Chapter 7* - Lakes

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7.1. Introduction

The sampling of lakes for the purpose of assessing water quality is a complex process, as is the interpretation of the data obtained. Consequently, these activities must be based on a relatively complete understanding of basic limnology. The strategies employed for sampling and data interpretation are also controlled by lake use, the issue or lake problem being addressed, and the availability of resources for undertaking an assessment programme.

A detailed discussion of limnology is beyond the scope of this book and the reader is referred to the many texts which cover the subject in detail such as Hutchinson (1957), Golterman (1975), and Wetzel (1975). This chapter presents the more detailed descriptions and examples of processes which are essential to provide sufficient understanding to aid the design of sampling programmes and sensible interpretation of monitoring data. Further examples of the assessment of water quality in lakes using particulate material or biological methods are given in Chapters 4 and 5.

A lake may be defined as an enclosed body of water (usually freshwater) totally surrounded by land and with no direct access to the sea. A lake may also be isolated, with no observable direct water input and, on occasions, no direct output. In many circumstances these isolated lakes are saline due to evaporation or groundwater inputs. Depending on its origin, a lake may occur anywhere within a river basin. A headwater lake has no single river input but is maintained by inflow from many small tributary streams, by direct surface rainfall and by groundwater inflow. Such lakes almost invariably have a single river output. Further downstream in river basins, lakes have a major input and one major output, with the water balance from input to output varying as a function of additional sources of water.

Lakes may occur in series, inter-connected by rivers, or as an expansion in water width along the course of a river. In some cases the distinction between a river and a lake may become vague and the only differences may relate to changes in the residence time of the water and to a change in water circulation within the system. In the downstream section of river basins, lakes (as noted above) are separated from the sea by the hydraulic gradient of the river, or estuarine system. The saline waters of the Dead Sea,
the Caspian and Aral seas are, therefore, strictly lakes whereas the Black Sea, with a direct connection to the Mediterranean via the Sea of Marmara, is truly a sea.

Lakes are traditionally under-valued resources to human society. They provide a multitude of uses and are prime regions for human settlement and habitation. Uses include drinking and municipal water supply; industrial and cooling water supply; power generation; navigation; commercial and recreational fisheries; body contact recreation, boating, and other aesthetic recreational uses. In addition, lake water is used for agricultural irrigation, canalisation and for waste disposal. It has been commonly believed that large lakes have an infinite ability to absorb or dilute industrial and municipal waste, and it is largely as a result of human waste disposal practices that monitoring and assessment are proving to be necessary in many large lakes.

Good water quality in lakes is essential for maintaining recreation and fisheries and for the provision of municipal drinking water. These uses are clearly in conflict with the degradation of water induced by agricultural use and by industrial and municipal waste disposal practices. The management of lake water quality is usually directed to the resolution of these conflicts. Nowhere in the world has lake management been a totally successful activity. However, much progress has been made particularly with respect to controllable point source discharges of waste. The more pervasive impacts of diffuse sources of pollution within the watershed, and from the atmosphere, are less manageable and are still the subject of intensive investigations in many parts of the world.

7.2. Characteristics and typology

7.2.1 Origins of lakes

In geological terms lakes are ephemeral. They originate as a product of geological processes and terminate as a result of the loss of the ponding mechanism, by evaporation caused by changes in the hydrological balance, or by in filling caused by sedimentation. The mechanisms of origin are numerous and are reviewed by Hutchinson (1957), who differentiated 11 major lake types, sub-divided into 76 sub-types. A full discussion is beyond the scope of this chapter, but a summary of the 11 major types of lake origin is given below (Meybeck, 1995).

**Glacial lakes:** Lakes on or in ice, ponded by ice or occurring in ice-scraped rock basins. The latter origin (glacial scour lakes) contains the most lakes. Lakes formed by moraines of all types, and kettle lakes occurring in glacial drift also come under this category. Lakes of glacial origin are by far the most numerous, occurring in all mountain regions, in the sub-arctic regions and on Pleistocene surfaces. All of the cold temperate, and many warm temperate, lakes of the world fall in this category (e.g. in Canada, Russia, Scandinavia, Patagonia and New Zealand).

**Tectonic lakes:** Lakes formed by large scale crustal movements separating water bodies from the sea, e.g. the Aral and Caspian Seas. Lakes formed in rift valleys by earth faulting, folding or tilting, such as the African Rift lakes and Lake Baikal, Russia. Lakes in this category may be exceptionally old. For example, the present day Lake Baikal originated 25 million years ago.
**Fluvial lakes:** Lakes created by river meanders in flood plains such as oxbow and levee lakes, and lakes formed by fluvial damming due to sediment deposition by tributaries, e.g. delta lakes and meres.

**Shoreline lakes:** Lakes cut off from the sea by the creation of spits caused by sediment accretion due to long-shore sediment movement, such as for the coastal lakes of Egypt.

**Dammed lakes:** Lakes created behind rock slides, mud flows and screes. These are lakes of short duration but are of considerable importance in mountainous regions.

**Volcanic lakes:** Lakes occurring in craters and calderas and which include dammed lakes resulting from volcanic activity. These are common in certain countries, such as Japan, Philippines, Indonesia, Cameroon and parts of Central America and Western Europe.

**Solution lakes:** Lakes occurring in cavities created by percolating water in water-soluble rocks such as limestone, gypsum or rock salt. They are normally called *Karst lakes* and are very common in the appropriate geological terrain. They tend to be considered as small, although there is some evidence that some large water bodies may have originated in this way (e.g. Lake Ohrid, Yugoslavia).

Excluding reservoirs, many other natural origins for lakes may be defined, ranging from lakes created by beaver dams to lakes in depressions created by meteorite impact.

### 7.2.2 Classification of lakes

As noted in the brief discussion above the first level of classification of lakes is defined by their origin. However, in the context of lake use and assessments such a classification is of little value. Two other systems of classification which are based upon processes within lakes, and which are used universally, provide the basis upon which assessment strategies and interpretation are based. These are the physical or thermal lake classification and the classification by trophic level.

### 7.2.3 Physical/thermal lake types

The uptake of heat from solar radiation by lake water, and the cooling by convection loss of heat, result in major physical or structural changes in the water column. The density of water changes markedly as a function of temperature, with the highest density in freshwater occurring at 4 °C. The highest density water mass usually occurs at the bottom of a lake and this may be overlain by colder (0-4 °C) or warmer (4-30 °C) waters present in the lake. A clear physical separation of the water masses of different density occurs and the lake is then described as being stratified. When surface waters cool or warm towards 4 °C, the density separation is either eliminated or reaches a level where wind can easily induce vertical circulation and mixing of the water masses producing a constant temperature throughout the water column. In this condition the lake is termed homothermal and the process is defined as vertical circulation, mixing, or overturn.

The nomenclature applied to a stratified lake is summarised in Figure 7.1 in which three strata are defined:
• the epilimnion or surface waters of constant temperature (usually warm) mixed throughout by wind and wave circulation,

• the deeper high density water or hypolimnion (this is usually much colder, although in tropical lakes the temperature difference between surface and bottom water may be only 2-3 °C), and

• a fairly sharp gradational zone between the two which is defined as the metalimnion.

The name metalimnion is not commonly used and the gradation is normally referred to as the thermocline. The thickness of the epilimnion may be quite substantial, and it is dependent on the lake surface area, solar radiation, air temperature and lateral circulation and movement of the surface water. Commonly, it extends to about 10 m depth but in large lakes it can extend up to 30 m depth. Stratification in very shallow lakes is generally rare since they have warm water mixing throughout their water column due to wind energy input. However, winter or cold water stratification can occur even in the most shallow lakes under the right climatic conditions.

The interpretation of a shallow lake has never been satisfactorily defined, although there is a relationship between lake depth and surface area which controls the maximum depth to which wind induced mixing will occur. Therefore, an acceptable definition of a shallow lake (for the purposes of this discussion) is one which will overturn and mix throughout its water column when subjected to an average wind velocity of 20 km h⁻¹ for more than a six hour period. As a general rule, wind exposed lakes of 10 m depth or less are defined as shallow water lakes.

Figure 7.1 Typical temperature profile from a stratified lake in the temperate zone, showing the division of the water into three layers of different density
The thermal characteristics of lakes are a result of climatic conditions that provide a useful physical classification which is based upon the stratification and mixing characteristics of the water bodies. These characteristics are illustrated in Figure 7.2 with the lake types and terminologies defined as below. *Dimictic lakes* occur in the cool temperate latitudes. Overturn occurs twice a year, normally in the spring and autumn. Heating in the spring results in stratification with a warm water epilimnion during the summer. The autumn overturn results in homothermal conditions (at approximately 4 °C) which then cool to create a cold water inverse stratification during the winter months. Spring warming results in mixing and a re-establishment of the annual cycle. The stratification and mixing processes for a large dimictic lake are illustrated in Figure 7.3. This type of lake is the most common form of lake. Since the cool temperate latitudes encompass most of the world’s industrial nations they have been subjected to the most intensive study and represent the greatest part of our limnological knowledge.

*Cold monomictic lakes* occur in cold areas and at high altitudes (sub-polar). The water temperature never exceeds 4 °C and they have a vertical temperature profile close to, or slightly below, 4 °C. They have winter stratification with a cold water epilimnion, often with ice cover for most of the year, and mixing occurs only once after ice melt.

**Figure 7.2 Lake thermal structure and classification based on mixing characteristics (After Häkanson and Jansson, 1983)**
Warm monomictic lakes occur in temperate latitudes in subtropical mountains and in areas strongly influenced by oceanic climates. In the same way as their cold water counterparts, they mix only once during the year with temperatures that never fall below 4 °C.

Polymictic lakes occur in regions of low seasonal temperature variations, subject to rapidly alternating winds and often with large daily (diurnal) temperature variations. These lakes have frequent periods of circulation and mixing and may be subdivided into cold polymictic, which circulate at temperatures close to 4 °C, and warm polymictic which circulate at higher temperatures. As defined above, all shallow lakes fall within this category.

