasonal deepening of the thermocline, inter- and underflowing density currents, hypolimnetic discharges and spatial heterogeneity. As a useful alternative, an anoxic factor (AF) that describes the relative degree of anoxia at the bottom of a lake was developed by Myers and Kennedy (1994). The factor calculates the total number of days that a sediment area equal to the lake surface area is overlain by anoxic water:

$$AF = \left(\sum_{i=1}^{n}(t_i a_i)\right) \Big/ A,$$

where: t_i = duration of anoxic period,
a_i = surface area of the anoxic water layer (m^2),
n = total number of anoxic periods,
A = reservoir surface area (m^2).

Anoxia is defined as an oxygen concentration less than 0.5 mg l^{-1}. The areal extent of the anoxic layer in a lake is obtained from the corresponding depth-area relations.

Comparison of trophic state indices

A composite index (*Lake Condition Index*) intended to indicate lake conditions with respect to management possibilities was developed by Uttomark and Wall (1975). In addition to characterizing the lake trophic state, it contains penalty points for the hypolimnetic dissolved oxygen, transparency, fish kills and extent of macrophyte or algal growth.

In an attempt to elaborate a complex index of trophic state for lakes in southern Germany, Schröder and Schröder (1978) analyzed 59 annual averages from 25 lakes, using multiple regression techniques. Their results are presented in Figure 3.6. The anabolic component of lake metabolism due to productivity processes, and the catabolic component due

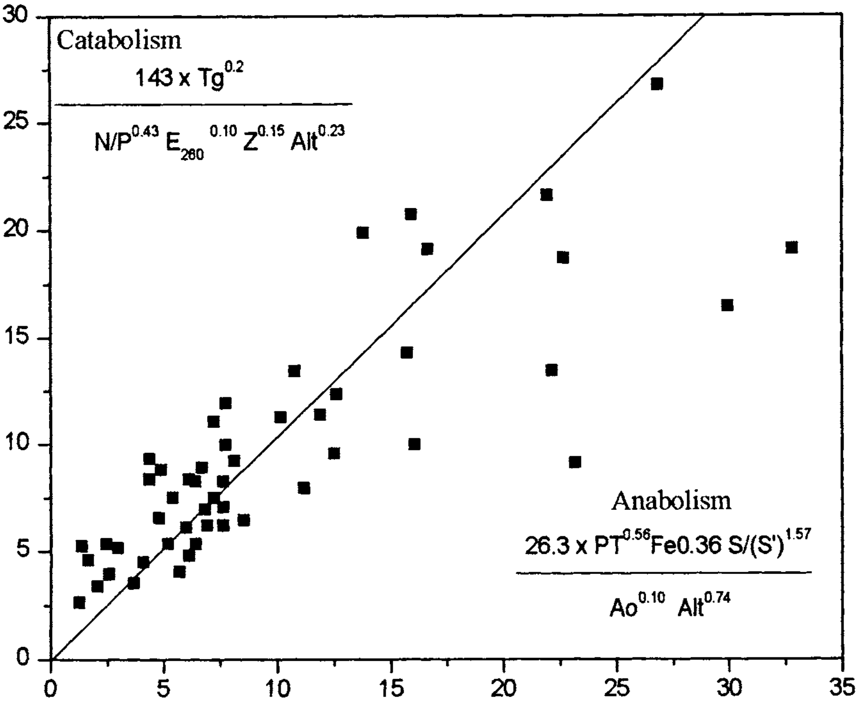

Fig. 3.6. The position of a number of German and Austrian lakes on the axes represented by anabolism and catabolism. The definitions of anabolism and catabolism are indicated on the figure (Schröder and Schröder, 1978).

to decomposition, were considered separately. In lakes in southern Germany, both components were in balance, which is expected for lakes where the dominate source of pollution is represented by eutrophication. However, in waterbodies that are not only eutrophic, but also polluted with nontoxic sewage (often found in less-developed regions), the separate evaluation of the catabolic and anabolic components might be important in distinguishing the ratio of different pollution sources (nutrients versus organic matter).

Lambou et al. (1983) examined several indicators for 44 lakes, selected from among 250 analyzed by the U.S. Environmental Protection Agency to represent as full a range of water clarity as possible (i.e., the Secchi depth ranged between 0.15–5.6 m). The surface areas of the study lakes ranged between 0.23–263 km^2, the mean depths between 0.9–21 m, the summer chlorophyll-a between 1.4–595 mg m^{-3}, and the total phosphorus concentrations between 5–1600 mg m^{-3}. A classification of the trophic state of lakes overgrown by macrophytic vegetation was attempted by Canfield et al. (1983).

Table 3.4 illustrates some relationships between the trophic state of a lake and its possible utilization for various purposes.

Table 3.4. Relation between the trophic degree of a lake and the possibility of its utilization for various purposes

Utilization	Trophic degree	
	Required	Tolerable
Drinking water production	Oligotrophic	Mesotrophic
Process water	Mesotrophic	Slightly eutrophic
Cooling water		Eutrophic
Irrigation		Strongly eutrophic[1]
Energy production		Strongly eutrophic[2]
Bathing	Mesotrophic	Slightly eutrophic[3]
Other water sports	Mesotrophic	Eutrophic
Fish culture—salmonids	Oligotrophic	Mesotrophic
—cyprinids		Eutrophic

[1] Danger of clogging of the irrigation spray jets.
[2] Impairment to turbines due to clogging by macrovegetation, to the mechanics of turbines by aggressive bottom waters.
[3] Within the concept of landscaping, the results of a natural eutrophic aging process can be desirable.

3.2.4 Ecological Indicators

Systems ecology is the branch of ecology that deals with whole ecosystems, rather than species populations and communities. A number of ecosystem state indicators has been developed with potential importance for management purposes (biomass—Margalef (1968); sensitivity (buffer capacity)—Jørgensen and Mejer (1977); exergy—Jørgensen (1982); ascendancy—Ulanowicz (1997)). Dalsgaard et al. (1995) have used biomass, harvest, production to biomass ratio, throughput of a critical element, and other indicators to assess the ecological sustainability of an agricultural ecosystem. Efforts also have begun to apply these indicators to characterize both the natural and stressed states of aquatic ecosystems. However, except for biomass and productivity, only preliminary indications for water quality applications of these indicators currently exist.

Biomass and related units

Biomass is a static measure of the amount of organisms present in a system, expressed either as total biomass, the biomass of specific groupings of organisms, or the biomass of individual species. The biomass of phytoplankton in lakes is a commonly used measure of eutrophication, with the biomass expressed in units of chlorophyll-a. Some ecological evaluation systems use the biomass of zooplankton or benthos, and mutual relations between zooplankton and phytoplankton (e.g., the Polish system of lake classification—Kudelska et al., 1991; Panczakowa and Szyszka, 1986; Hillbricht-Ilkowska and Kajak, 1986). *Harvest* is the mass of organisms that can be obtained by humans (harvested) from a natural or cultivated system. *Productivity* refers to the mass of organic matter produced by organisms per unit volume or area of water over a given unit of time, either from mineral elements (primary production) or by reworking already-formed organic matter (secondary production).

The *Productivity/biomass ratio*, (P/B), is the specific productivity (per unit biomass) that measures the activity of particular organisms or their groups. That is, how much biomass per unit time is an organism (or groups of organisms) able to produce per unit of biomass? *Respiration* refers to the decomposition of organic matter, which can occur simultaneously with production, or can occur separately (during the night for phytoplankton and vegetation). *Decomposition* refers to the mass of organic matter decomposed to lower forms of organic matter per a given unit of time. The *Production/respiration ratio*, if measured in the same units (i.e., mg O_2/unit volume of water per unit of time), expresses the intensity of an aquatic ecosystem's metabolism, and can be used as an indicator of the prevalence of either the autochthonous (i.e., naturally formed) production of organic matter (eutrophication), or of the prevalence of the decomposition of the organic matter brought to the waterbody from outside it. See also the anabolic and catabolic components of Schröder and Schröder (1978) in Section 3.2.2—*Comparison of Trophic State Indices.*

Exergy is a thermodynamic concept, defined as the amount of work a system can perform until it reaches a thermodynamic equilibrium with its surroundings. Exergy is used in ecological models to account for changes in the composition of assembly processes by various organisms. It is formulated to express not only the chemical composition of the organisms and their quantities, but also the information value of different organism groups, expressed by means of the number of their structural genes (Jørgensen et al., 1995). Exergy has been shown to express the degree of inverse chemical stress exerted on the systems in simple models of aquatic ecosystems and in experimental ecosystems (Xu et al., 1999).

Ascendancy

A recent scientific measure of the simultaneous changes in both the structure and function of an ecosystem suggested as useful for management purposes is the ascendancy concept of Ulanowicz (1986, 1997), which is suitable for addressing long-term effects, and the "scope for change in ascendancy" of Genoni and Pahl-Wostl (1990). Only preliminary management evaluations have been done with these measures.

Buffer capacity

This variable was coined by Jørgensen and Mejer (1977) as the sensitivity of one variable of an ecosystem to some input variable. An example is the sensitivity of phytoplankton changes in a model to changes in the input phosphorus concentrations. The buffer capacity describes the ability of particular model ecosystems to adjust to changes at a particular time.

3.2.5 Organism Species as Biological Indicators

There are three basic approaches to the use of biological indicators (bioassessment):

- Using the indicator value of individual species,
- Using species assemblages, either all representing one taxonomic group, or representing all species in a community in a particular habitat,

- Summarizing characteristics of the biological assemblage, via index of diversity, species richness, evenness, heterogeneity, etc.

The first two approaches are less general because the organism species have limited geographical distribution and the classification systems are only valid within the geographical limits of their occurrence. For large countries, it is even necessary to distinguish different ecoregions. The third approach is the most recent one and, therefore, not yet fully elaborated. However, it has the greatest potential for general use. Although there seem to be geographical trends in measures of diversity, with richer assemblages in tropical regions, the general trends of the summary characteristics are similar in temperate and tropical lakes. Loeb and Spacie provided an historical review of biological monitoring of aquatic systems in Europe.

Practical aspects of bioassessment and biocriteria are treated in detail in the technical manual of biological approaches to water quality management of the U.S. Environmental Protection Agency (EPA, 1998). Particular attention is given to primary producers (trophic state assessment) phytoplankton assemblages, submerged macrophytes, sediment diatoms, benthic macroinvertebrates, fish, zooplankton and periphyton.

There are a number classification systems based on *individual organisms* belonging to benthic Oligochata and Chironomidae, as well on phytoplankton, zooplankton and fish species (Premazzi and Chiaudani, 1992; Busch and Sly, 1992). The negative feature of this approach is the limited geographical distribution of most organisms, and the dependence of the ecological value of individual species (on which this approach is based) not only on abiotic variables, but also on biotic interactions with other species present in the locality. Thus, if the composition of the whole assemblage is different, the values derived from one condition are not necessarily valid for other conditions. A typical example of this approach is the saprobic Kolkwitz–Marsson system extensively utilized in several European countries for the general characterization of river stretches. Its use for lakes is of little value, however, because nearly all of them end up being characterized as β-mesosaprobic.

Because algae of different sizes interfere with the treatment of drinking water because of filter clogging, another idea was to create a treatability index. For example, based on sizing phytoplankton, a prediction of the water throughput capacity of water treatment plants employing rapid gravity filters can be developed.

