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# LAKE AND RESERVOIR MANAGEMENT

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# PREFACE

*Milan Straškraba died during the preparation of this book in Denver on 26th of July year 2000. We, his co-authors, not only lost a co-author but a unique friend, who in addition had an immense holistic knowledge about lakes, reservoirs and ecosystem properties and reactions. We will never forget his unique personality and his valuable contributions to our scientific discipline. We have therefore dedicated this book to Milan Straškraba's memory.*

*The authors are listed alphabetically to emphasize that all four main authors have contributed equally to the volume.*

Both the United Nations Environmental Program and World Health Organization supported the initial development of this book, and their support is gratefully acknowledged. Nevertheless, the views expressed in this book are solely the views of the team of authors who prepared it, and do not necessarily reflect the views or policies of either of the above-noted organizations.

This new volume was developed primarily (though not exclusively) for three groups of potential readers, including (i) biologists, chemists and engineers working in water quality laboratories; (ii) managers responsible for deciding on actions to be taken regarding lake and reservoir management; and (iii) local officials and politicians dealing with the environment, particularly its water components. Reading some of the more general parts of this volume also will be useful for the representatives of public organizations dealing with lakes or reservoirs. Differential reading of this volume by these three groups may be most efficient in its use. Biologists, chemists and engineers might be served most efficiently by starting with Chapter 1 and going through successively to Chapter 10, considering the most important conclusions. Water supply managers may wish to start with Chapter 10 as a technical outline, and subsequently select the chapters with more information on the subjects in which they are most interested. The last group of potential readers, with the broadest interests, may perhaps best start with Chapter 8, as a means of first getting a nontechnical overview of lake and reservoir water quality management. If they wish to explore a given topic in more detail, they may then proceed to the more technically-oriented recommendations contained in Chapter 10.

The presentations of lake and reservoir water uses and abuses (Chapter 2), monitoring of lake and reservoir water quality, emphasizing result evaluations (Chapter 3), preventive and corrective management measures in the waterbody and its drainage basin (Chapter 4) and the use of mathematical for management (Chapter 5) are of common interest in regard to both lakes and reservoirs, and are more technical in nature than the remaining chapters

of the book. Problems more specific to reservoirs are dealt with in Chapter 6, which focuses on reservoir management and considerations for constructing new reservoirs. Based on the more general problems, the development of water quality management strategies, the information regarding examples of poor lake and reservoir management, and an introduction to the problems of integrated management are also highlighted (Chapter 7). Several examples of lakes and reservoirs in very different natural and social environments, and discussion of their problems and management solutions are presented as case studies (Chapter 9).

The present volume represents a collaborative effort, with each chapter identifying a primary author and the additional contributing authors. All the co-authors of the book have commented on some or all portions of this book, and contributed various inputs (text, ideas, information) that shaped its final form. It is hoped that this process has produced a mature volume, with a relatively uniform structure and balanced presentation focusing on progress made in lake and reservoir water quality management. I wish to thank all the co-authors for their patience and understanding regarding the complexities involved in the organization and operation of such a heterogeneous group, and particularly for their willingness to adapt to the changes in scope and structure of the book as it has evolved from the initial intention to concentrate on a more scientifically-oriented, independent lake and reservoir management monograph, to the combination of scientific and management-oriented focus of the final document.

S.E. Jørgensen

*Chapter 1*

# LAKES AND RESERVOIRS AS WATER RESOURCES

## 1.1 INTRODUCTION

Many features of natural lakes and artificial lakes (reservoirs) are similar, and the approaches for their use and management also bear great similarity. Therefore, when the words "lakes" or "waterbodies" (without specification) are mentioned in this book, it refers to both natural lakes and reservoirs. However, because reservoirs and natural lakes also have a number of characteristics that differ significantly (Rast and Straškraba, 2000; also see Chapter 6), reservoirs and natural lakes will be identified specifically in this volume where it is appropriate.

This introductory chapter discusses the physical characteristics of lakes and reservoirs, and the roles they play for society. Section 1.2 presents their role from the perspective of how they are perceived by the public (Section 1.2.1), the roles they play for society (Section 1.2.2) and their economic value to humanity (Section 1.2.3). The opposite perspective also is discussed in this chapter; namely, what is the role of society in regard to managing lakes and reservoirs for their sustainable use for humanity and for nature (Section 1.2.4)?