Figure 7.3 Annual thermal stratification and mixing in a dimictic lake (After Häkanson and Jansson, 1983)

Oligomictic lakes occur in tropical regions and are characterised by rare, or irregular, mixing with water temperatures well above 4 °C.

Amictic lakes occur in the polar regions and at high altitudes. They are always frozen and never circulate or mix. Waters beneath the ice are generally at, or below, 4 °C depending on the amount of heat generated from the lake bed or by solar radiation through the ice. These lakes show an inverse cold water stratification.
In addition to the lake types noted above some additional characteristics and terminology need to be defined. Meromictic lakes are those which do not undergo complete mixing throughout the water column. Most deep lakes in tropical regions are meromictic, such as Lakes Tanganyika and Malawi. Complete mixing as described above characterises holomictic lakes. When lake stratification is due to density changes caused by salt concentrations (normally the case in meromictic lakes), the gradient separating the upper layer from the denser layer is termed a chemocline or halocline, as distinct from temperature separation by a thermocline.

Figure 7.4 The global distribution of thermal lake types in relation to latitude and altitude (After Wetzel, 1975)

The physical or thermal classification of lakes, as described above, is largely climate controlled and is, therefore, related to latitude and altitude as summarised graphically by Wetzel (1975) (Figure 7.4).

7.2.4 Trophic status

The concept of trophic status as a system of lake classification was introduced by early limnologists such as Thienemann (1925, 1931) and Nauman (1932), and has been subject to continuous development up until the present time (Vollenweider, 1968; Pourriot and Meybeck, 1995). The process of eutrophication underlying this scheme is one of the most significant processes affecting lake management and is, therefore, described in more detail. The underlying concept is related to the internal generation of organic matter which is also known as autotrophic production. External inputs of organic matter from the watershed (allofrophic) produce dystrophic lakes rich in humic materials (see Figure 7.5). Such lakes may also be termed brown water or polyhumus lakes. In these lakes, most of the organic matter is derived from the surrounding watershed and internal carbon production is generally low.
The continuous process of eutrophication results from autotrophic production of internal organic matter by primary producers (i.e. the photosynthetic plants and algae) from the nutrients available within the lake. Nutrients are derived from external inputs to the lake or by internal recycling from the decay of organic matter and dissolution from bottom sediments. The process is illustrated schematically, in a simplified form, in Figure 7.6. Eutrophic lakes range from oligotrophic to hypereutrophic (Figure 7.5). In many shallow lakes, eutrophication may be manifest by macrophyte growth, rather than in the growth of phytoplankton. However, the efficient utilisation of the nutrients depends on the interplay of a number of factors which together define the growth conditions, and hence the resultant total biomass production at the primary producer level (see causes in Figure 7.6). The grazing of phytoplankton by zooplankton (secondary producers) and predation by fish (tertiary consumers) constitute the carbon transfer system of the lake. The efficiency of the system is dependent on two factors. Firstly, the quantity of biomass created at the primary producer level and, secondly, the species composition which determines the efficiency of grazing and the quantity and quality of the fish which terminate the internal food chain. With death, organisms at the primary, secondary and tertiary levels decay, resulting in the recycling of nutrients to the lake system.
The effects of eutrophication can be highly detrimental to lake water quality and severely limit the uses for which the water is suitable. Some effects are listed in Figure 7.6. However, it should be recognised that the detrimental effects result from the inefficient use of the phytoplankton biomass which, in turn, is derived from high nutrient availability. This is a result of changes in the dominance of algal species which are not consumed, or are ineffectively consumed, by zooplankton grazers.

The classification of lakes based on trophic level is shown in Figure 7.5. As mentioned above, these represent a continuous range of nutrient concentrations and associated biomass production. The names given to the classifications represent empirically defined intervals ranging from very low to very high productivity, which are defined below.

**Oligotrophic lakes:** Lakes of low primary productivity and low biomass associated with low concentrations of nutrients (N and P). In temperate regions the fish fauna is dominated by species such as lake trout and whitefish. These lakes tend to be saturated with $O_2$ throughout the water column.

**Mesotrophic lakes:** These lakes are less well defined than either oligotrophic or eutrophic lakes and are generally thought to be lakes in transition between the two conditions. In temperate regions the dominant fish may be whitefish and perch. Some depression in $O_2$ concentrations occurs in the hypolimnion during summer stratification.
**Eutrophic lakes:** Lakes which display high concentrations of nutrients and an associated high biomass production, usually with a low transparency. In temperate regions the fish communities are dominated by perch, roach and bream. Such lakes may also display many of the effects which begin to impair water use. Oxygen concentrations can get very low, often less than 1 mg l⁻¹ in the hypolimnion during summer stratification.

**Hypereutrophic lakes:** Lakes at the extreme end of the eutrophic range with exceedingly high nutrient concentrations and associated biomass production. In temperate regions the fish communities are dominated by roach and bream. The use of the water is severely impaired as is described below. Anoxia or complete loss of oxygen often occurs in the hypolimnion during summer stratification.

**Dystrophic lakes:** As defined previously, these are organic rich lakes (humic and fulvic acids) with organic materials derived by external inputs from the watershed. Summary characteristics for each of these trophic lake types are given in Table 7.1.

### 7.2.5 Water balance

The water balance of a lake may be expressed simply as:

\[
\text{Input} (\text{major river + lake tributaries + precipitation}) - \text{Evaporation} \pm \text{Groundwater} = \text{Output}
\]

Seasonal climate changes, particularly rainfall and solar heating, result in seasonal variations in the water balance producing a predictable seasonal variation in water level. Such fluctuations may affect water use, and many lakes have man-made control structures to dampen the water level fluctuations and optimise year-round water use. Longer-term changes in climate, such as dry and warm years as opposed to cool and wet years (particularly in the temperate zones), have major effects on the water levels of a lake. Considerable impairment of water use may occur as a result of these long-term cycles (e.g. navigation/dredging, coastal erosion, shifts in biological habitat, flooding and loss of wetlands).

The concept of water residence time (or turnover time) is particularly important. Residence time may be expressed theoretically as the lake water volume divided by the rate of total outflow. Residence times in normal lakes extend from a few days to many tens of years, or even a century or more. However, the theoretical residence time may be expected to occur only rarely, since it is based on the homogeneous mixing of lake waters. Depending on their thermal structure, deep water mixing and water circulation characteristics, most lakes may be characterised at any one time by different water masses which change from season to season, and possibly from year to year. Warm water in the summer epilimnion overflows the colder water hypolimnion and has a much shorter residence time than a mixed water mass. Conversely, deep waters with poor circulation have a much longer residence time than the theoretical value (Meybeck, 1995).
Table 7.1 Nutrient levels, biomass and productivity of lakes at each trophic category

<table>
<thead>
<tr>
<th>Trophic category</th>
<th>Mean total phosphorus (mg m(^{-3}))</th>
<th>Annual mean chlorophyll (mg m(^{-3}))</th>
<th>Chlorophyll maxima (mg m(^{-3}))</th>
<th>Annual mean Secchi disc transparency (m)</th>
<th>Secchi disc transparency minima (m)</th>
<th>Minimum oxygen (%sat)(^{1})</th>
<th>Dominant fish</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ultra-oligotrophic</td>
<td>4.0</td>
<td>1.0</td>
<td>2.5</td>
<td>12.0</td>
<td>6.0</td>
<td>&lt;90</td>
<td>Trout, Whitefish</td>
</tr>
<tr>
<td>Oligotrophic</td>
<td>10.0</td>
<td>2.5</td>
<td>8.0</td>
<td>6.0</td>
<td>3.0</td>
<td>&lt;80</td>
<td>Trout, Whitefish</td>
</tr>
<tr>
<td>Mesotrophic</td>
<td>10-35</td>
<td>2.5-8</td>
<td>8-25</td>
<td>6-3</td>
<td>3-1.5</td>
<td>40-89</td>
<td>Whitefish, Perch</td>
</tr>
<tr>
<td>Eutrophic</td>
<td>35-100</td>
<td>8-25</td>
<td>25-75</td>
<td>3-1.5</td>
<td>1.5-0.7</td>
<td>40-0</td>
<td>Perch, Roach</td>
</tr>
<tr>
<td>Hypereutrophic</td>
<td>100.0</td>
<td>25.0</td>
<td>75.0</td>
<td>1.5</td>
<td>0.7</td>
<td>10-0</td>
<td>Roach, Bream</td>
</tr>
</tbody>
</table>

\(^{1}\) % saturation in bottom waters depending on mean depth

Sources: Häkanson, 1980; Häkanson and Jansson, 1983; Meybeck et al., 1989

The concept of residence time is important because it provides some indication of the recovery, or self-purification rates, for individual water bodies. If a water body becomes polluted with a soluble toxic element, and the source of the pollutant is entirely eliminated, the residence time provides an indication of how long it will take to remove the polluted water and replace it with non-polluted water. For Lake Superior, elimination of a soluble toxic element would take 100 years, whereas for Lake Erie it would take only 2.4 years. However, this is an over-simplification since it does not take account of lake structure or the geochemistry of the element. Most shallow lakes (< 5 m deep) have a short residence time of one year or less.