The lake classification based on *organism assemblages* was first developed by two eminent scientists, the Swedish limnologist Nauman and the German scientist Thienemann. The first dealt with phytoplankton, and this is probably the reason the Nordic countries predominantly use lake classification systems based on phytoplankton (Heinonen and Seuna, 1997). The classification system of Thienemann is based on lake benthos. A more recent example of an organism association-based classification system utilizes fish associations. Fish-based systems have been used, in relation to restricted geographical distribution of fish, for local classifications of lakes and reservoirs. Dolman (1990) classified water quality and fishery data from 132 larger Texas reservoirs by means of cluster analysis, recognizing five major groups of fish assemblages. When 19 water quality variables were used, a general east-to-west separation of species associations by water quality, and a northwest-to-southeast separation by surface elevation and growing season, emerged in this study.

Methods based on different *diversity indices* are rapidly developing, as a result of increasing public and scientific interest in biodiversity changes related to environmental degradation. Different types of diversity indices are in use, including species richness, the Shannon–Weaver index derived from informatics, evenness and heterogeneity (EPA, 1998). The first attempts to utilize diversity indices for water quality characterization focused on determining if aquatic diversity in river stretches exhibiting different states of pollution exhibited differing responses.

The simplest biological index is species richness, giving the number of species found in a particular locality. The dependence of this variable on the area being sampled, and the determination skill of different investigators using it, prevents the strict comparability of results from different studies. The *Shannon–Weaver index* (D_{SW}) is superior because of its independence of sample size, and its sensitivity to changes in evenness of distribution for a small number of species and its insensitivity to rarer, missing species. It is calculated as $D_{SW} = -\sum_s P_i(\log P_i)$, where s is the number of species and P_i is the frequency of the species i. Values approaching 3 indicate highly-diverse systems of high quality. The *index of evenness* of distribution of individuals among species by Patten (1963) reaches values of 1 for a distribution with the same number of individuals in each species, and 0 for an extremely skewed distribution. Good water quality is characterized by high values of the index. *Other indices* (e.g., Brillouin, Simpson or Williams' α) are less commonly used because the underlying conditions are quite restrictive. Ecologically, a difference is made between α, β and γ diversity, where α diversity is the species richness mentioned above, giving the number of species per unit area in a standard sample, β diversity is a measure of the replacement of species between different localities, and γ diversity is used for larger areas (rivers, lakes), combining α and β diversity for different localities into a common index.

There appears to be a relation between the Shannon–Weaver diversity index and the trophic state of lakes, with the average value of the index decreasing with increasing eutrophication, as observed for both shallow and stratified lakes in Poland by Hillbricht-Ilkowska and Kajak (1986).

3.2.6 Ecotoxicological Indicators

With the production and use of more and more chemical compounds, xenobiotics are increasingly threatening the environment. Methods and indicators of their effects on the environment and humans are being developed, but cannot keep pace with the rate of creation of new products by the chemical industry. Because of the rapid recent development of this field, this section differs from the others in this chapter in that methods and their problems will be highlighted (Cairns, 1986; McCarthy and Shuggart, 1991; Lewis, 1991; Dallinger and Rainbow, 1992; Hoffman et al., 1995; Quint et al., 1996; Wells et al., 1998).

The effects of xenobiotics have been detected for cells and organs, for individual organisms, and for populations, communities and ecosystems. It also is necessary to distinguish between the short-term effects (measured in minutes) and the long-term or chronic effects (measured in time scales of organism generations). The most commonly-used approaches for detecting the impacts of xenobiotics on aquatic life are as follows:

- *Classical toxicity* tests on individuals or cultivated populations of aquatic organisms are used to detect short-term, lethal effects. In the simplest case, the test consists of determining the LC_{50} (Lethal Concentration at which 50% of the test organisms die within a specific time interval) or the LD_{50} (Lethal Dose for 50% of the population). Standardization of aquatic toxicity tests to maximize their major advantage, namely the rapid comparative evaluation of many xenobiotics, is discussed by Soares and Calow (1993).
- A more recent method based on the use of semipermeable membranes (Sabaliunas and Sodergren, 1997).
- Methods for detecting longer-term, *sublethal effects* on individuals and populations focus on measuring the different reactions of test organisms. An example is the behavioral test with the freshwater bivalve mollusc, *Dreissena polymorpha* (Mouabad and Pihan, 1993). The detection limits of the effects of xenobiotics measured with this method are several orders of magnitude higher than those of short-term toxicity tests.
- Methods detecting *genotoxicity* focus on the effects of genetic character, acting on future generations. Genotoxic properties of environmental pollutants can be assessed by measurements of damage to deoxyribonucleic acid (DNA). Metallothionein is used to assess the exposure of animals to such xenobiotics as heavy metals (e.g., Garvey, 1990).
- *Biomarkers* can be used to detect the stress effects of the long-term exposure of individual organisms living in contaminated environments. Body fluids, cells or tissues that can highlight the presence of contaminants and/or the magnitude of the host response in biochemical or cellular terms are measured (McCarthy and Shuggart, 1991). Examples of markers include the enzyme activity in cells of target tissues, serum enzyme activity, the appearance or loss of the reaction product of an enzyme, or major changes of a cellular component (e.g., glycogen, fat, mucus).
- *Microcosm* and *mesocosm studies* are used to detect the effects on communities and ecosystems (e.g., Pritchard, 1981; Pritchard and Bourquin, 1984; Cairns and Pratt, 1985; Cairns, 1986; Bennet and Girling, 1991). When xenobiotics are added, the changes in some common properties, including species composition, diversity, photosynthesis and/or respiration, are measured.
- The long-term consequences of contaminants in nature can be monitored by evaluating the *species diversity* and loss of sensitive species in biocenoses and ecosystems. Low levels of perturbation may cause changes in community composition or structure without changing the function. A rate process, such as respiration or photosynthesis, may be reduced under sublethal stress but the community structure left intact. According to Howarth (1991), the most predictable response seems to be the disappearance of sensitive species from the communities. A similarity of responses across different types of ecosystems to a variety of toxic substances was detected in the above-noted study.
- Toxicity also can be predicted on the basis of the *molecular structure* of the respective substances, using Quantitative Structure Activity Relationships.
- Changes in the morphologic or morphometric characteristics of organisms, and detection of pathological individuals, also can be used to detect the impacts of xenobiotics on aquatic life.

Critical evaluation of different approaches to aquatic toxicity
Short-term, lethal toxicity tests have many advantages in evaluating aquatic ecosystems, as well as many disadvantages (Bartell, 1991). Their major advantage is the possibility of rapidly comparing the relative toxicity of specific chemicals, or of specific mixtures (e.g., effluents). The limitations become obvious when trying to use their results to evaluate an entire aquatic ecosystem. They do not account for the effects of a number of components, as follows:

- Chemical speciation in the environment,
- Kinetics and hysteresis in sorption of chemicals to sediments,
- Bioaccumulation through the aquatic food chain, in standing and flowing waters, from the lowest level of algae and plants to fish,
- Modes of toxic action on the survival, growth and reproduction of individual species populations.

For aquatic sediments, an additional inadequacy is that the sediment toxicity tests often rely on the equilibration of chemicals in a sediment–water mixture, or else employ benthic organisms whose survival, growth, or reproductive success may be dependent not only on the presence of toxicants, but also on the physical and chemical characteristics of the sediment.

The application of the results of laboratory toxicity tests to natural populations is especially risky under these circumstances:

- The manner in which laboratory organisms are exposed to pollutants differs from their exposure in the natural environment,
- Laboratory tests deal with single chemicals, while organisms can be exposed to complex chemical mixtures in the natural environment,
- The criteria for effects in the laboratory are not important functional end points in population and systems dynamics.

3.2.7 Complex Water Quality Classification Systems

Complex or multimetric water quality classification systems are based on several variables, usually of a physical, chemical and biological nature, as well as the public perceptions of lake condition. Each of the criteria may itself be multimetric, based on a few or many variables. The indices are based on comparison to an operationally defined and measurable reference standard. As seen with the example of the Finnish deep lake system provided below, the individual attributes are first standardized and the attribute scores summed. The sum is likewise normalized. The standardization process assigns equal weights to all the indicators, allowing their use with rather different units. Most existing classifications of this type are very local and, due to geographic differences in physical and chemical conditions and different species composition of organisms in different regions, any global generalization of their use is impossible.

Due to the large number of water quality variables and their high variation, a water quality index may help to describe the overall situation (Thanh and Biswas, 1990). However, these authors also pointed out that the number of water quality indices used in different countries and in states of the United States are as high as the number of variables themselves. Several index and classification systems exist for lakes. They are very local, based on the specific conditions in a given country or region, and no generally accepted lake classification system exists.

Lake health

Most recently it became popular to speak about lake health as a global measure of lake environmental quality. The U.S. Environmental Protection Agency (EPA, 1998) defined a healthy lake as having clean water, optimum algae growth, adequate oxygen levels and abundance and diversity of fish and bottom-dwelling invertebrates. Further, natural aquatic plants should flourish in appropriate habitats, and bottom habitat should be uncontaminated. Unfortunately, this definition is very general, based on fuzzy and undefined terms like "clean", "balanced", "adequate", "diversity". As a result, it is not operational. In fact, previously developed lake classification systems are now used to express lake health. Xu et al. (1999) used the ecosystem health notion to specify the indicators for assessing freshwater ecosystem health. Biotic integrity is a similar notion, for which an index also was developed and critically evaluated (Simon and Lyons, 1995).

The Finnish watercourse classification system. This system (Heinonen and Herve, 1987) is intended equally for rivers and lakes (jointly called watercourses). The classification is utilization-specific, being directed to recreational purposes, raw water classification, and fishing. The watercourses are classified according to their suitability for respective uses according to a number of quantifiable indicators. Five classes are distinguished, including excellent, good, satisfactory, poor and bad. For some water uses, a sixth class was characterized as "unsuitable" for the respective use. A composite general classification is derived by the summarization of the individual classifications. The list of indicators and their values for each class is very long, and is not reproduced here.

Finnish system for deep lakes. This is a specific lake water quality system presented by Malin (1984). The variables included in the index were chosen on the basis of principal component and cluster analyses of data for 160 Finnish lakes observed once in the summer and once in the winter. For each variable, a value function (S) was constructed (Fig. 3.7). Six different variables were included: oxygen concentration, pH, conductivity, color, and manganese and total phosphorus concentrations. Based on the observed concentrations, the water quality value functions (q_i) were read from the graphs given in Figure 3.7. The lake water quality index (I) is calculated as the average value: $I = 1/6 \sum q_i$. The results of evaluating the value of I for the 160 lakes were compared with similar and earlier developed indices for Finnish deep-water lakes. They were found comparable, the advantage of this index being its ease of calculation. It is clear, however, that this index, particularly the value functions, has a local flavor and can only be used safely in geographically similar areas of the north temperate region, and possibly also the mountainous areas of some more southern localities like the alpine region.