Section 1.3 reviews the distribution of lakes and reservoirs, focusing on their water volumes and surface areas (Section 1.3.1), their geographical distribution, and how their geographic location is reflect in their water quality and related management issues (Section 1.3.2).

## 1.2 THE ROLES OF LAKES AND RESERVOIRS

The large number of reservoirs constructed throughout the world demonstrates that lakes are needed for many purposes, and that they play a major role for society (Thornton et al., 1996; World Commission on Dams, 2000; Rast, 2003). The main need is for greater quantities of continuously available and easily accessible quantities of water of good quality, primarily for its value for meeting the water needs of the population, agriculture and industry.

### *1.2.1 Need, Uses and Public Perceptions of Lakes and Reservoirs*

Lakes and reservoirs are basins filled with water that is often (in the case of natural lakes) and always (in the case of reservoirs) used by humans. Why do humans need such large quantities of water? For daily consumption (drinking), humans only need about two liters of water daily. The remaining quantity is used for human hygiene, for the preparation of

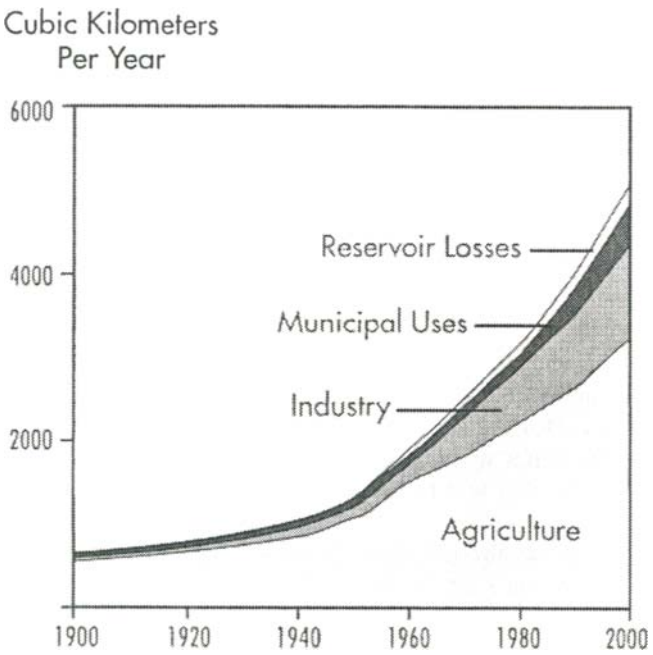



Fig. 1.1. Estimated water usage by different economic sectors for the period 1990–2000 (Atchia and Tropp, 1995).

food, and for the production of food and goods. The average daily water need per person differs significantly in different regions. It depends partly on the availability of water, and generally increases over time as the economic and cultural status of a region increases. Water availability is uneven around the world. Some regions face water shortages, including Africa, the Middle East, Australia and parts of western America. It has been estimated that total annual water use around the world increased tenfold between 1900 and 2000, from approximately 600 $km^3$ to nearly 5500 $km^3$. On a global scale (Fig. 1.1), approximately 69% of water usage is for agricultural purposes (primarily irrigation), 23% is for industrial usage, and 8% is for domestic consumption (Atchia and Tropp, 1995).

Water usage in large urban centers with centralized supplies is particularly high for various reasons, including often-large water losses to leakage from inadequate or broken pipe systems, and to its use in industry, watering of gardens, etc. A distinction is drawn between water that is consumed (i.e., water that is not returned directly to waterbodies) and water used (including water that is returned, usually in a polluted state, to waterbodies). This distinction is made because water that is consumed cannot be used more than once without having to re-enter the hydrological cycle and be returned to the Earth in the form of rainfall and snowfall. In contrast, in many countries water that is not consumed is re-used a number of times. This situation enhances the need to save water more than in situations in