7.3. Water quality issues

A water quality issue may be defined as a water quality problem or impairment which adversely affects the lake water to an extent which inhibits or prevents some beneficial water use. Since a water quality issue normally results from the deleterious effects of one or more human uses, major conflicts between users or uses may occur in lake systems subjected to multiple use. These kinds of water quality conflicts and associated human interactions bring considerable complexity to lake management. This often results in continuous arbitration between user-groups, and failure to take the necessary control actions to restore or maintain lake water quality. Since the problems of water quality relate to water use, water quality assessment strategies should be designed in these terms. The interpretation should be dependent on adequate knowledge of lake physics and structure, lake uses and associated water quality requirements, and the legislative powers or authorities which may be used to enforce and ensure compliance with standards. Monitoring data and the associated assessments are thus the basis upon which sound lake management should be conducted.
A number of issues affect lake water quality. These have mostly been identified and described in industrialised regions, such as the North American Great Lakes, which have progressed from one issue to another in a sequence parallel to social and industrial development (Meybeck et al., 1989). Meybeck et al. (1989) have highlighted that the developing nations of today are responsible for managing lakes which are being subjected to synchronous pollution from the simultaneous evolution of rural, urban and industrial development, as distinct from the historical progression which occurred in developed societies. This means that, in the developing world, multi-issue water quality problems must be faced with greater cost and complexity in assessment design, implementation and data interpretation. The current issues facing lake water quality are discussed below.

7.3.1 Eutrophication

There is much published literature available on the subject of eutrophication, both in specialised reports such as OECD (1982) and in the major limnological texts recommended in the introduction to this chapter. Numerous indices have been developed to measure the degree of eutrophication of water bodies. Many have been based on phytoplankton species composition, but are not recommended since they are complicated to undertake, difficult to interpret and are affected by local conditions. The most reliable methods are based on the classifications given in Table 7.1. Suitable alternatives for biological indicators are total particulate organic carbon (POC) and chlorophyll a, since they represent total biomass.

In its simplest expression, eutrophication is the biological response to excess nutrient inputs to a lake. The increase in biomass results in a number of effects which individually and collectively result in impaired water use. These effects are listed in Figure 7.6. Meybeck et al. (1989) highlight that eutrophication is a natural process which, in many surface waters, results in beneficial high biomass productivity with high fish yields. Accelerated, or human-induced, change in trophic status above the natural lake state is the common cause of the problems associated with eutrophication. Such human-induced changes may occur in any water body, including coastal marine waters, although the progression and effects of eutrophication are also mediated by climate. As a result warm tropical and sub-tropical lakes are more severely affected than their colder water counterparts.

High nutrient concentrations in a lake are derived from external inputs from the watershed. The final biomass attained is determined primarily by the pool of nutrients available for growth at the beginning of the growing season. The primary nutrients, such as nitrogen and phosphorus, are used until growth is complete and the exhaustion of the pool of either one of them places a final limit on the phytoplankton growth. By definition, the nutrient which is exhausted is the limiting nutrient in any lake system. Meybeck et al. (1989) suggest that, in waters with a N/P ratio greater than 7 to 10, phosphorus will be limiting, whereas nitrogen will be limiting in lakes with a N/P ratio below 7.
Figure 7.7 shows the seasonal cycling of nutrient concentrations (soluble reactive P, total P, and NO$_3$-N) relative to primary production (as expressed by chlorophyll a) in the epilimnion of Lake Windermere, UK. Primary production occurs from April to late September, inversely to the depletion in soluble reactive P and NO$_3$-N. Soluble reactive P is reduced rapidly, virtually to zero, from a maximum concentration of about 16 µg l$^{-1}$, indicating that growth in this lake is limited by phosphorus.

Changes in transparency in lakes may be caused by increasing turbidity due to increasing concentrations of mineral material or to increasing plankton biomass. Increases in mineral matter are caused by:

- turbid in-flow in fluvial waters with high watershed erosion rates,
- resuspension of bottom sediment by wave action in shallow lakes, or shallow areas of lakes (when wave height and wavelength developed during storm events allow direct interaction with the bottom to take place), and
- shore line erosion due to wave impingement and to surface water gully erosion of unconsolidated shore line material.
High primary production is followed by death, settling, decay and deposition of the phytoplankton to the lake sediment. It has been calculated that in the decay process 1.5 to 1.8 g of oxygen is used per gram dry weight of mineralised organic matter. Oxygen is taken up from the water and, under highly eutrophic conditions, serious or complete depletion of hypolimnetic oxygen may occur with severe impacts on the lake system. Depletion or loss of oxygen in the hypolimnion of lakes used for drinking water supply presents a particularly serious problem. When combined with undesirable plankton species, it produces serious taste and odour problems (see Chapter 8 for further details). In addition, oxygen depletion and reducing conditions at the sediment-water interface result in the recycling of iron, manganese and related trace elements from the bottom sediments.

Figure 7.8 shows the seasonal change in dissolved oxygen throughout the water column of Lake Mendota, Minnesota together with the chlorophyll a values for the same period. Oxygen in the deeper waters progressively declines following the onset of primary production in April to a minimum of below 1 mg l\(^{-1}\) in the hypolimnion for July to October, during lake stratification. The system is re-oxygenated in late October following overturn and complete mixing of the lake water. In many lakes oxygen is completely used up and the hypolimnetic waters become anoxic. A typical example was the anoxia in the hypolimnion of the Central Basin of Lake Erie in the 1960s which led to the massive clean up and phosphorus control programme of the Lower Great Lakes of North America between 1972 and 1981. Lake Erie is a dimictic lake of the cold temperate zone similar to Lake Mendota.

Deep water anoxia may be more prevalent in highly productive tropical lakes. Lake Tanganyika is a warm, amictic lake with a stable thermocline and epilimnion over a hypolimnion with a constant temperature of 23.5 °C. Oxygen, which is saturated in the surface water, declines throughout the water column with the decay of settling organic detritus. Oxygen is totally depleted at about 100 m depth and H\(_2\)S becomes abundant, resulting in the deeper waters becoming uninhabitable for fish.

Depletion of oxygen in lake bottom waters and the onset of anoxia results in the re-mobilisation of phosphorus and other elements from lake sediments. This mechanism was described by Mortimer (1942) and has been the subject of many investigations. Project Hypo (Burns and Ross, 1971) followed the release of phosphorus during anoxia in the hypolimnion of Lake Erie. It was also noted that with decreasing redox potential, manganese was the first element to be released from bottom sediments to the hypolimnion, followed by the synchronous release of iron and phosphorus. The association of phosphorus with iron in bottom sediments has been well established (e.g. Williams et al., 1976) and the reduction of Fe\(^{3+}\) to soluble Fe\(^{2+}\) results in the release of phosphorus. Other elements adsorbed by, or co-precipitated with, the oxides of iron and manganese are also released to the bottom waters under anoxic conditions. Significant quantities of hydrogen sulphide are not generally seen in freshwaters and were not seen in Lake Erie (Burns and Ross, 1971). Sulphate occurs in much lower concentrations in freshwaters than in marine waters and consequently, when reduced, it tends to precipitate as iron mono-sulphide. When all the sulphate has been reduced to sulphide, bacterial energy is derived by methane generation. Large quantities of this gas are generated under anoxic conditions in freshwaters. In anoxic water, nitrogen is commonly found as NH\(_4^+\). As with most reduced forms (Mn\(^{2+}\), Fe\(^{2+}\), H\(_2\)S, CH\(_4\)), the presence of NH\(_4^+\) may severely impair the use of the water, particularly as a drinking water source (due to
odour, taste, precipitation of metals upon re-aeration, etc.). This loss can be one of the most detrimental effects of eutrophication.

Figure 7.8 Oxygen concentrations and chlorophyll a in Lake Mendota, Minnesota during 1976. Depletion of oxygen in the hypolimnion in July, August and September is related to the period of high algal biomass (indicated by chlorophyll a) and is caused by biological degradation of the algae as they sink to the bottom (After ILEC, 1987-1989)

Phosphorus released to the hypolimnion usually remains in the deep water mass during the period of oxygen depletion. At overturn when the water is re-aerated, the phosphorus is mixed throughout the water column, precipitating as it becomes adsorbed onto the insoluble Fe II hydroxides and oxides and fine particles in the water column. In this manner phosphorus regenerated from bottom sediments (internal loading or secondary cycling) is made available for the subsequent algal growth cycle.

7.3.2 Health related issues and organic wastes

When located in densely populated areas, lakes may become polluted by inadequately treated, or untreated, human and animal wastes. This causes eutrophication, makes the water unfit for human consumption and increases health risks to water users. In an overview of Lake Managua in Nicaragua, Central America, Swaine (1990 pers. comm.) indicated that 1,587.3 deaths per 100,000 inhabitants in Nicaragua are due to waterborne diseases which include typhoid and paratyphoid fever, amoebiasis, viral jaundice, dysentery and various parasitic diseases. He noted that the first sewers were
built in Managua in 1922 and that by 1981 about 50 per cent of the 600,000 inhabitants were connected to the sewer system. As a result, in 1988 approximately 150,000 m$^3$ of untreated sewage were discharged to the lake. Coliform counts were routinely found in the order of 10$^6$ per 100 ml although no direct effect on human health had been established. However, with the high population centred in Managua there was a major effect on the health statistics of the country as a whole.

In many tropical lakes, the onset and progression of eutrophication results in increased growth of lake macrophytes, which in turn provide increased habitat for water fowl and snails which serve as secondary hosts for many human parasites such as Schistosoma.

In addition to bacterial and viral infection from poor or untreated sewage disposal, other major problems may occur. Wastes high in organic matter directly increase the chemical and biological oxygen demand in the receiving waters. This results in localised areas of oxygen depletion and the release of many trace elements by the reduction of iron and manganese (see above). In industrial societies, factories are often directly connected to the sewers and discharge metal and organic chemical pollutants to the sewage treatment system. In many cases, such effluents are highly toxic to aquatic organisms and should be subjected to analysis using bioassays (see Chapter 5). If found to be toxic, they must be analysed for trace elements and organic pollutants.