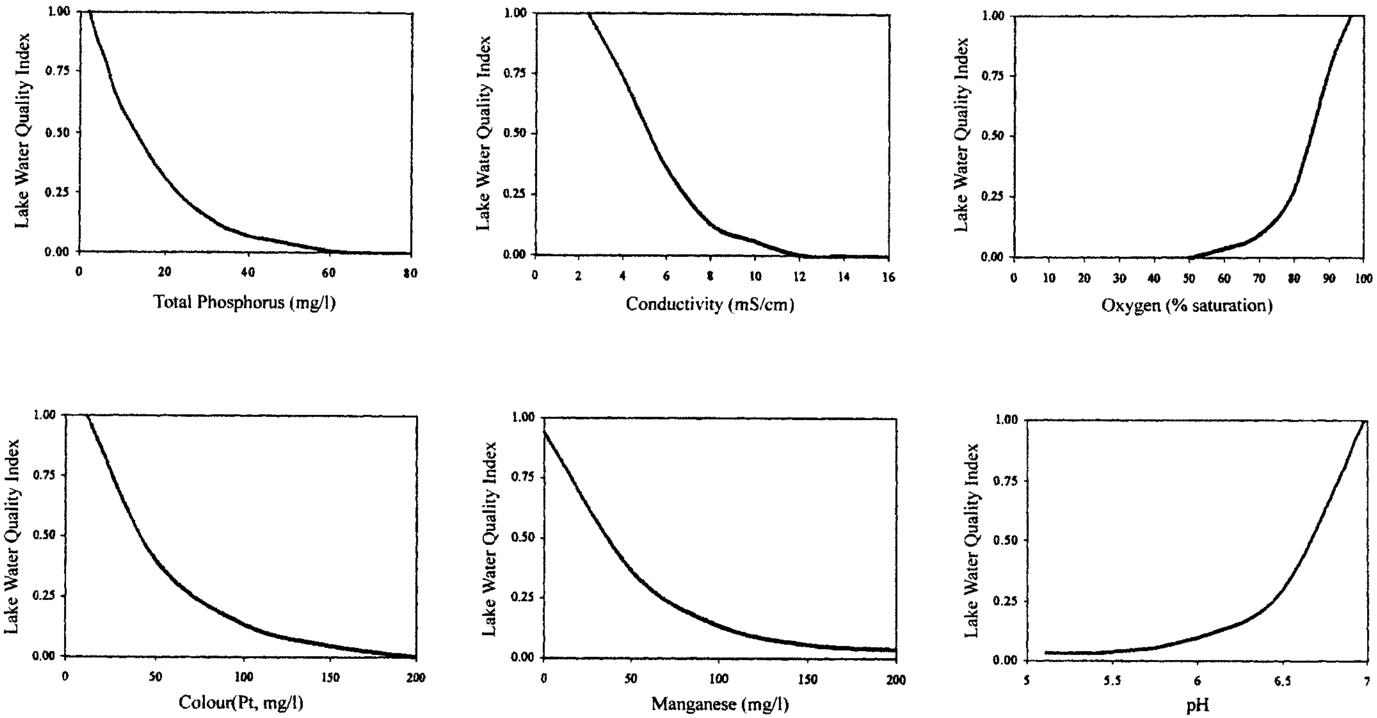

Fig. 3.7. Value functions for the General Lake Water Quality Index for Finnish deep-water lakes. The functions normalize the effects of each variable so that a value between 0 and 1 is achieved, with 1 for the best and 0 for the worst water quality. The effect is positive for some variables and negative for other variables. The functions represent an idealization of the effect of each variable and the normalized values can then be added (Malin, 1984).

Ohio Lake condition index. The system by Davic and DeShon (1989) is intended for lakes and reservoirs, being composed of 14 variables of physical, chemical and biological nature, as well as public perceptions of lake conditions. Physical and chemical measurements are made on a monthly basis from spring to fall, and biological measurements made once each year. The biological components include the fish index of biotic integrity, macrophytes, phytoplankton chlorophyll, fecal coliform bacteria, and fish tissue contamination. Classification into good, fair or poor conditions is based on comparing data against water quality standards or general criteria.

System of the Economic Commission for Europe for the ecological quality of lakes. This most broadly based system is intended for independent comparisons of the state of lakes throughout Europe, to the local conditions of lakes not affected by human activity. It uses five classes to enable sufficiently detailed comparisons to demonstrate time trends of improvement and deterioration. Basic groups of criteria are the oxygen regime, eutrophication indicators, acidification, metals, chlorinated micropollutants and other hazardous substances, and radioactivity. The system is summarized in Table 3.5.

3.3 LAKE SAMPLING GUIDELINES

The monitoring goals, the sources of pollution in a watershed, waterbody type, size and morphometry, and legislature and rules define a sampling program, while the available human and financial resources restrict it. Thus, a good monitoring schedule should optimize lake sampling efforts, with the goal of gaining the maximum possible relevant information. A number of general monitoring guideline documents exist (e.g., Chapman, 1996; Bartram and Balance, 1996). Several documents directed to monitoring lakes and reservoirs also exist (Thornton et al., 1980; Gaugush, 1987; Straškraba and Tundisi, 1999; Thornton et al., 1996; Rast and Thornton, 1999). Nevertheless, conditions are often so variable that sampling creativity is necessary. Further, new types of pollution and new measurement techniques continue to appear. Thus, this section discusses general sampling rules, rather than detailed sampling instructions. Wherever possible, however, specific guidelines are identified as well.

3.3.1 Selection of Variables

Section 3.2 identified a number of water quality variables which are helpful in evaluating lake and reservoir water quality. However, a selection of specific variables to be sampled must be made for each monitoring program, based on the goals of the investigation. Two major groups of programs can be distinguished: research investigations and routine monitoring programs. In both cases, the efforts can be extensive (aimed at comparing multiple waterbodies) or intensive (aimed at the operational control of an individual waterbody). The monitoring program depends completely on the questions to be answered. The goals of

Table 3.5. Economic Commission for Europe classification of the ecological quality for lakes (Premazzi and Chiaudani, 1992)

Variables	Class				
	I	II	III	IV	V
A. *Oxygen regime*					
Dissolved oxygen (DO) (%) in:					
epilimnion (stratified waters)	90–100	110–120	120–130	130–150	> 150
hypolimnion (stratified waters)	90–70	70–50	50–30	30–10	< 10
unstratified waters	90–70	70–50	50–30	30–10	< 10
Dissolved oxygen (DO) (mg l^{-1})	> 7	7–6	6–4	4–3	< 3
Chemical oxygen demand (COD) (mg O_2 l^{-1})	> 3	3–10	10–20	20–30	> 30
B. *Eutrophication*					
Total P (μg P l^{-1})	< 10	10–25	25–50	50–125	> 125
Total N (μg N l^{-1})	< 300	300–750	750–1000	1500–2500	> 2500
Chlorophyll (μg l^{-1})	< 2.5	2.5–10	10–30	30–110	> 110
C. *Acidification*					
pH (values < 9.0 only)	9.0–6.5	6.5–6.3	6.3–6.0	6.0–5.3	< 5.3
Alkalinity (mg CaCO l^{-1})	> 200	200–100	100–20	20–10	< 10
D. *Metals*					
Aluminium (μg l^{-1}; pH < 1.6)	< 1.6	1.6–3.2	3.2–5	5–75	> 75
Arsenic (μg l^{-1})	< 10	10–100	100–190	190–360	> 360
Cadmium (μg l^{-1})	< 0.07	0.07–0.53	0.53–1.1	1.1–3.9	> 3.9
Chromium VI (μg l^{-1})	< 1	1–6	6–11	11–16	> 16
Copper (μg l^{-1})	< 2	2–7	7–12	12–18	> 18
Lead (μg l^{-1})	< 0.1	0.1–1.6	1.6–3.2	3.2–82	> 82
Mercury (μg l^{-1})	< 0.003	0.003–0.007	0.007–0.012	0.012–2.4	> 2.4
Nickel (μg l^{-1})	< 15	15–87	87–160	160–1400	> 1400
Zinc (μg l^{-1})	< 45	45–77	77–100	110–120	> 120
E. *Chlorinated micropollutants and other hazardous substances*					
Dieldrin (μg l^{-1})	0	n.a.	< 0.0019	0.0019–2.5	> 2.5
DDT and metabolites (μg l^{-1})	0	n.a.	< 0.001	0.001–1.1	> 1.1
Endrin (μg l^{-1})	0	n.a.	< 0.0023	0.023–0.18	> 0.18
Heptachlor (μg l^{-1})	0	n.a.	< 0.0038	0.0038–0.52	> 0.52
Lindane (μg l^{-1})	0	n.a.	< 0.08	0.08–2.0	> 2.0
Pentachlorophenol (μg l^{-1})	0	n.a.	< 13	13–20	> 20
PCBs (μg l^{-1})	0	n.a.	< 0.014	0.014–2.0	> 2.0
F. *Radioactivity*					
Gross-activity (mBq l^{-1})	< 50	50–100	100–500	500–2500	> 2500
Gross-activity (mBq l^{-1})	< 200	200–500	500–1000	1000–2500	> 2500

operational control are directed to the use of an individual waterbody, whether it is a natural lake from which water is withdrawn for a particular purpose, or a reservoir constructed for that purpose. There also are no strict overall rules in this case. Local legislation, the types of pollution, citizen requirements, and available manpower and resources play a role as well. Nevertheless, it is possible to suggest a framework for most lake sampling efforts.

Table 3.6 provides a suggested framework. In all instances, the main sample site location is situated either in the center of a lake, or near the dam of a reservoir (in the case of no specific outlet site) or else in the place in the waterbody from which the water is withdrawn. For drinking water supply reservoirs, the withdrawal of water from specific water layers, which can result in a short-circuit effect in the movement of water through the reservoir, must be taken into account. In waterbodies used for power generation, the internal water currents produced by the irregularities of the water flowing out (possible also for reservoir systems) also must be considered.

Every waterbody is a system, with the most important physical components represented by the inflow, the main waterbody and the outflow/outlet. To develop appropriate management options, it may be necessary to monitor all three components. Indeed, it would be difficult to develop recommendations for improving, or at least maintaining, water quality without knowing the causes of the observed water quality changes, with such causes typically contained in the inflows to the lake from the watershed.

Table 3.6. Suggested framework for monitoring waterbodies used for different principal purposes. Modified from Chapman (1992, 1996)

Principal water use	Sampling frequency	Physical and chemical variables	Biological variables
Potable water supply	Continuous, daily to weekly	Temperature, Secchi disc, oxygen, color, turbidity, phosphorus, pH, odor, suspended solids, organics, metals, nitrogen	Coliforms, pathogens, phytoplankton, chlorophyll-a
Industrial water supply	Continuous, daily to weekly	Temperature, pH, hardness, dissolved and suspended solids, major ions	Pathogens
Power generation	Daily to weekly	Temperature, oxygen, conductivity, dissolved and suspended solids, major ions	
Irrigation supply	Weekly to monthly	pH, total dissolved solids, sodium, chloride, magnesium, nutrients	Faecal coliforms
Fisheries and recreation	Weekly to monthly	Temperature, suspended solids, Secchi disc, phosphorus and nitrogen, oxygen, ammonia, pesticides	Phytoplankton, chlorophyll-a, zooplankton
Flood control	Monthly to annual	Suspended solids, turbidity, Secchi disc	

3.3.2 Selection of Sampling Stations, Depths and Timing

Sampling logistics depend on the goals of the respective investigation. If the investigation is only extensive, one or a few stations sampled once a year or during characteristic seasons may be adequate. For systematic investigations, more stations, depths and shorter intervals between sampling dates are necessary. In most cases, the number of horizontal and vertical sampling sites will typically be a compromise between the costs in time, money and manpower and the sampling objectives, bearing in mind the size, seasonality and thermal structure of the waterbody being sampled. It is well known that the information gained in a lake sampling program does not significantly increase above a certain number of samples. Rather, sampling representativeness is a matter of coupling spatial and temporal scales.