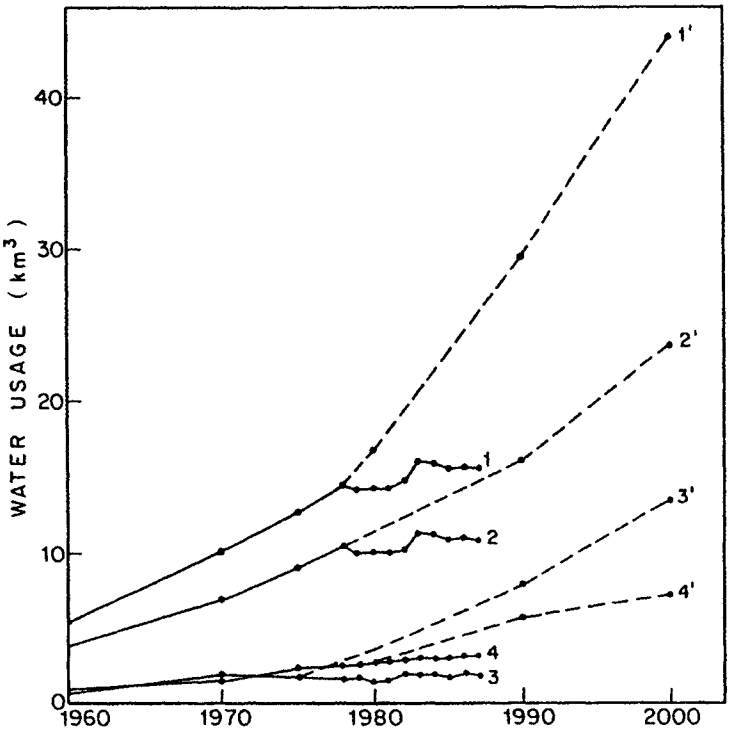



Fig. 1.2. Predicted and actual water usage in Poland, 1960–2000, showing discrepancies from 1976 (modified from Mitchell, 1990). Legend: 1—total users; 2—industry, including energy; 3—agriculture, particularly irrigation; 4—municipal sector.

which water is present in excessive quantities. In spite of rapidly increasing water needs on a global scale, it is fortunate that the rise is exponential in nature. In fact, per capita water need often decreases in developed countries to the extent that there can be a discrepancy between water needs predicted on the basis of simple extrapolation of trends and actual water consumption (Fig. 1.2).

Lakes represent a large proportion of the world's readily available water supply, following rivers. The importance of lakes in a given region depends partly on their numbers and distribution. In Scandinavia, for example, lakes occupy almost 10% of the total land area, while lakes occupy less than 1% of the total land area in China and Argentina (see Section 1.3.1).

Lake water is not used only for human consumption. Other water uses include industrial applications and processes, transportation, and generation of hydropower (particularly reservoirs), as discussed further in Chapter 2. Both the production of food and of industrial products, for example, requires water. Plants and animals consume water. Industry requires

water for purposes such as facilitating chemical reactions, cleaning and cooling (Ryding and Rast, 1989; Rast and Thornton, 1999).

The public perception of lakes and reservoirs in different regions of the world depends on the economic and cultural state of the regions. This perception is closely related to the concept of the "quality of life" (Galloping and Oberg, 1991; also see Glantz, 1999). By definition, the quality of life (QOL) is a subjective concept, dependent on cultural perspectives and values. There is a tentative consensus, however, on some of the factors considered essential for a good life, including (i) access to clean water for domestic purposes; (ii) sufficient food; (iii) adequate shelter; (iv) security; and, with respect to health, (v) access to adequate medical care. It is also suggested that education and gender equality be added to this list (Daly and Cobb, 1989).

Applied to lakes and reservoirs, the basic demands for quality of life include (i) protection from floods and drought; (ii) protection from water-based and waterborne diseases; (iii) clean water; and (iv) protection of food (protein) in lakes and reservoirs. The latter also may include satisfaction with work, such as irrigation, fisheries and production of huts, mats, boats, etc., from macrophytes (aquatic plants). Direct access to a lake or river also was a preferred condition in situations of early development, be it in historical periods for the developed regions, or more recently for the developing regions. The public perceptions of lakes, however, do not depend only on how they are used. They also have a role in the aesthetics of the natural landscape, representing important natural beauty for satisfying human needs and feelings. This is seen not only from many earlier and recent paintings of artists, but also from the recent appreciation of both residential and recreational housing along lake shorelines. The prices of such housing are typically much higher than elsewhere. The will of people to pay higher prices for water also is seen in places with good central water supply systems, and with people who utilize water for sports and recreational purposes.

Recreation along lake shores is quite desirable, as can be seen from the high concentrations of recreational housing along lake shorelines. There are many water-based recreational activities in lakes and along their shorelines. The negative effect of recreation, however, is obvious with mass tourism, which can significantly degrade landscapes along lake shorelines and which, so far, have largely been planned carelessly and often irresponsibly (Ostendorp and Krumscheid-Plankert, 1993).