7.3.3 Contaminants

The major shift in management concern from nutrients to the issue of toxic chemicals in lake systems is largely due to the realisation that mercury contamination in Minimata Bay, Japan was responsible for major human health problems (D’Itri, 1971) together with the work on mercury transfer processes in freshwater carried out in Sweden (Jernalov, 1971). In parallel with the mercury problem, major concerns over the pesticide DDT have been investigated. The observation that DDT inhibited the hatching of lake trout eggs in Lake Michigan was a major factor in banning the use of DDT in North America in 1970 (Swaine, 1991 pers. comm.). Subsequent investigations of the toxicity and environmental cycling of metals and organic pollutants have resulted in the implementation of a variety of management strategies to alleviate real and perceived problems. Within this context, assessment of lake systems should provide the information upon which managerial actions can be taken. Such information includes:

- presence or absence of a metal or compound,
- concentrations of pollutants in water, sediment and biota,
- spatial distributions showing areas in which objectives or guidelines are exceeded (i.e. areas of non-compliance with regulatory requirements), and
- temporal changes in concentration resulting from changing socio-economic conditions and demography, or changes resulting from management intervention.
Sources of contaminants to lakes

The source of toxic pollutants to lakes is usually material derived from human activities. In many areas natural rock substrates may also result in high levels of toxic metals but these rarely cause human health problems since natural cycles of weathering, transport and deposition are slow processes. Human activity may be responsible for increasing the erosion rates (by mining and manufacturing use) of naturally occurring elements or by changing their chemical form, thereby allowing concentrations to reach levels which may be a hazard to aquatic ecosystems and to man. Organic compounds synthesised and manufactured industrially are not natural substances. Their occurrence in the environment is a direct manifestation of loss during manufacture, transportation and use of the compound.

Inputs, or general sources of contaminants to lakes may take a variety of pathways:

- Direct point sources, municipal and industrial effluent discharges.
- Diffuse agricultural sources: wash-off and soil erosion from agricultural lands carrying materials applied during agricultural land use, mainly herbicides and pesticides.
- Diffuse urban sources: wash-off from city streets, from horticultural and gardening activities in the sub-urban environment and from industrial sites and storage areas.
- Waste disposal: transfer of pollutants from solid and liquid industrial waste disposal sites and from municipal and household hazardous waste and refuse disposal sites.
- Riverine sources: inflow in solution, adsorbed onto particulate matter, or both. The cumulative input is the sum of contaminants from all of the rivers draining the watershed into a lake.
- Groundwater sources: groundwater systems polluted from point and diffuse sources (noted above) flowing into rivers, and directly into lake beds.
- Atmospheric sources: direct wet and dry atmospheric deposition to the lake surface amplified by the erosional recycling of atmospheric deposition on the drainage basin land surface. This latter process is defined as secondary cycling.

To determine loads and contaminant mass balances in any lake system, efforts must be made to estimate the contributions from each of the above noted sources. In many cases, (e.g. river inputs) detailed monitoring may be necessary to provide reasonable estimates of the total inputs.

Contaminant loads

The discussion on phosphorus above indicated that inputs have been controlled in order to achieve lake concentrations which, in turn, determined the level of productivity in the lake. A similar philosophy has been formulated in many nations in the management of toxic substances. Approaches have ranged from a regulated input level to total elimination of inputs to recipient water bodies.
Organisms react to the concentration and exposure time of a contaminant in the water body. The lake concentration, either in the water or the sediment, is a result of the contaminant load (mass per unit time) distributed in the lake, which is termed the loading (mass per unit volume [or area] per unit time). These differences can be confusing for water management policies which have established guidelines or objectives on the basis of concentrations rather than on an understanding of the input loads of the sources of contaminants.

The concept of inputs or loads with respect to river inflows to lakes, and the impact on local lake water, is illustrated below. However, in conditions where the range of the river discharge is greatly in excess of the range of contaminant concentrations, the following relationship does not hold true.

A. High river water volume \( \times \) Low contaminant concentration = High load \( \Rightarrow \) Slow, whole lake deterioration

\[ 10^6 \text{ m}^3 \text{ d}^{-1} \times 10^{-3} \text{ g m}^{-3} = 10^6 \text{ g d}^{-1} \]

B. Low river water volume \( \times \) High contaminant concentration = Low load \( \Rightarrow \) Impairment close to river input

\[ 10^3 \text{ m}^3 \text{ d}^{-1} \times 10^{-2} \text{ g m}^{-3} = 10 \text{ g d}^{-1} \]

In the case of A above there will be no effects on biota in the river since the concentration is low. However, the high lake load will result in the degradation of the receiving waters with long-term deleterious effects on the associated biota. In the second case (B), real effects will be observed on the biota in the river due to the high concentrations in a low water volume, but this in turn does not necessarily imply degradation of the whole lake which may adequately accommodate the low load delivered. Ordinarily, some form of control would be applied to the river input under scenario B, but would not apply under scenario A which actually represents a far worse condition for the general quality of the lake. The calculation of load can be illustrated for the Niagara River, using the same annual average river volume, as follows:

<table>
<thead>
<tr>
<th>Average annual concentration in Niagara River</th>
<th>Annual load to Lake Ontario</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 gm(^{-3})</td>
<td>26,380 t</td>
</tr>
<tr>
<td>1 mg m(^{-3})</td>
<td>26.38 t</td>
</tr>
<tr>
<td>1 µg m(^{-3})</td>
<td>26.38 kg</td>
</tr>
<tr>
<td>1 ng m(^{-3})</td>
<td>26.38 g</td>
</tr>
</tbody>
</table>

It has been estimated by Swain (1990, pers. comm.) that for top predator fish in Lake Ontario to achieve a concentration of 25 ng g\(^{-1}\) TCDD (tetra chlorinated dibenzo dioxin) in their tissues (i.e. the guideline concentration for human consumption) the lake would require a load of 5 g a\(^{-1}\).
The determination of load in a lake should result in the ability to produce a mass balance which provides an insight into the proportional contributions of different sources, and which allows a control strategy to be optimised. Examples of such mass balances are provided in Table 7.2 for lead in Lake Ontario. Despite some difficulty in estimating the direct deposition of lead to the lake surface by numerical models or direct measurement of land-based rain chemistry stations, relatively good mass balances have been achieved. Evaluation of the loads in the soluble and suspended solids phases display the important role played by sediment in the cycling of contaminants in lakes and reservoirs.

**Table 7.2** Lead mass balance for Lake Ontario, North America and proportional loadings from major sources

<table>
<thead>
<tr>
<th></th>
<th>Inputs (t a⁻¹)</th>
<th>% of total input</th>
<th>Deposited and outputs (t a⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Accumulating in sediment</td>
<td></td>
<td></td>
<td>725</td>
</tr>
<tr>
<td>Niagara River suspended solids</td>
<td>534</td>
<td></td>
<td></td>
</tr>
<tr>
<td>All other rivers suspended solids</td>
<td>176</td>
<td>69.6</td>
<td></td>
</tr>
<tr>
<td>Solute - all rivers</td>
<td>195</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Shoreline erosion</td>
<td>50</td>
<td>3.8</td>
<td></td>
</tr>
<tr>
<td>Dredged spoil</td>
<td>65</td>
<td>5.0</td>
<td></td>
</tr>
<tr>
<td>Airborne input (Precip. chem.)</td>
<td>280</td>
<td>21.6</td>
<td></td>
</tr>
<tr>
<td>Output suspended solids</td>
<td></td>
<td></td>
<td>547</td>
</tr>
<tr>
<td>Output solute</td>
<td></td>
<td></td>
<td>207</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td>1,300</td>
<td></td>
<td><strong>1,479</strong></td>
</tr>
</tbody>
</table>

Source: IJC, 1977

The major source of lead to Lake Ontario is fluvial, followed by atmospheric. The largest river load is from the Niagara River which drains from Lake Erie. The output from Lake Erie to the Niagara River is estimated at 496 t, whereas the Niagara River load to Lake Ontario is 754 t, indicating a direct contribution from other sources to the Niagara River of 258 t. Therefore, a reduction in load in Lake Erie will show a significant reduction to Lake Ontario. Furthermore, much of the fluvial load to both lakes arises from the recycling of lead deposited on the land surface from the atmosphere.

**Pathways**

Once in a lake system, pollutants follow a number of physical, chemical and biological pathways which are dependent, to a large extent, on the chemical characteristics of the element or compound. Elements and/or compounds which are soluble in water (hydrophilic) are transported with the physical circulation of the water mass in the lake. In the simplest circumstances, such a pollutant mixes throughout the lake water and is eliminated within the water residence time of the lake, which may be hours or decades depending on the lake. Pollution of a lake with a long turnover time is a potentially serious event because remedial actions only result in a slow reduction or elimination of the contaminant.
The distribution of water soluble elements is actually more complex than presented above. The input of high loads or concentrations impacts directly on the lake margin. Dispersion into the lake is controlled by diffusion, water mass movements resulting from hydraulic flow and lake circulation as induced by wind direction and velocity. Another factor which has a direct effect on retention time and distribution of soluble elements is related to lake stratification. Soluble elements in warm river water entering a stratified lake mix rapidly as an overflow into the warm waters of the epilimnion. Since the epilimnion often represents a small fraction of the lake water volume, the mixing and the turnover time are accelerated. The elimination rate can then be considered as the epilimnion volume divided by the discharge during the period of stratification. Conversely, as river waters cool in the autumn, they may inter-stratify in the hypolimnion or directly under-flow to the deeper waters of the hypolimnion. In this case mixing tends to be less rapid and the elimination period extends in an unpredictable fashion (i.e. without some precise knowledge of deep water circulation in the lake). The processes described above can only be understood when detailed sampling of the water column at a number of lake sampling points is carried out. Additionally, the location of water withdrawal points may accelerate or retard removal of soluble compounds depending on the thermal structure of the water body (see Chapter 8 for more details).

Most toxic trace elements and many organic pollutants (e.g. PCBs, DDT) of low solubility (hydrophobic) or which are fat soluble (lipophilic) are predominantly adsorbed by particles of inorganic or biological origin. The distribution and elimination of these kinds of materials closely follow the processes of sediment sorting and deposition which, in turn, are the products of lake circulation, wave induced hydraulic energy and settling of particles of different sizes in waters of different temperatures. These processes are discussed in more detail in Chapter 4. Elimination is mainly by sedimentation as illustrated for Pb in Lake Ontario in Table 7.2.