Sample elaboration is the most expensive, time-consuming process of monitoring schemes. It is advisable, therefore, to use *integrated samples* to obtain averaged values from one or a low number of sample analyses. A distinction is made between depth-integrated samples and layer-integrated samples. A depth-integrated sample is obtained by taking water, continuously or stepwise, between the surface and bottom of a waterbody (or from a certain depth range). The individual samples obtained in this manner are then mixed to produce a mixed sample. A layer-integrated sample is obtained by mixing samples from a particular layer at various points of the waterbody. Tube samplers, for example, allow investigators to get a mixed sample covering the mixed (productive) zone at a cross-profile of a longitudinal lake or reservoir. The integrated samples give a more representative figure than individual samples, particularly when there is no obvious regularity in the vertical or horizontal direction in a waterbody.

Sampling spots and depths of sampling

The most completely monitored station in a lake or reservoir should be located where the water is withdrawn for specific uses, or where other interests are directed. To make accurate assessments for management purposes, it is necessary to sample the inflow(s), the main body, and the water diverted to major users or flowing out of the lake/reservoir. The sampling localities on the inflows are generally selected on the basis of their importance regarding water quality differences. Any tributary sampling site must be located above the potential maximum flooding sites and below the last pollutant point source. The sample collection should comprise the entire inflow tributary cross-section, and the water flow rate at the time of sampling must be determined. Water flowing directly into a waterbody from a major pollutant source must be sampled separately.

The number and spacing of vertical sampling depths depends on the lake/reservoir depth, and its degree of thermal stratification. The waterbody "surface" is sampled at a 20–30 cm depth under the water surface. It should not be sampled near the shoreline because of possible accumulation of wind-blown scum and other nearshore effects. The "bottom" is sampled at a depth of 1–2 m above the bottom. In a deep stratified waterbody, individual vertical sampling depths should not be further apart than 10 meters, at least for the top 100 meters of the water column. A detailed temperature profile, using an electric thermometer, is a reliable means of determining the spacing of individual depths.

Sampling frequency
Generally speaking, the more frequently samples are taken, the more representative and reliable are the results and subsequent conclusions. Monthly intervals may give good comparative results. The highest desirable frequency of manual sampling is weekly. Monthly sampling may result in strong bias in the data, particularly if the effects of precipitation or other discharge effects (usually short-lasting) are high. Further, samples should always be taken at a fixed time during the day.

Sampling and measurements in lakes and reservoirs must be done at regular intervals to obtain a reliable picture of changes in water quality. Regular intervals also allow for an easier statistical evaluation (e.g., the calculation of "time-representative" averages) and data summarization (e.g., trend calculation, including seasonal components). The detection of trends is easier with regular sampling (Gaugush, 1987). For inflowing streams, riverine lakes and rapidly flushed reservoirs, however, regular sampling has the drawback that flood situations and low flow situations of shorter duration often are not covered. This can result in an underestimation of the pollutant load, because these situations often result in higher pollutant loads than for normal flows. Methods for making accurate estimates in such situations are available (e.g., Kortmann, 1984). An event-oriented sampling program directed to covering floods and other irregular events has an advantage when irregular events are of crucial importance for the state of the lake (Longabucco and Rafferty, 1998).

A summary of methods of detecting trends in water quality is given by Esterby (1997), with different trends in freshwater discussed Peters et al. (1997).

3.3.3 *Automatic Monitoring*

Two types of automatic sampling can be distinguished:
- Depth recording, in which measurements are made during the lowering of a probe through different depths,
- Recording over time.

In some instances, these two approaches can be combined.

Many types of recording instruments for both methods are available. Data storage is often done automatically with a special storing device (data logger), or by connecting the instrument to a computer. The second alternative has the advantage that data processing is immediately possible, in the form of both statistical elaboration techniques (recalculations for selected units, averages, integral over some periods, etc.) and graphical representations. Data from specialized loggers have a limited range of built-in data elaboration, and the data must first be put into a computer for a complete analysis. The advantage of data loggers is that they are more rigorous, less sensitive to weather conditions, and require little energy. Some loggers can even record several water quality parameters over hourly or even shorter intervals, and remain unattended for months.

The sensitivity of modern electronic instruments is usually high. However, their accuracy is highly dependent on the accuracy of their calibration. Automatic instruments used without careful calibration can easily give erroneous results. Although many recently-developed instruments have built-in procedures for their automatic calibration, the need for a periodic check of their calibration, using classical chemical or other determination methods, is still stressed. Calibration for extreme low and high water quality values results in the highest accuracy.

Two basic kinds of data recording can be obtained; namely analog and digital. An analog record (in the form of a time or depth graph) is very useful for rapid orientation regarding water quality changes. A digital record provides more exact individual values. The storage of digital values and their simultaneous presentation in an analog form is the optimum combination. Instantaneous values of signals with high frequency changes are relatively useless, and it is advisable to use sampling integrals over specific periods. For example, wind speed can vary significantly within seconds, and integral (averages) over some time interval provides more reliable information.

Depth profiling instruments can be classified into two types:

- Those with slow reaction times, which are lowered to a certain depth in a lake and the record started manually,
- Free-falling probes, capable of automatically recording through the water profile with high depth resolution (up to millimeters).

It is now possible to continuously record algal concentrations in a waterbody, using a fluorimeter (e.g., as from Turner Designs). Although the recorded values are relative, they closely correlate with the classical chlorophyll-a measurements.

3.3.4 Remote Sensing

Remote sensing is a useful monitoring technique for large waterbodies. It is normally obtained as satellite imaginary or aerial pictures. Both ways can be combined and different resolution and scale coverage obtained by each method. Present methods of evaluating satellite pictures are more relative, rather than producing absolute quantitative values. Thus, relating satellite images or aerial pictures to ground observations is essential. To be successful, tools for monitoring water quality, both high altitude (satellite) or low altitude (airplane) images, must be calibrated permanently to the results of classical sampling and/or monitoring. The variables useful for determining the relevant management characteristics most easily measured with remote sensing techniques include chlorophyll-a and suspended material concentrations, turbidity and temperature. Recently, attempts are being made to distinguish not only the horizontal distribution of these variables, but also their depth distribution.

The use of satellite and aerial imaginary helps in developing water quality management decisions by providing the following information (Fischer et al., 1991):

- The distribution of temperatures at the surface of a lake,
- The horizontal distribution of chlorophyll-a and the locations of low and high phytoplankton concentrations, and the horizontal distribution of Cyanobacteria blooms (Jupp et al., 1994),
- Areas of high and low turbidity,

- The horizontal displacement of river plumes carrying pollutants and suspended material into a waterbody,
- Areas of suspended inorganic and organic matter concentrations,
- The location of mass fish kills in a lake,
- The distribution of total phosphorus and nitrogen can be depicted for a specific lake, with careful analyses and correlation with other variables (e.g., chlorophyll-a, temperature and transparency or adsorption coefficients),
- The detection of mean or high macrophyte concentrations, and areas of severe plant stress,
- The delineation of groundwater flow into lakes,
- Attempts to simultaneously estimate total phosphorus, total inorganic nitrogen, chlorophyll-a and suspended solids concentrations, Secchi disc transparency, turbidity and water temperature also have been made (e.g., Vande Castle et al., 1988).

3.4 HOW TO EVALUATE WATER QUALITY MONITORING DATA

This section focuses on basic data storage and handling, and particularly on the evaluation of measurements of several important water quality variables. Attention is given to the most commonly-measured variables: temperature, water transparency, biochemical oxygen demand (BOD), and dissolved oxygen and chlorophyll-a concentrations.

3.4.1 Data Storage and Handling

Before data obtained from measuring individual water quality variables with individual samples can be used as a basis for further assessments, they must be processed in some manner. The first step is to enter the data records into a software database. One basic data processing technique is to calculate annual or seasonal averages, and to determine minimal and maximal data values. Temporal trends also are determined, in order to evaluate whether the water quality is improving or deteriorating. Factors complicating the evaluation of water quality changes include seasonal variability and their dependence on the water flow rate. Water quality changes arising from the effects of water storage, often measured as the difference between the quality of the inflowing and the outflowing or diverted waters, are affected by thermal stratification and other in-lake processes that cause horizontal water quality gradients. Such gradients are particularly strong along the length of reservoirs (Thornton et al., 1996; Rast and Straškraba, 2000), particularly if the reservoirs have water retention times within the critical range of their theoretical retention time (see Chapter 6). The assessment of water quality changes is facilitated by estimating the budget of chemical substances (i.e., differences between their inflow and outflow quantities). The inflow/outflow difference is affected by long-term trends, as well as flow rates and stratification within the waterbody.

Water quantity and quality data in the inflow, outflow and the waterbody body are typically stored. Modern PC spreadsheets and databases (e.g., EXCEL, PARADOX, FRAMEWORK; see Neethling (1986) for a review) are best suited for this purpose, containing rows that can represent water quality variables, and columns that can represent the dates they were measured. Special tables can be prepared for the stratification of each variable, with rows representing individual depths and columns representing sampling dates. All the depth measurements are recorded, even if they were only measured once in a given period. When a depth is not measured, the corresponding cell in the spreadsheet or database should be left blank. The advantages of this storage method are the ease of calculation of basic statistics, the pseudographic expression of results, and the ease of data transfer for their use in graphic representations and statistics software packages.

The evaluation of monitored water quality data can be done with several types of data elaboration methods:

- *Characterization of the type of statistical distribution of individual water quality variables.* Water quality data often are not normally distributed. A more common situation is a log-normal distribution (i.e., a normal distribution of the data after they have been transformed to logarithms). This is important if it is desirable to make predictions on the basis of statistical data analysis. A normal data distribution is a prerequisite for using a number of statistical techniques, with a different handling of data (e.g., using their logarithms) or different evaluation of the results obtained, being a function of this characteristic. This situation can be treated in a general form in any statistical analysis program.
- *Determination of trends.* If observations are recorded over a number of years, it is possible to predict, by means of trend analysis, if an increase or decrease in the value of specific water quality variables is taking place and at what rate.
- *Correlation between variables.* For this analysis, one variable is plotted against a second variable, and a known type of curve fitted to the data by means of nonlinear regression. The concentrations in the inflows to a waterbody, for example, often show strong correlations with water flow. Knowledge of the concentration/flow relation can be used to predict the load to a waterbody in its inflows for which recording does not exist, or from more detailed flow-weighted load estimates. A clear relation with flow makes it possible to judge the basic character of many pollution sources (i.e., point and diffuse sources; Fig. 3.8). A correlation with temperature indicates periodicity, and direct or indirect dependence on biological processes in a waterbody or watershed. Although the use of easy, shortcut methods of linear regression is very popular, the nonlinear character of many relations among water quality variables often necessitates the use of nonlinear regression techniques.
- *Multiple regression and methods of multivariate statistics* are popular methods of data treatment, although they have many pitfalls for water quality evaluation. The most important drawback is that the strict statistical requirements for such analysis (e.g., normal distribution pattern of all variables) are often not met. Very uneven values, often very

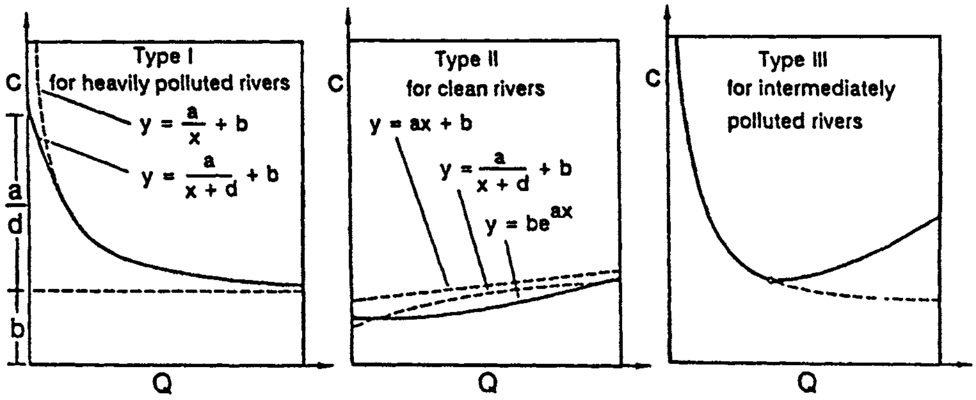

Fig. 3.8. Types of dependence of water quality variables on flow. Pollution from point sources is diluted by an increased flow, while pollution from diffuse sources increases as more material is washed by the precipitation from the soil and surroundings.

narrow ranges of some variables, and nonlinear dependencies among variables are the primary causes.