People that live close to lakes, or who like visiting them, typically appreciate not only the lake scenery, but also its water quality. Good water quality is an obviously desirable trait, while degraded water is not. Particularly negatively impressions are observed among swimmers when lake water is turbid, the quantity of algae in the water is high, and where filamentous algae overgrow the shallower lake areas. Swimmers typically feel a physical disgust when such algae touch their skin or when it accumulates in their swimming apparel. Dense growths of aquatic plants can be disturbing to swimmers, as well as boaters, particularly when they become tangled in the propellers of boats. In fact, dense stands of tropical aquatic plants, such as *Eichhornia* or *Azolla*, can prevent the movement of boats. The appreciation of clean water can be simply demonstrated by the statistical patterns of visitors when they have a choice between clean lakes and lakes beset with algae and aquatic plant

problems. This is due not only to negative perceptions as cited above, but also to negative experiences with waterborne diseases and infections (Section 2.2.3) and water pollution.

A quantitative indication of the public perception of water quality is represented by the willingness of consumers to pay a premium for water. The values assigned by the buying public to bottled water, and to home sites located on "clean" lakes and reservoirs, are indicative of the complex relationship between human perceptions (i.e., the objective assessment of a situation) and human behavior (i.e., the objective response to the situation). Interestingly, Thornton and co-workers (Thornton et al., 1989; Thornton, 1993) reported that, in studies including temperate zone lakes and rivers, as well as subtropical reservoirs and lagoons, few respondents indicated any interference with their use of the waterbodies for water-based recreational purposes (e.g., their behavior), despite indications that water quality conditions may have influenced their enjoyment of the recreational experience (e.g., their perceptions).

In these and other studies (David, 1971; Kooyoomijian and Clesceri, 1974; Heiskary, 1989; Quick and Johansson, 1992; Wiseman and Sowman, 1992), public perceptions of water quality were based upon visual factors as primary determinants of water quality. In studies on African lakes, enriched reservoirs generally attracted a wider range of negative responses than other types of lakes, especially in terms of odors and tastes where the waterbodies were used for drinking water supply. In contrast, there was little unanimity, in terms of behavioral responses, with the limnological differences between waterbodies playing a part in determining user responses. In reservoirs dominated by algal growth, for example, few respondents expressed concerns over aquatic macrophyte growths. In addition, the types of water use played a role in determining human behavioral responses. In lakes where recreational boating and swimming were the primary uses, the presence of excessive quantities of algae and aquatic plants resulted in more frequent negative expressions among respondents than for lakes where recreational angling was the primary use, despite similar or even more abundant growths of plants and algae in the latter lakes. Such use-specific behavioral links were reported by Kooyoomijian and Clesceri (1974), who found that lake contact recreational users (e.g., swimmers and anglers) had more negative impressions of water quality than from noncontact recreational users (e.g., picnickers and sight-seers). Similar judgments also were identified in relation to the proximity of the respondent's residence to a waterbody, with persons living closer to a waterbody being more severe judges of water quality (David, 1971).

### *1.2.2 The Roles of Reservoirs and Lakes in Society*

The water requirements of people vary greatly around the world. Populations are even found inhabiting deserts, semi-arid regions and other localities with limited water resources. At least a limited quantity of water must be available in all such locations, however, either as surface water or groundwater, if people are to survive. The hydrological reality is that where there is no water, there is no life. This applies to all life, plants, animals and humans. Under hot dry conditions, humans will survive only few days without drinking water, and the same would be true even at the seacoast, which is characterized

by access to large quantities of water, but of high salinity, rendering it unfit for human consumption.

Throughout history, ancient civilizations flourished, particularly on or near lakes, which provided water for drinking, cooking, cultivation of plants and domestic animals, washing of clothes, a medium for transportation, and sometimes even a hydrological barrier that protected them from their enemies.

At the community and national scales, the cultural importance of water is reflected in the basic human need for it. Most large cities and, indeed, the majority of human settlements around the world, lie in close proximity to water resources. Almost all the large cities of the world lie within 100–200 km of rivers, lakes and seas. The cultural importance of waterbodies extends to the demarcation of boundaries, ranging from the most elementary moats surrounding European castles to the most elegantly negotiated international boundaries between countries. In such situations, waterbodies assume the role of protector, isolating the community from its potentially-hostile neighbors. Indeed, the construction of protective moats is one of the earliest uses of water diversionary structures. At this scale, waterbodies often have a role as landscape features, exclusive of their other societal roles (drinking water supply, irrigation and industrial water source, recreational resource, transportation corridor, etc.).