The characteristics of a compound, therefore, not only determine pathways but also indicate in which lake compartment (water, sediment or organisms) the pollutants can most probably be found. It is also significant that, as a general rule, toxicity is inversely related to solubility (the greater the solubility the lower the toxicity). Many toxic elements and compounds are often adsorbed or scavenged by particles in the source rivers, or in the lake itself, and are wholly or partially removed from contact with most lake organisms by sedimentation processes. However, this general rule is an over-simplification since major exceptions, such as the methylation of mercury and incorporation of lipophilic compounds in the tissues of benthic organisms, may result in recirculation in the lake food web.

The final factor involved in the rate of elimination of an element from a lake is its persistence. Organic compounds chemically decay at a defined rate depending on the physico-chemical conditions of the receiving lake water. Compounds may break down completely or may be changed to a different form, e.g. DDT to DDD or DDE; aldrin to dieldrin, etc. In some cases the decay product or metabolite may be more toxic than the parent compound.

**Bioaccumulation and biomagnification**

The processes of bioaccumulation and biomagnification have been discussed in Chapter 5. These processes are extremely important in the distribution of toxic substances in
freshwater ecosystems. Bioaccumulation and food chain amplification of concentrations of toxic compounds determine the exposure and consequent effects of these substances at each trophic level, including humans. Examples of food chain biomagnification of Hg and PCB in Lake Ontario are given in Figure 7.9. Not all substances are subject to biomagnification as is shown for zinc, lead and copper by organisms taken from the Lake Ontario food chain (Table 7.3).

Contaminant effects on lake biota

Current knowledge of the effects of contaminants on lake organisms is limited. High concentrations of pollutants such as mercury in fish and benthic organisms appear to have little effect upon the organisms themselves (Jackson, 1980). However, evidence that DDT and toxaphene have affected the hatching success of lake trout in Lake Michigan was instrumental in the banning of these compounds in North America (Swaine, 1991 pers. comm.). The emergence of mayflies (aquatic insects) in western Lake Erie ceased for a period from the 1950s to the late 1970s and is commonly believed to be due to the effects of persistent organochlorine compounds.

Evidence of the incidence of carcinoma (tumours) in some fish species in lakes affected by their proximity to urban centres has been presented by Black (1983). Black (1983) also determined that some of the tumours could be induced by sediment-bound PAHs (polychlorinated aromatic hydrocarbons). These findings have been supported by similar studies from estuarine and coastal marine environments (Malins et al., 1987).

Figure 7.9 Biomagnification of mercury and PCB in the food chain of Lake Ontario (After Thomas et al., 1988)
Table 7.3 Trace element concentrations in Lake Ontario biota

<table>
<thead>
<tr>
<th></th>
<th>Zinc (µg g⁻¹ wet weight)</th>
<th>Lead</th>
<th>Copper</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lake trout</td>
<td>10.35</td>
<td>0.61</td>
<td>1.30</td>
</tr>
<tr>
<td>Smelt</td>
<td>21.71</td>
<td>0.18</td>
<td>0.59</td>
</tr>
<tr>
<td>Sculpin</td>
<td>15.71</td>
<td>0.23</td>
<td>2.43</td>
</tr>
<tr>
<td><em>Pontoporeia</em></td>
<td>13.10</td>
<td>0.57</td>
<td>18.72</td>
</tr>
<tr>
<td><em>Mysis</em></td>
<td>13.66</td>
<td>0.46</td>
<td>0.90</td>
</tr>
<tr>
<td>Net plankton</td>
<td>15.16</td>
<td>1.07</td>
<td>0.92</td>
</tr>
</tbody>
</table>

Source: Thomas *et al.*, 1988

Effects of contaminants at lower trophic levels, particularly the phytoplankton, have not been widely demonstrated. Work by Munawar and Munawar (1982) has shown that small species of phytoplankton, less than 20 µm in size, are more susceptible to contaminants than the more robust plankton greater than 20 µm in size. Changes in lake biota related to changes in contaminant loads are not well documented. It is possible that indigenous plankton populations with short life cycles adapt to the loads and concentrations to which they are constantly exposed.

The effects of lead on fish in lakes have been demonstrated by Hodson *et al.* (1983), who showed a physiological response in the synthesis of the enzyme ALA-D, as determined by blood analysis. Continuous exposure to elevated lead concentrations results in a darkening of the tail (BlackTail) and ultimately death.

Current techniques for assessing the effects of contaminants, both soluble and particulate, are discussed in Chapter 5. With respect to lakes, the best techniques are currently based on bioassay methods which will continue to be used until adequate research improves knowledge of the community and physiological responses to pollutant stress by the organisms naturally occurring in lakes.

7.3.4 Lake acidification

One of the major issues related to lakes in particular, and to freshwaters in general, is the progressive acidification associated with deposition of rain and particulates (wet and dry deposition) enriched in mineral acids. The problem is characteristic of lakes in specific regions of the world which satisfy two major critical conditions:

- the lakes must have soft water (i.e. low hardness, conductivity and dissolved salts), and
- the lakes must be subjected to “acid rain” or more precisely, to total deposition enriched in sulphate (H₂SO₄) and nitrogen oxides (NOₓ) creating nitric acid.

All lakes have some acid buffering capacity due to the presence of dissolved salts from the watershed. However, in non-carbonate terrain, such as in areas of crystalline rocks or quartz sandstones, this buffering capacity is rapidly exhausted and free H₂ ions create...
a progressive acidification of the lake. This buffering capacity has been defined by Jeffries et al. (1986) who characterised lake sensitivity to acidification in terms of the acid neutralising capacity (ANC) as:

\[
ANC = \Sigma \text{base cations} - \Sigma \text{strong acid anions} = ([\text{Ca}]+[\text{Mg}]+[\text{Na}]+[\text{K}]) - ([\text{SO}_4]+[\text{NO}_3]+[\text{Cl}])
\]

Jeffries et al. (1986) used an ANC scale to define lakes in Eastern Canada as follows:

<table>
<thead>
<tr>
<th>ANC (µeq l(^{-1}))</th>
<th>Sensitivity</th>
</tr>
</thead>
<tbody>
<tr>
<td>≤ 0</td>
<td>Acidified</td>
</tr>
<tr>
<td>0-40</td>
<td>Very sensitive</td>
</tr>
<tr>
<td>40-200</td>
<td>Sensitive</td>
</tr>
<tr>
<td>&gt; 200</td>
<td>Insensitive</td>
</tr>
</tbody>
</table>

Lake sensitivity is a reflection of sub-surface geology and the associated soils (which may be easily mapped), whereas acidification is the result of acid deposition on lakes of differing sensitivity. These two factors have been well defined in Eastern Canada and are illustrated in the colour maps of Figures 7.10 and 7.11. The occurrence of high deposition (> 20 kg ha\(^{-1}\)) with highly sensitive terrain occurs as a band across the southern regions of the country from Central Ontario across Southern Quebec, almost to New Brunswick (Fraser, 1990 pers. comm.). Lake studies have indicated that the lakes in these areas have been subjected to progressive acidification and that any improvement will require substantial reduction in acid deposition from the 1982-1986 average levels shown in Figure 7.11. Summary statistics for pH and ANC for the lakes of Eastern Canada are given in Table 7.4.

The acidification of lake waters, together with the acid leaching of lake basin soils, results in the hydrolysis of the hydrated oxides of iron, manganese and aluminium with the resultant release of many trace elements which are toxic to aquatic organisms. Such release has been demonstrated in sediments from Norwegian lakes. An example is presented in Figure 7.12, in which depletion of zinc in the surface sediment occurred due to dissolution following lake acidification. The dissolution of aluminium has proved to be particularly dangerous for fish due to asphyxiation caused by the deposition of aluminium oxide on the gill filaments as it precipitates out of solution.

A correlation analysis of dissolved Al to H ion concentration in the lakes of Eastern Canada failed to show a good relationship. However, when the lakes were combined into a set of regional aggregates based on water hardness, geology and deposition, a clear relationship could be observed as is shown in Figure 7.13. Lake acidification is a major problem, limited geographically to those areas of the world where sensitive lakes occur downstream of major emissions of mineral acids. Localised problems exist in all industrial nations using large quantities of fossil fuels, but the major impacts currently occur in the Eastern USA, Canada, Central Europe and Scandinavia. Future problems may occur in China, Africa and other industrialising regions.
Table 7.4 pH and acid neutralising capacity (ANC) in lakes of Eastern Canada