- *Spectral characteristics* of the data by means of spectral analysis (i.e., time series analysis) indicates the character of the periodic nature of a given water quality variable. As an example, temperature values usually exhibit a pronounced annual and daily periodicity, as well as some less pronounced several-day periods. Methods for this type of data analysis are summarized by Box and Jenkins (1970).
- *Cross-spectral analysis* of several variables allows detection of time lags between water quality changes.
- Creation of *statistical input–output models* for identifying interrelations between measured variables, which can then be used to predict the future conditions of water systems.
- Use of data for *dynamic models* allows the provision of in-lake estimates of certain variables. Most determinations of dynamic system parameters are biased by the experimental technique. A water subsystem of interest is taken out of the whole systems context, for example, and its conditions changed by enclosure of the water volume in an experimental set-up. This approach does not take into account the rapid adaption of aquatic organisms. Thus, the results may produce information about the events in the experiment, rather than in the waterbody, because different conditions may develop in the former as the result of confinement and isolation from the full system interactions.

Another tool for the elaboration of extensive spatial data is *Geographical Information Systems* (GIS), which are proving very useful for data concerning whole watersheds.

Chapman (1996) provides more information on water quality data handling and presentation. Simplified procedures for assessing and predicting lake eutrophication are provided with several models developed by W.W. Walker.

Evaluation of stratified water quality variables by the shapes of depth profiles

Stratification of some water quality variables occurs because of many processes, and individual types of depth profiles can be distinguished by the following characteristics:

(A) A marked change in the upper (productive) water layers—this type of curve usually denotes the effects of biological production processes in the surface water layers.

(B) A marked change at the lake bottom—this type of curve indicates the effects of decomposition. The cause of diminished oxygen concentrations, for example, or enhanced carbon dioxide concentration can be direct or indirect. The indirect cause for the state of a number of water quality variables is an anaerobic (oxygen depleted) environment at the bottom.

(C) A pronounced change in the middle depths—this type of curve indicates either the effects of decomposition in the metalimnion (i.e., metalimnetic minima), or the production of certain types of algae in the metalimnion, which is common in the *Oscillatoria* (i.e., *Planktothrix*) lakes (i.e., metalimnetic maxima). It can be caused in reservoirs the inflow of water layers with different concentrations of water quality variables. Processes at the bottom in the upper, shallower region of the reservoir, and spreading of the resulting changes down the length of a reservoir, also may be responsible.

Evaluation of stratified variables by depth–time isoline charts

There are a number of possibilities for presenting and evaluating or annual changes in temperature stratification (or stratification of any other variable) in a waterbody. Figure 3.9A illustrates the same data expressed in three ways: (i) in the form of an isoline depth–time map, (ii) in the form of temperature profiles at given dates, and (iii) as periodic curves corresponding to temperature changes at particular depths. Another possibility is to present the observed depth profiles in a three-dimensional graph (Fig. 3.9B). The goal of any representation is to give transparent, condensed figures that enable easy evaluation and comparison between the stratification of different water quality variables, between sampling spots within the waterbody, between years and between lakes. The seasonal trend on the surface of a lake, or at certain depths, also can be represented by isolines (Fig. 3.9C). Several PC programs are useful for computer-generation of such graphs. For isoline charts, for example, one program with broad evaluation capabilities is GRAPHER by Golden Software Inc.

Isoline charts are probably the most concise representation, as they clearly show the depths and periods of high and low values. The depths at which concentration changes are most pronounced are indicated by the highest density of isolines. The maximal and minimal values, resulting in vertical profiles of type (C) from the previous subsection, will appear as closed areas. The course of stratification in a given lake and day or year can then be evaluated by:

- The periods of circulation,
- The density of isolines in different layers,
- A decrease or increase in water quality values on the surface or at the bottom of a lake—the time is evaluated at which a certain isoline intersects the bottom or surface of a waterbody, and the depth to which it reaches,

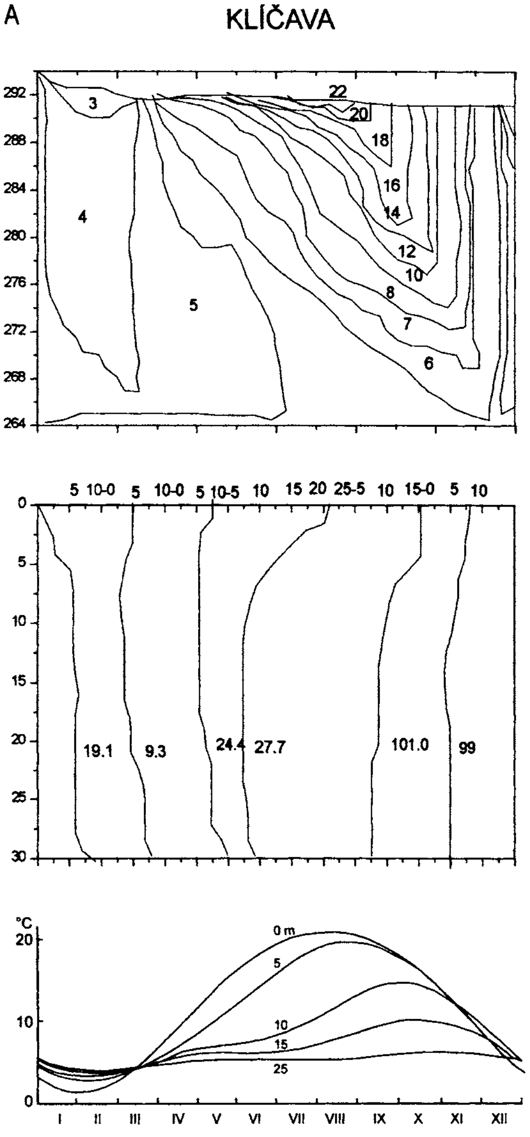

Fig. 3.9. Different ways of representing the changes of depth–time distribution or space–time distribution of daily or annual changes of temperature and other variables in lakes. A—Three ways of presenting annual changes of temperature at different depths. The above graphs represent isoline chart, the presentation of profiles and the periodic changes of temperature at particular depths, respectively.

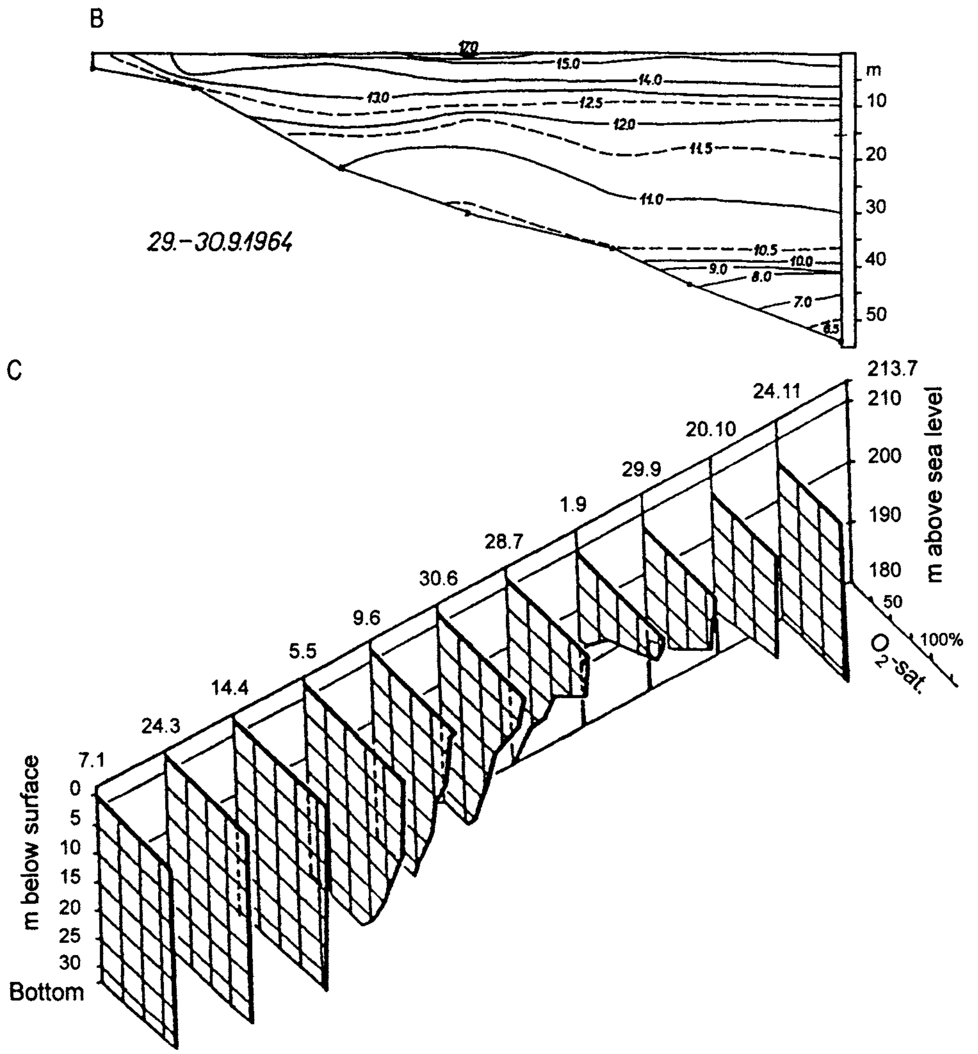

Fig. 3.9 (*continued*). B—Horizontal distribution during one sampling; C—representation of profiles in a three-dimensional form. When there is much over-crossing of the profiles, the graph becomes much less transparent.

- Integrals of the area of certain isolines—the extent of the depth–time areas of selected isolines that intersect twice the surface or the bottom, or are closed, can be evaluated numerically or with planimetry.

Evaluation of horizontal concentration differences
Pronounced, irregular horizontal differences in concentrations occur in shallow waters as a result of differences among individual tributaries, and differences in the intensity of processes in various parts of a waterbody. The irregularity of the horizontal distribution in concentrations in shallow waters is due to the domination of wind effects. Wind drifting of water masses, internal seiches, blowing of surface waters with their constituent plankton, and accumulation of scums and their associated decomposition can produce local changes, which can rapidly move when wind speed and/or direction change.

In deep, stratified reservoirs, horizontal distinctions are regular, particularly in the direction from the main inflowing stream to the dam. The inflow zone shows the greatest difference (Section 6.1.3—*Longitudinal Differentiation*). Thornton et al. (1982) describe a method for evaluating horizontal differences based on the analysis of variance.

Evaluation of water quality trends
A water quality trend typically describes a systematic directional change in water quality values over a long period of time. Long-term trends can be identified with some degree of certainty only when the data extend over a period of at least ten years. Simple functions of time can then be used to express the trend(s).

The existence of long-term trends can be determined by calculating the annual averages (i.e., to extract seasonal trends) with a parametric or nonparametric test. The nonparametric Spearman's and Kendall's test for calculation of the coefficients of rank correlation of annual means of time is very useful for this purpose. A useful parametric test is Sach's test, for the calculation of the sum of squares of differences between successive values of annual mean values (e.g., Conover, 1980; Sokal and Rohlf, 1981; Taylor, 1990; Helsel and Hirsch, 1992).

When a time series analysis of date defines a seasonal component (fluctuation with a one-year period), the periodicity can be calculated simultaneously with trends. Box and Jenkins and Hipel and Fang provide some calculation possibilities. Gaugush (1987) also gives practical hints for detecting water quality trends.

A necessary condition for making reasonable predictions, however, is the homogeneity of the water quality data. Simply stated, this means that no major changes occur suddenly in a respective waterbody or its watershed. It would be pointless to estimate any simple statistical monotonical trends, for example, if a factory that caused major water quality impacts was constructed in a watershed, or if a new pollution management technology was applied in a given case, during the period that water quality data was being collected.

A water quality trend may be random (i.e., no clear trends), linear, exponential, cyclic or stepwise. Combinations of all types also are possible. An example of water quality trends in a lake is illustrated in Figure 3.10. Exponential water quality trends that increase rapidly are often found. They can be very problematic, in that they can start innocently, but then lead rapidly to adverse water quality conditions. Moreover, it is difficult to identify the starting phase of an exponential trend from analysis of a linear trend. Extrapolation of a linear or an exponential trend can lead to very different conclusions. Values encountered

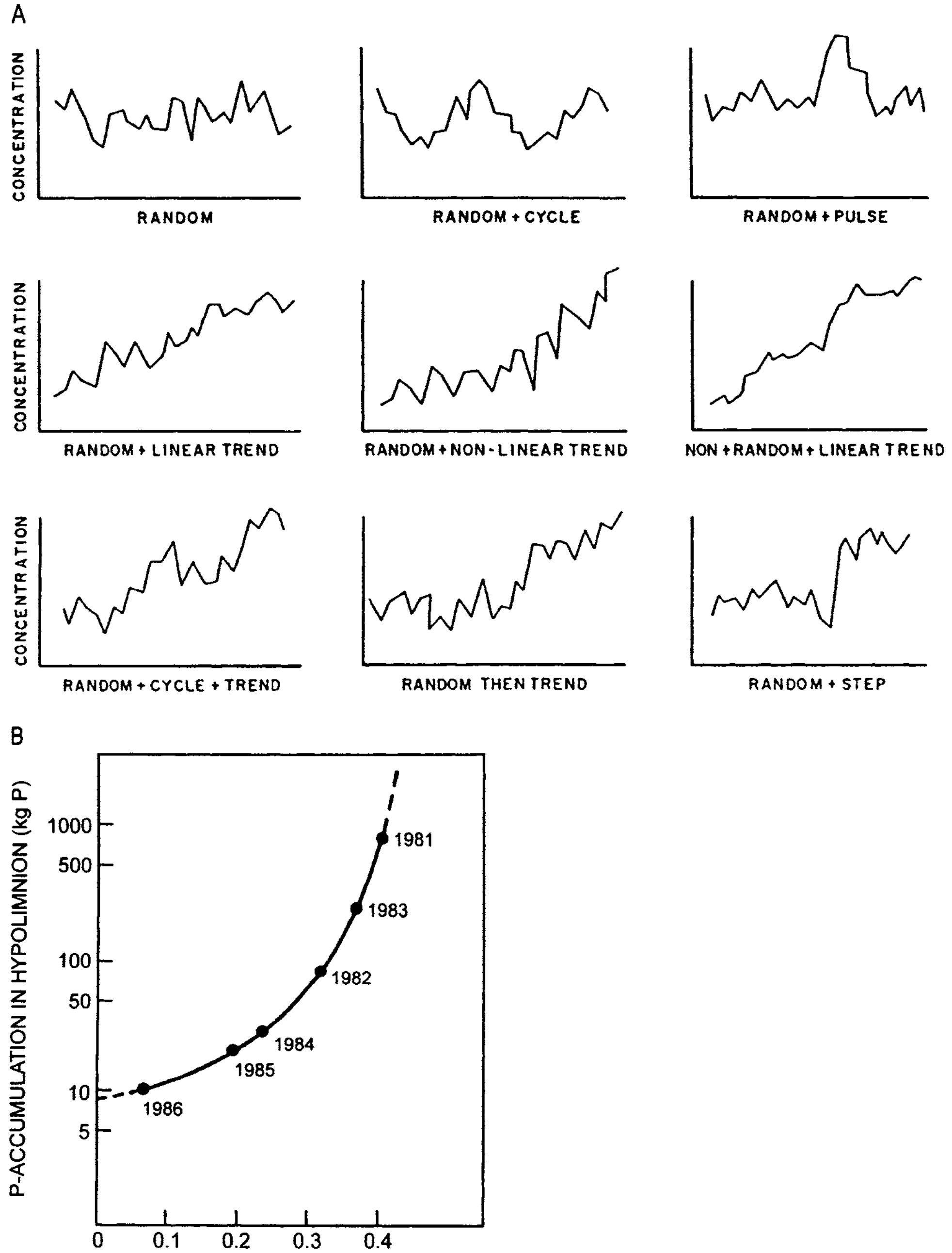

Fig. 3.10. Different types of trends of water quality variables. A—schematic representation of different possible types of trends; B—An example of a typical water quality trend.

within a few years after a water quality trend is established can greatly exceed predictions based on a linear trend. If a time series analysis is conducted on data for only a short period, however, it is difficult to be certain a suspected exponential trend is not simply a component of a slower periodic change.

3.4.2 Guidelines for Water Quality Evaluation

The following evaluations are based largely on data from temperate regions, as presented in Straškraba et al. (1993) for reservoirs. Whenever possible, consideration also was given to tropical conditions, as illustrated in the guidelines provided by Thornton et al. (1996). In contrast to an all-encompassing lake classification scheme, this section highlights the strength of classifying lakes on the basis of separate consideration of individual variables. This is primarily because, for different waterbody uses and specific conditions, different water quality variables can assume differing importance (i.e., be weighted differently). More elaboration is needed, rather than just attempting to classify a waterbody on the basis of a single prescribed formula. However, this approach is much more useful for lake management purposes.

The degree of lake stratification
To compare the extent of stratification of different water quality variables in a waterbody, it is desirable to use a quantitative measure. The degree of stratification of a variable can be expressed by the coefficient of the distinctness of stratification (*CDS*), which can be calculated as: $CDS = x_{max}/x_{min}$, where x_{max} is the maximum concentration of the respective variable in the depth profile and x_{min} the minimum concentration.

This value is independent of the absolute values. However, it depends on the units used for the variable, because the total variation for a given variable is a primary factor. If the numerical values of the variable in the units used are low (e.g., units or fractions), a larger *CDS* value is necessary to indicate that the variable is significantly stratified than if the variable has larger values (e.g., tens or hundreds). This difficulty can be overcome by considering different critical values. Table 3.7 contains critical values for water quality parameters with a possible range of variation below 10 (e.g., pH, alkalinity) and above 10 (e.g., most chemical and biological variables). It illustrates approximate critical values for individual degrees of distinctness of stratification. One can differentiate between *CDS* for a given vertical profile, CDS_{max} during maximal stratification, average CDS_S for the summer (in temperate regions) or dry (in tropical regions) portion of the year, and annual CDS_{aver}. Characterization of a longer period by means of average values can be misleading, because stratification may be of completely different degree over different periods.

The two groups of water quality variables that can be characterized by the distinctness of their stratification (*CDS*) in deep lakes/reservoirs are as follows:
(A) Markedly and consistently stratified,
(B) Nonstratified or only temporarily stratified.

Table 3.7. Evaluation based on critical value of the coefficient of distinctness of stratification (CDS_{crit}) (Straškraba et al., 1993)

For the range of variation of the respective variable, CDS_{crit}		Stratification
< 10	$\geqslant 10$	
$\geqslant 1.5$	$\geqslant 1.2$	Stratification conspicuous
1.1–1.5	1.05–1.2	Nonstratified or indistinctly stratified

As discussed above, this kind of characterization evidently does not express the shape of the profiles.

Evaluation of water transparency

The combined effect of water color (due to dissolved substances), mineral turbidity, and the presence of algae is responsible for water transparency, as measured by the Secchi disc (SD). Waters in peat bog regions, or waters polluted by paper mills, are typically very colored. The color usually does not change significantly over the course of a year, although some changes can occur because of bleaching effects in the summer, or because of more intensive paper production activities. In contrast, a pronounced seasonal variation is characteristic of algal turbidity in temperate and subarctic lakes, with algal occurrence in a waterbody being minimal in autumn to early spring and culminating in a short time period. In contrast, in the tropics and dry regions, it can persist for long periods because of finer-sized soil particles entering a waterbody.

The evaluation objective is to classify waterbodies on the basis of their transparency (Table 3.8) and to estimate the degree of eutrophication. Water transparency differences between early spring (low algae) and spring–summer (high algae) are indicative of the degree of eutrophication in situations where the turbidity due to factors other than algae (i.e., non-algal turbidity) is low, or remains constant (Table 3.9). The estimate is approximate, because the difference in water transparency during the periods characterized by low and high algae content may have other causes as well. Further, in the tropics there can be large quantities of algae even in periods of their minimal occurrence in a given waterbody. If the transparency has been measured systematically over time, only the measurements that do not include significant flood-related turbidity are used to assess the trophic condition of a lake.

Organic matter

For surface waters, only a small proportion of the organic compounds determined by measuring the chemical oxygen demand (COD) is easily-decomposed biologically. A standard biochemical oxygen demand test conducted over a five-day period (BOD_5) is typically used to estimate the decomposable organic compounds in a waterbody. The proportion of the easily-degraded compounds (BOD_5) in a COD_{Cr} test usually ranges between 0.10–0.15. Values lower than 0.10 are found in waters with high contents of slowly decomposing organic substances (humic substances). Values over 0.15 indicate either the presence of large quantities of algae or recent water pollution. With increasing flow rates, the proportion of

Table 3.8. Classification of waters by transparency as measured by Secchi disc (meters) (Straškraba et al., 1993)

Secchi disc transparency during period of low algae concentrations	Reservoir
< 0.3	Not transparent
1–2	Poor transparency
3–6	Semi-transparent
> 6	Clear

Table 3.9. An estimate of the effects of eutrophication, based on changes in transparency over the course of a year, between the periods of low and high amounts of algae (Straškraba et al., 1993)

Difference in transparency (m)	Effects of eutrophication
0–0.5	None
0.5–1	Slight
1–3	Medium
> 3	Strong

particulate organic substances increases and may become higher than that of the dissolved substances. This depends largely on the erosion in the watershed, and the resulting quantity of sediments in the inflowing streams. Although organic compounds resistant to decomposition do not significantly affect the oxygen regime in a waterbody, they may spoil the taste or smell of treated water used for drinking water, and they may be precursors of haloforms and other mutagenic organochlorides resulting from the disinfection of the water with chlorination.