<table>
<thead>
<tr>
<th>Region</th>
<th>Sample (n)</th>
<th>Minimum pH</th>
<th>Median pH</th>
<th>Percentage of pH</th>
<th>Acid neutralising capacity</th>
<th>Median (µeq l⁻¹)</th>
<th>Percentage</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>≤ pH 5.0</td>
<td>≤ pH 5.5</td>
<td>≤ pH 6.0</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>North West Ontario</td>
<td>1,080</td>
<td>5.03</td>
<td>7.10</td>
<td>0</td>
<td>0</td>
<td>4</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>≤ 5.5</td>
<td>≤ 6.0</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>0</td>
<td>4</td>
<td>1,078</td>
<td>-3</td>
<td>254</td>
<td>1</td>
</tr>
<tr>
<td>North East Ontario</td>
<td>1,820</td>
<td>4.00</td>
<td>6.85</td>
<td>6</td>
<td>10</td>
<td>20</td>
<td>1,805</td>
</tr>
<tr>
<td></td>
<td></td>
<td>≤ 4.5</td>
<td>≤ 6.0</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>6</td>
<td>10</td>
<td>1,805</td>
<td>-99</td>
<td>165</td>
<td>9</td>
</tr>
<tr>
<td>South Central Ontario</td>
<td>1,619</td>
<td>4.22</td>
<td>6.29</td>
<td>2</td>
<td>8</td>
<td>28</td>
<td>1,578</td>
</tr>
<tr>
<td></td>
<td></td>
<td>≤ 4.5</td>
<td>≤ 6.0</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>2</td>
<td>8</td>
<td>1,578</td>
<td>-66</td>
<td>63</td>
<td>3</td>
</tr>
<tr>
<td>Quebec</td>
<td>434</td>
<td>4.40</td>
<td>6.33</td>
<td>3</td>
<td>12</td>
<td>30</td>
<td>429</td>
</tr>
<tr>
<td></td>
<td></td>
<td>≤ 4.5</td>
<td>≤ 6.0</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>3</td>
<td>12</td>
<td>429</td>
<td>0</td>
<td>36</td>
<td>1</td>
</tr>
<tr>
<td>Labrador</td>
<td>198</td>
<td>4.84</td>
<td>6.36</td>
<td>1</td>
<td>2</td>
<td>20</td>
<td>182</td>
</tr>
<tr>
<td></td>
<td></td>
<td>≤ 4.5</td>
<td>≤ 6.0</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>1</td>
<td>2</td>
<td>182</td>
<td>2</td>
<td>46</td>
<td>0</td>
</tr>
<tr>
<td>New Brunswick</td>
<td>84</td>
<td>4.51</td>
<td>6.34</td>
<td>7</td>
<td>15</td>
<td>36</td>
<td>81</td>
</tr>
<tr>
<td></td>
<td></td>
<td>≤ 4.5</td>
<td>≤ 6.0</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>7</td>
<td>15</td>
<td>81</td>
<td>-22</td>
<td>38</td>
<td>11</td>
</tr>
<tr>
<td>Nova Scotia</td>
<td>232</td>
<td>4.20</td>
<td>5.20</td>
<td>39</td>
<td>63</td>
<td>82</td>
<td>198</td>
</tr>
<tr>
<td></td>
<td></td>
<td>≤ 4.5</td>
<td>≤ 6.0</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>39</td>
<td>63</td>
<td>198</td>
<td>-82</td>
<td>0</td>
<td>51</td>
</tr>
<tr>
<td>Newfoundland</td>
<td>270</td>
<td>4.84</td>
<td>6.17</td>
<td>3</td>
<td>14</td>
<td>44</td>
<td>176</td>
</tr>
<tr>
<td></td>
<td></td>
<td>≤ 4.5</td>
<td>≤ 6.0</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>3</td>
<td>14</td>
<td>176</td>
<td>-16</td>
<td>28</td>
<td>7</td>
</tr>
</tbody>
</table>

Source: Modified from Jeffries et al., 1986
Figure 7.10 Map of soil sensitivity to acid deposition in Eastern Canada (compare with Figure 7.11). (Analysis and figure provided by Mr. A. Fraser, National Water Research Institute, Environment Canada)

Figure 7.11 Map of wet sulphate ($SO_4$) deposition in Eastern Canada averaged for the years 1982-1986 highlighting areas of high deposition (> 20 kg ha$^{-1}$) where lakes have been shown to be subject to progressive acidification. (Analysis and figure provided by Mr. A. Fraser, National Water Research Institute, Environment Canada)
7.3.5 Salinisation

Salinisation is becoming a widespread water quality issue (Meybeck et al., 1989) which often leads to the loss of usable water resources, including lake waters. In virtually all
cases this is due to the mismanagement of the water for agricultural purposes. Natural Salinisation occurs over long periods of time. In its simplest form it is caused by a change in the natural water balance of the lake, i.e. when the output is less than the input and the water balance is maintained by high levels of evaporation (Meybeck, 1995). This leads to a progressive increase in salt content, as can be seen in the Dead Sea, where salt concentrations reach 300 g l\(^{-1}\).

The input of saline groundwater, which represents only a proportion of the total water balance, results in a meromictic condition for normal lakes. However, in shallow lakes, mixing may occur leading to the creation of lakes with brackish water or moderate salt composition. This condition often exists in coastal lakes, particularly when over-use of fresh groundwater allows marine salt water intrusion. Temporary saline lakes and saline groundwaters may also occur following marine flooding resulting from storm surges in flat, low-lying areas. With time, the normal water balance is restored by freshwater dilution, but flooding with saline water can always re-occur.

By far the most common form of lake Salinisation is related to changes in the water balance and salt content caused by irrigation. Diversion of water from a lake to the land surface results in: i) loss of water during irrigation due to infiltration to groundwater and loss by evaporation and evapotranspiration, and ii) increased salt concentration in the water returning to the lake or river as a result of soil leaching processes. When combined, these produce a slow Salinisation of the lake water. However, the process may be greatly accelerated if water losses during lake use fall below the water output rate of the lake. In this case lake levels also fall indicating that the lake water is being over-used. Control and diversion of the inlet waters to the Sea of Aral, Russia from the Amu Darya and Syr Darya rivers (for the purpose of large scale irrigation) have resulted in a disastrous lowering of the lake level. This has caused major disruption to other water uses, such as navigation, fisheries and water supply and has increased the salinity of the water due to the effects of uncompensated, high levels of evaporation.

7.3.6 Issue summary

The issues discussed above indicate that major impairment of water use can occur if efforts are not made to control or prevent the causes of the associated water degradation. A summary of the issues, causes and effects, and the associated requirements for the three levels of water quality assessment discussed in Chapter 1 are given in Table 7.5. To understand the degree to which a water body has been affected by any specific use or issue, some investigative monitoring and assessment is required. This assessment process, if undertaken in sufficient detail, provides the basis for the infrastructure required to design an appropriate control strategy. Once this strategy has been implemented, the assessment must be continued to ensure that the expected results, or improvements, in water quality are actually achieved. The assessment strategies and variables to be measured are discussed briefly in section 7.5 and in more detail in Chapters 2 and 3.

7.4. The application of sediment studies in lakes

The role of sediment studies in limnology has been a rapidly developing field over the last two decades. This importance is reflected in this book by the incorporation of a separate chapter dedicated to the monitoring and assessment of fluvial and lacustrine
Sediments (see Chapter 4). Therefore, the discussions included in this chapter relate only to special considerations for sediment studies within the limnological context. For more detail see Chapter 4.

Sediments interact with lake water and soluble constituents in such a manner that they give many unique insights into limnological processes. As a consequence, they provide important basic information on the geochemical origin, dispersion throughout the lake, and limnological fate of soluble and particulate constituents.

Sediment texture and mineralogy are closely inter-related. Finer particles of less than 64 um consist mostly of clays, hydrated oxides of iron and manganese, and organic matter. These elements provide the geochemically active sites which allow for the uptake and release of chemical elements and compounds. Adsorption or desorption occur during transport and sorting, and are related to the solid/liquid concentration as a function of solubility, for the range of pH and redox conditions that occur in the transporting medium. A high solid contaminant concentration in equilibrium with river water, results in an initial desorption of the contaminant when entering the more dilute conditions of normal lake water. However, resuspension of fine bottom sediment occurs under storm conditions in a shallow lake, resulting in adsorption of elements from the water column and elimination under uniform settling conditions. Once settling has occurred, the contaminant (under normal pH and in oxygenated bottom waters) is effectively removed from the water column. Release back to the bottom water can occur under the following conditions:

- **Resuspension**: Physical disturbance of bed sediment and release of interstitial waters, including possible desorption of some contaminants to the water column.

- **Bioturbation**: The disturbance of sediment by benthic organisms leading to a re-distribution of the contaminants in the deeper sediment to the surface layers where they may be released to the water column (by the same processes as in resuspension).

- **Changes in pH and Eh**: These change the mineralogical composition of the sediment and release elements by solubilisation (e.g. release of adsorbed P and metals by reaction and solubilisation of iron (Fe$^{3+}$ $\Rightarrow$ Fe$^{2+}$) resulting from oxygen deficiency during bottom water anoxia).

- **Biological transformation**: Bacterial modification of trace elements such as Hg, As, Se and Pb by conversion to soluble or volatile organo-metallic complexes.

- **Changes in lake water concentration**: Reduction in water concentration by management control (e.g. phosphorus control) which changes the equilibrium concentrations and results in release from sediments, producing a new equilibrium condition.
Table 7.5 A summary of water quality issues, causes and effects in lakes together with the principal monitoring and assessment activities according to various levels of sophistication

<table>
<thead>
<tr>
<th>Water quality issues</th>
<th>Causes</th>
<th>Water quality effects</th>
<th>Water quality assessment level(^1)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>I</td>
</tr>
<tr>
<td>Safety</td>
<td></td>
<td></td>
<td>(I) II</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>(I+II)+III</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>------------------------------------------------------------------------------------------------------</td>
</tr>
<tr>
<td>Eutrophication</td>
<td>Excess nutrients</td>
<td>Increased algal production</td>
<td>Estimates of biomass Cell counting Chlorophyll a during stratification Transparency</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Oxygen depletion in hypolimnion</td>
<td>Analysis of TP, SRP at frequent intervals Analysis of sources of P Vertical profiling + O₂ measur. Species composition</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Release of Fe, Mn, NH₄ and metals in hypolim.</td>
<td>------------------------------------------------------------------------------------------------------</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Loss of biotic diversity at all trophic levels</td>
<td>------------------------------------------------------------------------------------------------------</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Full nutrient budget Vertical and spatial analysis Fe, Mn and other metal analyses</td>
</tr>
<tr>
<td>Health effects due to community waste</td>
<td>Human and animal organic waste</td>
<td>Bacterial and viral infection</td>
<td>Bacterial counts Analysis of metals Toxicity bioassay</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Other effects as for eutrophication</td>
<td>------------------------------------------------------------------------------------------------------</td>
</tr>
<tr>
<td>Acidification</td>
<td>Atmospheric deposition of sulphate and nitrogen oxides</td>
<td>Decrease in pH Increases in Al and heavy metals Loss of biota</td>
<td>Measure of pH Measure of Σ cations and anions</td>
</tr>
<tr>
<td>Toxic pollution</td>
<td>Industrial waste disposal; Agricultural and municipal</td>
<td>Increased concentration of metals and organic pollutants in water sediment and biota Bioaccumulation and biomagnification</td>
<td>Simple bioassay e.g. <em>Daphnia magna</em>; Microtox</td>
</tr>
<tr>
<td>Salinisation</td>
<td>Changes in water balance; marine incursions; increased salt to soil leaching; mainly irrigation</td>
<td>Increasing salt concentrations</td>
<td>Conductivity Lake level measurement Analysis of ionic content of water</td>
</tr>
</tbody>
</table>

\(^1\) I, II, III Simple, intermediate and advanced level monitoring and assessment

- P Phosphorus
- TP Total phosphorus
- SRP Soluble reactive phosphorus
Hydrophobic trace elements and compounds are adsorbed rapidly by fine sediment particles. Analysis of the fine particles throughout the lake and the observed concentration gradients identify the source of the element of interest. When the source is a river input, the same techniques applied to the river sediment allow the isolation and identification of the direct source of the pollutant. A fine-grain sediment pollutant concentration throughout the whole lake defines the transportation pathway within the lake system, and identifies the final “sink” regions.