When algae in a waterbody are abundant, the BOD_5 value indicates not only the quantities of dissolved organic compounds, but also can be affected by algal respiration and decomposition. If it is determined that this factor might interfere with the evaluation of a waterbody, a correction to the calculations can be made on the basis of the chlorophyll-a content (Straškrabová et al., 1993), with 1 milligram of chlorophyll-a equal to 0.025 milligrams of BOD_5.

Water color depends in part on the contents and character of organic compounds. Unpolluted surface water is colored mainly by humins and ferro-compounds. Other undissolved and dissolved compounds from in-lake processes (autochthonous sources) result from the production of phytoplankton and other organisms (e.g., bacteria). The autochthonous portion of the total supply of organic compounds to a reservoir increases with increasing eutrophication and with a prolonged water retention time (i.e., longer flushing time; Straškrabová, 1975). The total BOD_5 load to a waterbody consists of the inflow load and the load resulting from the primary production of organic matter by phytoplankton in the waterbody. The primary production can be converted to BOD_5, with 0.5 of gross primary production (expressed as oxygen) equal to the BOD_5. The percent reduction

Table 3.10. Percent reduction in biochemical oxygen demand (BOD_5) of total load (average values for the warmest half of the year) in shallow and deep reservoirs at different water retention times (z = average depth, expressed in meters; RT = water retention time, expressed in days) (Straškrabová, 1975)

RT	$z < 7$	$z > 7$
0.5	15	20
1	40	35
5	60	45
10	80	55
50	75	70
100	75	70
200	70	75

of BOD_5 of the total load depends on the water retention time and depth of the waterbody (Table 3.10).

Oxygen

Critical oxygen concentrations in the hypolimnion of a waterbody, indicating the suitability of the water as a drinking water supply, are given in Table 3.11.

Anoxia of bottom waters can be prevented by reducing eutrophication. Metalimnetic oxygen minima can occur in waterbodies with sharp thermoclines and high primary production. This situation can produce water quality deterioration in the middle water layers. The occurrence of metalimnetic oxygen minima also can be prevented by reducing eutrophication.

The occurrence of anaerobic zones in the upstream, inflow portion of a reservoir usually indicates an oversupply of biologically-decomposable particulate organic compounds from the watershed.

Phosphorus

Phosphorus is the element most frequently found to limit primary production. In unlimited of phytoplankton populations, the nitrogen : phosphorus atomic ratio (N : P) of the algal biomass is typically taken as approximately 16 : 1. Rast and Ryding (1989) provide a means of converting the N : P atomic ratio to their corresponding concentrations in waterbodies, thereby allowing an easy comparison of the ratio of their in-lake concentrations in identifying the limiting nutrient for algal biomass. However calculated, higher N : P ratios indicate the maximum algal biomass is usually controlled or limited by the level of phosphorus in the waterbody.

The only complicating factor in this determination is consideration of the chemical forms of the phosphorus. The biological availability of different chemical forms of phosphorus to algal varies. The orthophosphate or dissolved reactive forms of phosphorus are typically the forms most readily available for immediate uptake and use by algae in a waterbody. Soils can retain phosphorus as long as soil particles are not washed away. In the tropics, the phosphorus bound to soil particles represents a large proportion of the

Table 3.11. Indication of the suitability of lakes and reservoirs for drinking water supply, based on critical oxygen concentration in the hypolimnion (modified from Dillon and Rigler, 1975)

Oxygen concentration (mg l^{-1})	Suitability of reservoir
> 5	Excellent
< 5	Suitable
< 2 briefly	Not very suitable
< 2 for a long time	Unsuitable

phosphorus in waterbodies. Thus, a distinction must be made between particulate and soluble phosphorus (both as total phosphorus or as orthophosphate). The determination of the soluble inorganic orthophosphate ($\sim$ reactive phosphorus) is very sensitive to changes during its transport to a laboratory for analysis (microbial activity, uptake by organisms in the water sample, adsorption on particles settled at the bottom of the sample bottle, rupture of sensitive algae, release from dead organisms, etc.). Thus, most values of dissolved reactive phosphorus determined by standard laboratory procedures are very uncertain. The most accurate phosphorus determination is made after the sample is filtered through a fiberglass filter when it is collected, and the sample is analyzed within two hours of its collection, also assuming it is protected from sun, heat and frost. In hard-water lakes, the phosphorus concentrations are typically less than in soft-water lakes of otherwise similar conditions, due to its co-precipitation in hard-water lakes with calcite (see also Section 3.2.1, with consequences shown below in the *Chlorophyll*-section).

In industrialized countries, a significant contribution to the phosphorus loads to some lakes may result from atmospheric inputs (e.g., precipitation). Data about this source, however, are limited for most places.

Chlorophyll-a

Critical limits of chlorophyll-a (CHA) for delineating the trophic state of lakes, and the adequacy of the water for treatment, are given in Table 3.12.

The *CHA* : *TP* ratio can be used to predict whether water quality can be improved by reducing the external phosphorus load or, alternatively, what the consequences of its increase will be. The relation between the two parameters is usually considered to be exponential (expressed by a power function), increasing rapidly at low concentrations (Dillon and Rigler, 1974). However, when the total phosphorus concentration reaches a certain level, the CHA concentration will stop increasing, and the correlation will be sigmoid over a whole range of values (Straškraba, 1976, 1978; Prairie et al., 1989; McCauley et al., 1989; Straškraba and Tundisi, 1999). Recent investigations have shown that once a certain concentration of total phosphorus is reached in a waterbody, the CHA concentration will then depend on the optical properties of the water (the CHA concentration is lower in colored or turbid waters), the mixing depth (the CHA concentration is higher in waters mixed only to a shallow depth), the water retention time (the phytoplankton is rapidly flushed from a lake with water retention times less than about 10 days), and on the fish stock (the CHA concentration is low if the fish stock is controlled, and consists of a balanced proportion

Table 3.12. Trophic grades of waterbodies, based on their chlorophyll-a (CHA) concentrations (Straškraba and Tundisi, 1999)

CHA (mg l^{-1})		Trophic grade	Raw water	Treatment
Summer average	Annual maximum			
0.3–5	< 10	Oligotrophic	Excellent	Standard
5–10	10–30	Mesotrophic	Suitable	Standard
10–25	30–60	Slightly eutrophic	Not very suitable	Exceptions
> 25	> 60	Highly eutrophic	Unsuitable	Special treatment

of predatory and planktivorous species—see Section 4). The CHA concentration also is low in highly calcareous waters, due to the co-precipitation of phosphorus with calcium (Koschel, 1997).

Nitrates

In most freshwater localities, nitrogen compounds are not a limiting factor for phytoplankton biomass. However, nitrogen limitation appears to exist in the tropics. Further, nitrates easily leach out of soils, especially during heavy rains and thaws, and can be detrimental to human health when present in drinking water in high concentrations. Toxic nitrites also can result from nitrate reduction. In the presence of nitrogenous organic compounds, the nitrites can become precursors of carcinogenic nitrosamines.

Any lake classification scheme based on nitrogen will depend on local conditions and health limits, which can vary. In certain geologic and lands use conditions, nitrates are very low and any increase will create concern, even at concentrations which would be considered an indication of unpolluted conditions in heavily-settled regions. Soils in the same region may have different nitrogen retention capacities, and the same amount of fertilizers applied in agriculture also can result in significantly different water pollution. Health limits in different countries vary between 20–50 mg NO_3 l^{-1}, and up to 100 mg l^{-1} in some places. Based on these considerations, the following classification of aquatic concentrations (expressed in mg NO_3 l^{-1}) can be delineated:

low levels	< 20,
critical levels	20–50,
dangerous levels	> 50.

Microbial parameters

Standard bacteriological parameters are indicative of allochthonous (usually human-induced) pollution (from tributaries and banks) or autochthonous (in-lake) pollution resulting from phytoplankton decomposition by psychrophilic bacteria.

Psychrophilic bacteria

Higher psychrophilic bacteria counts indicate a high content of easily-decomposed organic matter, which may not be of fecal origin. These compounds can make water treatment difficult because of their odor. High psychrophilic bacteria counts are found in polluted inflowing streams and at high water flow rates. They decrease slowly in a lake, via their sedimentation and their elimination by zooplankton. In stratified waters, psychrophilic bacteria counts are usually only hundreds per milliliter of water. Bacterial counts comparable to the inflow of a reservoir (or higher) are sometimes found in water layers near the reservoir dam, and usually indicate a short-circuiting undercurrent of polluted water from the inflow directly to the dam area. In eutrophic waters, with very high levels of primary production, psychrophilic bacteria may increase to counts of thousands/milliliter and higher because of decomposition of large phytoplankton biomasses.

Mesophyllic bacteria
These bacteria indicate the presence of easily decomposed organic matter, and the inflow of bacteria from an environment that is warmer than the water surface. In surface waters, mesophyllic bacterial counts are usually 1–2 orders of magnitude lower than psychrophilic bacterial counts. They are not of fecal origin. High proportions of mesophyllic bacteria, compared to psychrophilic bacteria, indicates contamination from manure, silage and similar sources. Mesophyllic bacteria do not reproduce in water, originating instead exclusively from sources outside a lake. Their numbers decrease by sedimentation and elimination by zooplankton once they enter a waterbody.

Coliform bacteria
These bacteria are exclusively of fecal origin (from human and warm-blooded animals), and enter waterbodies from external sources. Their numbers decrease in waterbodies as a result of dying, sedimentation and elimination by zooplankton. These elimination processes are slower during the cold season (sedimentation is the process least affected by temperature). However, the elimination processes also depend on water flow, the concentration of organic compounds, the content of sedimenting suspended matter (which reduces the bacterial counts) and other in-lake factors. In lakes, fecal bacteria counts are always lower in surface water layers than in the inflowing waters. In deep, stratified reservoirs, with theoretical water retention times of one month and more, the inflowing bacterial concentrations decrease within the reservoir by two or more orders of magnitude. Their concentrations near the reservoir dams, or in the center of lakes, usually are so low that their determination is not accurate and, therefore, not recommended. On the other hand, in shallow, nonstratified waterbodies with short water retention times and polluted water inflows, high coliform bacteria concentrations in some layers are always probable, especially after perturbation of the sediments (e.g., bathing, boats).

Faecal streptococci
These bacteria indicate recent water pollution. In inflowing waters, they usually reach numbers one order of magnitude lower than coliform bacteria. In lakes and reservoirs, their counts decrease more rapidly in the center of a lake or toward the dam in a reservoir. They are two orders of magnitude lower in number than coliform bacteria in the some spot. Their other characteristics are identical to those of coliform bacteria.

3.5 HOW TO MAKE CONCLUSIONS FOR LAKE MANAGEMENT

The objectives of this section are to (i) outline approaches by which conclusions can be drawn from data determined by methods described in the previous two chapters, and (ii) to make decisions about necessary measures for improving water quality. The types of conclusions can be divided into three categories:

- Conclusions concerning the watershed (Section 3.5.1),
- Conclusions concerning management of the waterbody (Section 3.5.2),
- Conclusions concerning water-treatment plants (Section 3.5.3).

3.5.1 Assessing the Watershed Effects on Lake Water Quality

The water quality and chemical composition of a waterbody is a function of its watershed. To estimate the probable composition of water that will flow into a planned reservoir (if reliable data are not available), or the causes of water quality changes observed in an existing waterbody, it is advisable to begin with the least variable parameters. These parameters include pollutant loads from the watershed (particularly valuable for compounds that are briefly or not at all retained in the watershed, such as nitrates, sulfates and chlorides), the dependence of pollutant loads from the watershed on the water inflow, the proportion of forests and farmland in the watershed, the number and magnitude of the point sources in the watershed, and the watershed geology. Information on weathering (especially for mixed subsoil), and on the effects of individual types of soil, is less definitive.

To estimate the total concentration of salts (conductivity) and hydrogen carbonates (alkalinity), it is useful to estimate the proportion of the watershed areas with sedimentary and crystalline subsoil. As an approximation, the alkalinity of water from a crystalline subsoil will be less than 0.2 mmol l^{-1}, and greater than 1.5 mmol l^{-1} from a sedimentary subsoil. Conductivity values less than 100 microSiemens cm^{-1} ($\mu S\ cm^{-1}$) can be expected in water from crystalline subsoils, and values greater than 300 $\mu S\ cm^{-1}$ in water from sedimentary subsoils. An estimate of the resultant concentrations (not including anthropogenic inputs) can then be calculated on the basis of the proportion of the two main types of subsoil in the watershed area.

An estimate of anthropogenic effects on the concentrations of water quality parameters in a lake can be made as follows:

(1) The watershed is divided into forested areas and farmland,
(2) The material loads to the waterbody are estimated, and
(3) The average in-lake concentrations of the materials are then calculated from specific load values.

The estimate is complemented by the magnitude of the point source loads (e.g., sewage, population, number of cattle in stables, industrial sources with their respective technologies; in all cases, the effectiveness of sewage treatment plants must be taken into account in regard to a given parameter). Temporal changes based on the dependence of the concentrations on the flow can then be envisaged.

If the in-lake concentrations of sulfate (SO_4^{2-}) are higher than those estimated by the above procedure, no simultaneous increases in nitrate (NO_3^-) and chloride (Cl^-) concentrations are seen and subsoil sources can be excluded, an atmospheric source of sulfate is indicated. The impact of atmospheric acidification is particularly strong in forested areas (especially coniferous forests) and reaches its maximum at altitudes about 1000 meters above sea level (m.a.s.l.).

Approximately simultaneous changes in the concentrations of NO_3^-, SO_4^{2-}, Cl^-, and usually Ca^{2+}, which are directly proportional to the flow rate, indicate a substantial load from farmlands. In large, mixed watersheds, the dependence of concentrations on flows is less regular (e.g., shifts in time and weight, a slight asynchrony with rainfall, water retention in impoundment). However, the annual average concentrations are proportional to the annual average flows in such watersheds. Point sources, and substances present in the watershed only in small quantities, are indicated by an indirect dependence of concentration on flow.

If the contents of heavy metals (especially cadmium, mercury and beryllium) exceed the World Health Organization norm for drinking water (WHO, 1984), they most often originate from three sources:

(a) Waste from metal-finishing plants—in these cases, waste discharges into sewerage or streams must be prevented,
(b) Some fertilizers, especially imported phosphates—in these cases, different types of fertilizers must be used,
(c) Mine water or pit heaps, even those which have not been worked for very long periods of time.

The function of *protective zones* is to prevent the direct pollution of water at the water withdrawal site (offtake site). However, because of their small area in a waterbody, they cannot protect the waterbody from the load of compounds entering it from the watershed. Chemical water quality parameters are affected by the entire watershed.

In order to be useful in preventing pollutants from entering a main reservoir, *pre-impoundments* must be deep enough for temperature stratification to occur (i.e., average depth (z) greater than 8 meters, length (L) greater than one kilometer, average water retention time (RT) greater than 2–3 weeks). In contrast, shallow reservoirs with short retention times seem to serve primarily as a type of culture tank, enhancing phytoplankton development. The short retention time prevents the growth and reproduction of herbivores which normally could help control the phytoplankton. In addition, dredging a reservoir before approximately one quarter of it is filled with mud is necessary to prevent the accumulation of sediments in the waterbody. If these prerequisites are met, pre-impoundments can successfully retain significant quantities of phosphorus and sediment-bound organic pollutants (except for nitrates).

Recreational use of drinking water supply waterbodies can be a serious public health problem, resulting from direct contamination of a waterbody by human pathogens and pollution by trodden mud. Thus, the public-health aspects of lakes must receive the highest priority. Nevertheless, although raw drinking water is taken from the outflow or the reservoir proper, bathing is permitted in many reservoirs. In such cases, certain minimum measures must be taken. The recreation should be restricted to zones distant from the drinking water intakes. One measure is the installation of dry latrines for bathers as far from the waterbody as possible, using concrete cofferdams insulated by 10 meter layers of sand mixed with iron swarf (1–2% of the weight of sand). Furthermore, parking lots and stalls should not be closer to the waterbody than the second zone. Restaurants and shops must not be closer than 300 meters from the shoreline, and their sewage diverted from the waterbody. Regular removal of all kinds of garbage must be strictly enforced. The hazard of bacterial contamination is particularly serious in shallow waterbodies, whose sediments are intensively disturbed by bathers. Faecal bacteria (including pathogenic species) can survive in sediments for several months, even in winter. Recreation use should be prohibited at small, shallow, water supply waterbodies of the fish-pond type.

3.5.2 Conclusions Concerning the Waterbody

The most important water quality changes within a waterbody, compared to its inflow, include the following:

(A) *Mineralization of organic compounds*, indicated by a decrease in biochemical oxygen demand (BOD), chemical oxygen demand (COD) and water color. The effects of mineralization are positive only up to a certain limit. If dissolved oxygen is depleted in some water layers (usually at the bottom), the situation described below in paragraph (C) will take place. The intensity of mineralization depends on the load, concentration and composition of organic compounds in the inflowing stream, and the water retention time, as well as circumstances that either enhance or suppress the development of organisms that support mineralization.

(B) Improvement of water quality (a decrease in phosphorus, organic substances, non-dissolved (particulate) matter), due to the *sedimentation of particulate matter and dissolved substances* that becomes particulate matter after they enter a waterbody. As with mineralization, the effects are desirable only up to a certain level, primarily when the waterbody becomes silted or when anoxic conditions develop—see paragraph (C). The impacts of these factors depend on the quantity and the nature of inflowing substances and on water flows, convection and mixing in the waterbody.

(C) Deterioration of water quality, due to *intensive decomposition of organic compounds*, especially if anoxic conditions develop, as well as the related processes of either releasing or binding certain substances in the sediments (e.g., releases of iron, manganese, ammonia, hydrogen sulphide and nitrites; binding of nitrates). In addition to factors given above in paragraph (A) and others, the concentrations of released and bound substances also can play a decisive role in defining the water quality in a stream or waterbody.

(D) Deterioration of water quality, due to an *excessive production of organic matter* in the form of phytoplankton. The load of phosphorus (and other critical nutrients), the optical qualities of water (color, turbidity), pH and the carbon dioxide (CO_2^+) concentration of in the inflowing stream are of prime importance. The critical nutrient concentration in a waterbody depends on its internal cycling (e.g., binding or release in sediments, incorporation by organisms), the development of phytoplankton, temperature, light and mixing, and biotic relations (i.e., consumption of phytoplankton by higher trophic level organisms in the food chain).

Other processes to consider when attempting to improve the water quality in a waterbody include the extinction or proliferation of pathogenic microorganisms, toxic effects on organisms, internal circulation of substances and acidification.

3.5.3 Conclusions for Water Treatment Plants

Water treatment technology must be planned in such a way that raw water of any quality occurring in a waterbody over the course of a year can be treated and the desired quality of the produced drinking water guaranteed (WHO, 1984). The water quality depends primarily on the inflowing water quality and on processes taking place in the waterbody in different seasons.

Criteria, based on analyses of the inflow waters, can aid in the determination of relevant treatment technology (Table 3.13).

The principal criteria for planning treatment technology are (i) the content and composition of organic substances in the inflowing water, and (ii) the potential development of phytoplankton, which can produce autochthonous organic compounds and which depend on the quantity of phosphorus in the inflowing waters.

Table 3.13. Tentative criteria for the design of the technology for treating a waterbody, based on the quality of the inflow

Criterion	Approximate range	Technology
Dissolved organic	< 4	—
carbon (DOC) (mg l^{-1})	4–10	Coagulation + filtration
	10–20	Coagulation + ozonation + activated carbon adsorption + filtration
	> 20	Unsuitable for drinking water production
Trophic status	Oligotrophic	Filtration
	Mesotrophic	Coagulation + filtration
	Eutrophic	Coagulation + ozonation + activated carbon adsorption + filtration
Nonsettling colloids	< 5	—
(as turbidity)	> 5	Coagulation + filtration
Alkalinity (mmol l^{-1})	< 0.2	Deacidification, or coagulation with a prepolymerized coagulant
Contents of calcium (Ca^{2+})	> 0.2	—
	< 0.4	Alkalinization, augmentation
Magnesium (Mg^{2+}) ions (mmol l^{-1})	0.4–5	—
of the contents of Ca^{2+}/Mg^{2+} ions	> 5	Unsuitable for drinking water production

Dissolved organic compounds are a particular problem in treating water, primarily because suspended compounds are easily removed in most cases. Their quantities also can diminish as a result of in-lake sedimentation.

The specific technological removal procedure must be chosen with particular attention given to the composition of dissolved organic compounds in the raw water. Naturally-colored substances originating in a watershed are the easiest compounds to remove, especially the macromolecular compounds.

The theoretical water retention time is of primary importance in regard to the ability of a reservoir to equalize the water quality variations of its inflowing waters, and to thermal stratification (Section 5.1). If the long-term annual average water retention time (*RT*) is less than 20 days, it is difficult to assess the ability of a reservoir to equalize the quality of inflowing water at high flow rates. If *RT* is longer than 100 days, the effect of water quality variations of the inflowing water is not significant (except for floods).

The maximum depth and water retention time, as well as the morphology of the waterbody and its surroundings (which affects wind mixing) determine waterbody stratification and, therefore, the possibility of optimizing the depth of the water withdrawal structure (i.e., offtake depth).

In all water supply reservoirs with maximum depths more than 10 meters and water retention times greater than 20 days, it should be possible to choose the offtake depth. Table 3.14 summarizes the criteria for optimizing the offtake of raw water.

Table 3.14. Raw water criteria for selection of offtake depth in water supply reservoirs (Straškraba et al., 1993)

Variable	Action
COD_{Cr} (COD_{Mn}) (mg l^{-1})	Minimization, or minimization of dose of coagulants in coagulation test
Alkalinity (mmol l^{-1})	Optimization with regard to chemical oxygen demand (COD) concentration
Color, absorbency at 250–370 nm	Minimization
Temperature (°C)	< 12 suitable > 12 unsuitable
Dissolved oxygen (% of saturation)	> 20 suitable; must be increased at least to 50 during treatment < 20 unsuitable; danger of an increase in Fe, Mn, NH_3 and H_2S concentration
Iron (mg l^{-1})	< 0.3 = suitable > 0.3 = separation during treatment necessary
Ammonia and ammonia ions (mg l^{-1})	< 0.5 = not noxious > 0.5 = must be removed during treatment
Organisms (counts of bacteria phytoplankton and zooplankton)	Minimization

In deep, stratified reservoirs, water of optimal quality for treatment usually is found throughout the year at a depth corresponding to the depth of the upper part of the hypolimnion during the period of summer stratification.

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