The analysis of sediment at source (i.e. river inputs, coastal erosion, atmospheric deposition), together with estimates of deposition rates, allows for the computation of mass balances. Such mass balances have proved to be of considerable value in optimising control strategies by targeting the most amenable and cost effective sources to be controlled.

This summary of the use of sediment information in understanding water quality issues in lakes has demonstrated the need to include sediment measurements and sampling into lake monitoring and assessment programmes. However, the subject area is specialised and, ideally, studies should be undertaken by trained sedimentologists and geochemists. If this is not possible their advice and guidance should always be sought.

### 7.5. Assessment strategies

Appropriate monitoring strategies must be selected in relation to the objectives of the assessment programme (see Chapter 2). Sediment sampling has been discussed in Chapter 4 and is, therefore, not covered here. Water sampling is usually comparatively straightforward, although certain factors must always be taken into account. These factors include obtaining an adequate sample volume for all necessary analyses, cleanliness of samples and sample bottles, requirements for filtration of the sample and sample storage methods, etc. (see Chapter 3 and Bartram and Ballance (1996)). The measurement of phytoplankton chlorophyll, as an indication of algal biomass, is the most common routine biological monitoring strategy applied in lakes.

#### 7.5.1 Sampling site location and frequency of sampling

The structure of the lake must be defined if a rational sampling design is to be used. Depth and temperature profiles must be obtained to establish the location of the thermocline (see later). The number of samples to be taken is related to lake size and morphology and to the water quality issue being addressed. Large lakes must be described, even in the simplest context, by more than one sample. For example, Lake Baikal in Russia requires a minimum of two sampling stations due to the occurrence of two sub-basins, and because the major fluvial input and output occur in the southern part of the lake. Consequently, the northern section of Lake Baikal has a significantly different physical and chemical structure. Lake Erie consists of three distinct basins and, therefore, a minimum of three stations is required for initial characterisation. As a general rule, samples should be taken from each section of a lake which can be regarded as a homogeneous water mass. A small lake with a single water mass may be described by a single sampling station.

If only one sample is taken, it should be located at the deepest part of the lake where oxygen deficits are likely to be greatest. A number of samples (3-15) should be taken...
throughout the water column, and there should be at least three sampling occasions per year. For temperate lakes, samples must be taken prior to spring stratification, late in the summer stratification and after the autumn overturn. In a large lake, such as Lake Ontario, many water masses may be identified, each of which can be represented by a single sampling station (El Shaarawi and Kwiatkowski, 1977). To identify the homogeneous water masses in Lake Ontario, statistical analyses were carried out on water quality surveys repeated at about one hundred stations. This kind of intensive water quality survey cannot be conducted without access to considerable resources (i.e. finance, man-power, equipment) and it is, therefore, necessary to use a logical sampling design based on purpose and limnological knowledge. Theoretical examples are described below.

7.5.2 Examples of sampling programme design

For the purposes of this guidebook sampling design is best described in relation to a number of precise water quality issues. Control and resolution of these issues require a clear understanding of the sources of materials and pollutants to a lake. Sources of pollution in water bodies are discussed in Chapter 1 and summarised in Table 1.4. An indication of the complexity of monitoring and assessment with respect to lakes is given (according to the three levels of complexity discussed in Chapter 1) in Table 7.5. Variables which should be measured for major uses of the water and major sources of pollution are suggested in Tables 3.7 to 3.10.

Eutrophication

Assessment of trends in eutrophication usually includes the monitoring of nutrients, major chemical ions and chlorophyll. A single station only is required in small lakes (Figure 7.14A), whereas a single station within each homogeneous sector is best for large lakes (Figure 7.14A). Samples should be taken frequently, ideally once a week but not less than monthly. These enable the understanding of seasonal changes and the computation of mean values (by season or by year). Samples should be taken using strings of bottles in relation to the physical structure of the lake (Figure 7.14B). For phytoplankton samples and associated chemical parameters, depth integrated samples of the epilimnion may be adequate. The results obtained can be used to illustrate the spatial and temporal variability of concentrations in lakes and to make comparisons of depth profiles over a season.

Spatial and temporal variations in silica (a major nutrient for diatoms) in Lake Geneva are shown in Figure 7.15. In February silica shows vertical homogeneity in the upper 250 m and a positive increase near the sediments due to a partial lake overturn. In May silica concentrations decrease in the top 5 m due to uptake by diatoms during a spring bloom. By July the silica is nearly exhausted in the epilimnion, but concentrations start to recover in October with decreased diatom activity. Concentrations become homogeneous throughout the water column in December due to the complete lake overturn. An alternative way of presenting similar seasonal depth data is by iso-contour lines as in Figures 7.8 and 7.19.
Input - output budgets

Assessment of the influence of a lake on the throughput of materials in a river-lake system may be accomplished by frequent sampling and analysis, close to the input and at the output of the lake (Figure 7.14A). Samples should be depth integrated and calculated as mass input per year and compared to mass output over the same time scale. The addition of open lake samples provides sufficient information to calculate material mass balances for the lake. It must be recognised, however, that such mass balances must take account of all sources, including the atmosphere, and the significant role in recycling played by the sediments. Input-output measurements for nutrients and polluting substances are particularly important for verifying calculated predictions based on mathematical models.

Figure 7.14 Theoretical examples of sample site location (A) and sampling strategies (B) in lakes
Impacts of point sources

The localised effects of a discharge can be assessed by the random sampling of a scatter of points around the zone of influence of a point source, such as an industrial effluent or municipal sewage discharge pipe (Figure 7.14A). The effects of industrial outfalls should be determined by sampling and analysing for the materials likely to be discharged from the industrial process, whereas the effects of sewage outfalls should be studied by measurements of bacterial populations, nutrients and toxic pollutants.

Figure 7.15 Seasonal variation in the vertical profile of dissolved silica in Lake Geneva. Note the break in the depth scale. (Data from the International Commission for the Protection of the Waters of Lake Geneva (CIPEL, 1984) and M. Meybeck, Université Pierre et Marie Curie, Paris (unpublished data))

Comprehensive source assessment in lake basins

Comprehensive source assessment involves monitoring to determine the proportional contribution of different sources of specific materials. Frequent analysis of point and non-point (diffuse) sources is required to determine loads, and any changes in loads resulting from management actions. All rivers, as well as point and diffuse sources of industrial, urban, agricultural and natural materials (including the atmosphere), must be analysed to determine the complete impact of all human activities on water quality. Background conditions must also be established so that the anthropogenic contribution can be determined. Samples should be analysed at least monthly, but preferably with greater frequency for bacteria, nutrients and toxic pollutants. When combined with detailed, open lake monitoring and sediment analysis this approach is the most advanced and sophisticated with respect to lacustrine systems.

7.6. Approaches to lake assessment: case studies

Water quality assessments are conducted to define the conditions of the lake, usually with respect to its uses. Nutrients, trophic status and toxic substances must generally be evaluated and a management strategy designed to maintain, or enhance, the use of the lake. Once implemented, assessments must be continued to ensure that goals are being
achieved and water uses maintained. Some examples of this kind of assessment are given below.

There are many examples of the assessment of lakes in the published literature. However, few good examples exist from the developing world and, as is so often the case, most studies have been carried out in the western, industrial nations, particularly Scandinavia, North America and Western Europe. The bias towards wealthy nations provides good examples in the northern temperate zone and little detailed assessment information from elsewhere. The type of assessment undertaken is related to the resources of the nation carrying out the study. Consequently, examples of sophisticated biological monitoring for trace pollutants are usually restricted to the industrial nations. For these reasons, the following examples are taken primarily from the industrialised regions of the world, but it is hoped that they will encourage the development of increasingly sophisticated lake assessments in the developing regions.

7.6.1 General basic surveys

From the discussions above, it is clear that some primary information is required before a fairly detailed understanding of lake processes can be obtained. Two major lake characteristics must be established, bathymetry and thermal structure. Lake bathymetry is required:

• to check the occurrence of several sub-basins and to compute lake volumes for residence time calculations,

• as a basis for sediment mapping and sediment and water sampling and,

• as a basis for predicting the occurrence of specific organisms that may be required for future analysis.

Bathymetry is most conveniently obtained using a recording echo-sounder along straight lines run at a constant speed between known shore stations. Large lakes require the use of electronic positioning equipment and in these situations charting bathymetry is, therefore, expensive to perform. Small lakes may be surveyed using a measured, weighted line and the positions of each measurement determined from two or more angles to known targets, or land marks, on the shore. The use of a sextant is the traditional method of establishing position for each sounding. An example of lake bathymetry for Lake Vättern, Sweden is given in Figure 7.16.

The thermal structure of lakes has been discussed in detail in section 7.2.3. An understanding of lake processes cannot be achieved without a relatively detailed evaluation of the thermal structure. This can be done simply by rapid measurement of the temperature of water samples taken throughout the water column, either by a single bottle lowered to different depths, by several bottles hung in series at fixed depths on a single cable, or by electronic depth and temperature profiling. Water bottles are closed by trigger weights (messengers) once the line has been set, and in some cases the bottles are equipped with reversing thermometers for in situ temperature measurement. Depending on the lake size, survey requirement and the availability of resources, temperature profiles may be taken at many stations or at a single point. An example of many profiles taken during a single survey cruise in Lake Ontario is given in Figure 7.17.
However, in order to reconstruct the thermal characteristics of Lake Ontario such profiles must be taken many times during the year. An example of a single station thermal profile taken many times in order to determine seasonal thermal structure is given in Figure 7.3.

Figure 7.16 Bathymetry of Lake Vättern, Sweden (After Häkanson and Ahl, 1976)
7.6.2 Assessment of trends

The simplest form of assessment is to establish the trend in concentrations of a nutrient, particularly phosphorus, during a period of holomixis when the lake is fully mixed and isothermal. Measurements to achieve this may be made at a single station, or more appropriately at a number of stations. Samples are normally taken in early spring before the onset of stratification. An example of this type of monitoring is given for Lake Constance from the early 1950s to 1988 in Figure 7.18. The trends are quite clear: phosphorus concentrations rise rapidly until the late 1970s when, following the large scale control of sewage effluents, a rapid decline can be observed.

The assessment of long-term trends is related directly to lake management strategies whereas short-term, or seasonal, trend monitoring is required to understand the processes occurring within the lake and to provide more detailed information on a variety of indicators of lake condition. An example of seasonal trends is given for Lake Windermere, UK which was sampled on a weekly basis (Figure 7.7). The trends in chlorophyll a indicate the algal biomass which results in a summer depletion of soluble reactive phosphorus. The depletion of soluble reactive P occurs during stratification and the regeneration occurs during the winter when the water is mixed. No clear trends can
be observed in total phosphorus concentrations, although a minor reduction occurs in summer resulting from soluble reactive P depletion. This suggests that the bulk of the total P is non-bioavailable and probably occurs bound with apatite and mineralised organic matter. The trend in NO₃-N is more interesting and indicates summer decreases probably related to algal production. In the summer (August) of 1974, nitrate was depleted almost to the point where it could have become limiting for algal growth. The seasonal trends portrayed for Lake Windermere are relatively simple because they represent trends in the surface waters.

A more complete understanding of nutrient and element cycling may be obtained through seasonal trend analysis of samples taken at many depths at one location. When the concentrations are contoured, as shown in Figure 7.19, they show a high degree of resolution in the seasonal development of the variables measured. This example from the Heiligensee, Germany summarises the seasonal cycling of O₂, H₂S, soluble reactive P and NH₄. Oxygen is distributed throughout the water column in December at concentrations of 10 mg l⁻¹. During the winter stratification caused by the January to March ice cover, some depletion of oxygen occurs in the hypolimnion, with re-oxygenation occurring almost to the bottom of the lake in April during the spring overturn. Summer stratification from May to November is characterised by progressive declines in oxygen concentrations in the deep water. Complete anoxia occurs in August and September, during which time major release of H₂S occurs from the bottom sediment. The cycling of soluble reactive P and ammonia conform to the seasonal cycle of oxygen, with the major release of soluble reactive P and NH₄ from the bottom sediments occurring at the time of oxygen depletion. Complete vertical mixing occurs during the autumn overturn in December, producing constant concentrations throughout the water column during the winter. This lake is hypereutrophic.
7.6.3 Spatial surveys of trace constituents

The distribution of contaminants in a lake can be determined by spatial surveys. These surveys can highlight areas in the lake which might be susceptible to possible effects of the contamination. The distribution of the metals zinc, lead and copper in Lake Ontario is given in Figure 7.20. Analyses were performed on unfiltered water from samples which were grouped into zones of relative homogeneity. The metals Zn and Pb were grouped into three zones and Cu into four. The groups are numbered sequentially from the highest mean concentration in zone 1 to the lowest in zones 3 and 4. Mean values for each element are given in the table included in Figure 7.20. The distribution of the zones shows an inshore to offshore decline in the mean values of the three elements which probably relates to a decrease in particle concentrations. Zone one occurs in the extreme west end of the lake for all metals and extends to the north west shore for Zn and Pb, and also occurs on the south shore (just east of the central part of the lake) for
Pb. The occurrence of zone 1 on the western shore is due to the steel manufacturing complex at Hamilton. The Pb on the southern shore is related to the presence of the city of Rochester, New York, USA and the north shore occurrence of high mean concentrations of Pb is adjacent to the city of Toronto. Zone 1, therefore, can be ascribed to anthropogenic sources of these elements, whereas the other zones reflect the mixing processes involved in the dispersion of these elements throughout the lake. Similar distributions for organic pollutants in the water of Lake Ontario have also been produced (Thomas et al., 1988).

7.6.4 Biological monitoring

With the increasing concern over the potential impacts of toxic substances on the health of human populations, biological monitoring (see Chapter 5) has become an extremely important component of lake assessment programmes.

Examples of the analysis of toxic compounds in the tissues of organisms are available from the Great Lakes, as shown in Figure 7.9. These illustrate the biomagnification of pollutant concentrations within the food chain of Lake Ontario (for more details on bioaccumulation and biomagnification see section 5.8).

Organisms from a number of trophic levels in lakes may be used to assess the spatial and temporal variation of specific pollutants. Concentrations of contaminants in fish are normally used for these studies. An example of probably the most successful biological monitoring programme undertaken in North America is given in Figure 7.21A. In this programme, a number of organic pollutants were analysed each year in the eggs of the Herring gull taken from bird colonies scattered throughout the Great Lakes. The Herring gull feeds on fish from the Great Lakes. The analysis of Herring gull eggs from Lake Ontario for PCB and 2378-TCDD (polychlorinated dibenzo-p-dioxins, particularly the “2378” tetra chlorinated form “2378-TCDD”) showed a clear downward trend between 1971 and 1984. Figure 7.21B summarises the results for 1983 for PCBs and 2378-TCDD. These compounds occur throughout the lakes but particularly high concentrations have been found at Green Bay, Lake Michigan, Saginaw Bay in Lake Huron and at the outlet of Lake Ontario to the St. Lawrence river.
Figure 7.20 Distribution of total Zn, Pb and Cu in the waters of Lake Ontario together with the mean values for each zone. Zone 1 is the most polluted and zone 4 the least polluted (After Thomas et al., 1988)

<table>
<thead>
<tr>
<th>Zone</th>
<th>1</th>
<th>2</th>
<th>3</th>
<th>4</th>
</tr>
</thead>
<tbody>
<tr>
<td>Zn</td>
<td>4.4</td>
<td>1.4</td>
<td>0.9</td>
<td>-</td>
</tr>
<tr>
<td>Pb</td>
<td>0.76</td>
<td>0.53</td>
<td>0.31</td>
<td>-</td>
</tr>
<tr>
<td>Cu</td>
<td>2.33</td>
<td>1.71</td>
<td>1.47</td>
<td>1.12</td>
</tr>
</tbody>
</table>
7.6.5 Sediment monitoring

Many examples of the use of sediment analyses to determine spatial and temporal distributions of various chemicals have been provided in Chapter 4. The spatial and temporal distribution of mercury in lake sediments has been combined into a single illustration by Häkanson and Jansson (1983) for Lake Ekoln in Sweden (Figure 7.22). The Hg distribution patterns in the lake indicated the major source as the River Fyris. Vertical profiles at the three core sites showed that Hg concentrations started to increase towards the end of the last century, around 1890, and have been rising up to the present time. The core from Graneberg Bay showed wide fluctuations in concentration since 1930 over the 35 cm of sediment thickness. However, these oscillations have been damped out in the other two cores. The source of Hg, as represented by the cores, was relatively constant over 40 years but remained at sufficiently high concentrations to give a continuous increase in concentrations further away from the source.
7.7. Summary and conclusions

Lakes are an essential source of freshwater for human populations. They provide water for a multitude of uses ranging from recreation and fisheries through to power generation, industry and waste disposal. As a consequence of the latter two uses most lakes have suffered water quality degradation to some extent or other. Degradation in the form of bacteriological pollution (health related), eutrophication, gradual increases in toxic pollutants and salinisation have all been observed. Even remote lakes are currently suffering degradation in water quality due to atmospheric deposition of mineral acids, nutrients and toxic chemicals.

The control of lake water quality is based on sound management practice (Jorgensen and Vollenweider, 1989) in relation to the required water uses and a reasonable, preferably detailed, knowledge of the limnological characteristics and processes of the water body of interest. The basic information required to characterise the ambient condition of a lake is based upon surveys of the various components of the lake system. These surveys provide the scientific basis for the development of the lake in order to meet the most specific requirements, and sensitive uses, to which it is subjected. Once this has been established a management protocol can be determined and implemented.
Figure 7.22 Spatial distribution of mercury concentrations in the sediments of Lake Ekoln, Sweden together with the temporal trends indicated by three sediment cores (After Häkanson and Jansson, 1983)

An assessment programme must be put in place right from the time of initial implementation of a specific lake management programme, in order to ensure that the lake water quality is adequate for the requirement of the management plan. Examples of such requirements include: increasing pH in an acidified lake; low bacterial counts to ensure potable water and contact recreation; appropriate phosphorus concentrations to maintain the desired levels of lake productivity; inputs of toxic substances which are low enough to ensure the concentrations in fish do not exceed a required guideline level; and preservation of all of the multitude of water uses required by human populations.

By undertaking properly designed lake assessment programmes, those agencies responsible for water management in the newly industrialising regions of the world will benefit from the lessons learned from an abundance of past errors in water quality management in industrialised countries.
7.8. References


Fraser, A.S. 1990 Personal Communication, Rivers Research Branch, National Water Research Institute, Canada Centre for Inland Waters, Box 5050, Burlington, Ontario, Canada.