Lakes and reservoirs are landscape features with great significance for people that extends far beyond their geographic existence or their immediate economic values. This significance is both cultural and aesthetic, and much is deeply ingrained in society. Brakke (1996), Khaka (1996), Chiu (1996) and Paszkowski (1996) all relate various cautionary and fantastic tales among all cultures of the world, wherein water plays a central, and often sinister, role in society. However, such tales always contain a valuable moral that generally reinforces a positive societal characteristic deemed desirable by the culture within which it was created.

While waterbodies in folklore may have sinister overtones, waterbodies often assume a cleansing connotation in regard to their religious significance. Virtually all of the world's great religions utilize water as an essential element of their religious ceremony, both publicly and privately. This role is partly a basic recognition of water's attribute as the universal solvent. This role also recognizes the basic societal concern over personal hygiene and the universal use of water for ablutions.

Over time, the role of water and waterbodies has permeated other societal aspects, ranging from its practical scientific and technical uses and concerns to the more esoteric realms of arts and literature. Upon reflection, although the role of water in art and literature should not be too surprising, given the more fundamental role of water in religion and daily life, rivers, lakes and reservoirs form the basis for numerous works of art. Examples include the landscape works by John Constable and others, and the literature of Samuel Clemens (Mark Twain) and others.

The varied role of lakes in society not only highlights their great value, but is also the cause of water controversies and management difficulties. As an example, reservoirs originally constructed for a single purpose are becoming multipurpose in their use, with correspondingly increasing conflicts among their different users. The conflicts include the

allocation of the available water for individual uses, as well as water quality concerns. Water quality degradation by one can limit water use by others. Because of the dependence of water quantity and quality not only on proper water management, as well as landscape management (Falkenmark et al., 1999), the recent tendency is toward the integrated management of both resources (see Chapter 7) directed to air, water and land (United Nations, 1993; Thornton et al., 1999; Rast, 2003; Rast and Holland, 2003).

Climatic conditions are one important variable dictating consumptive water use. If the climate conditions in a region can be characterized by the aridity index, there is a positive relationship with consumptive water use (Fig. 1.3). The continental differences in the figure are due to the differing physiographic and economic status of the continents.

With few exceptions, most industrialized countries can provide their citizens with a sufficient supply of good quality drinking water. In contrast, measures such as river diversion (e.g., Greece, South Africa, Australia), reservoir and aqueduct construction, use of groundwater reserves, etc., often at high cost, are employed in semi-arid and arid regions. Many Middle Eastern countries (Saudi Arabia, Emirates, Israel, etc.) employ expensive electrodyalitic desalinization, and may also use solar heat to produce drinking water. In most industrial countries, high-quality water (drinking-water quality) is still used for industry, irrigation, cleaning of city streets, etc. In fact, water used for such purposes collectively in Norway, Switzerland and Austria could provide drinking water for at least half the European population. In contrast to this group of countries with abundant supplies of good quality, or the financial resources needed to provide it, about 30 developing countries face drastic deficits of safe drinking water. According to estimates, 60 more countries will face this shortcomings in the year 2000 (World Meteorological Organization, 1997). Rapidly increasing population densities, desertification and inadequate financial resources for water development are primary reasons for this situation (Rast, 1999, 2003).

On the basis of political boundaries existing prior to 1990, nearly 40% of the world population lives in international watersheds, with 23 of these watersheds being shared by 4 to 12 countries. In approximately 50 countries, 75% or more of the national territory falls within an international water basins (Shiklomanov, 1997). On a global scale, there are currently 261 international river basins (Wolf et al., 1999). Collaborative efforts to adequately managing these basins for sustainable water use is frequently obstructed by ongoing political differences, poor international relations, competition for the resource, and differences in social and economic goals. The conflicts about water resources in general, and lakes in particular, are becoming major issues in countries with water shortages. It may become the cause of local wars where countries share joint water resources (Atchia and Tropp, 1995). Water shortages are expected to increase in various locations in the future because of population growth and global climate change. The present status of water shortages for European countries, for example, and some estimates of future shifts in water demands, are given in Figure 1.4.

Some international watersheds are likely to be sources of political tensions, either presently or in the future, between the upstream and downstream countries in river basins. Former Egyptian Foreign Minister, and later U.N. Secretary-General, Boutros Boutros

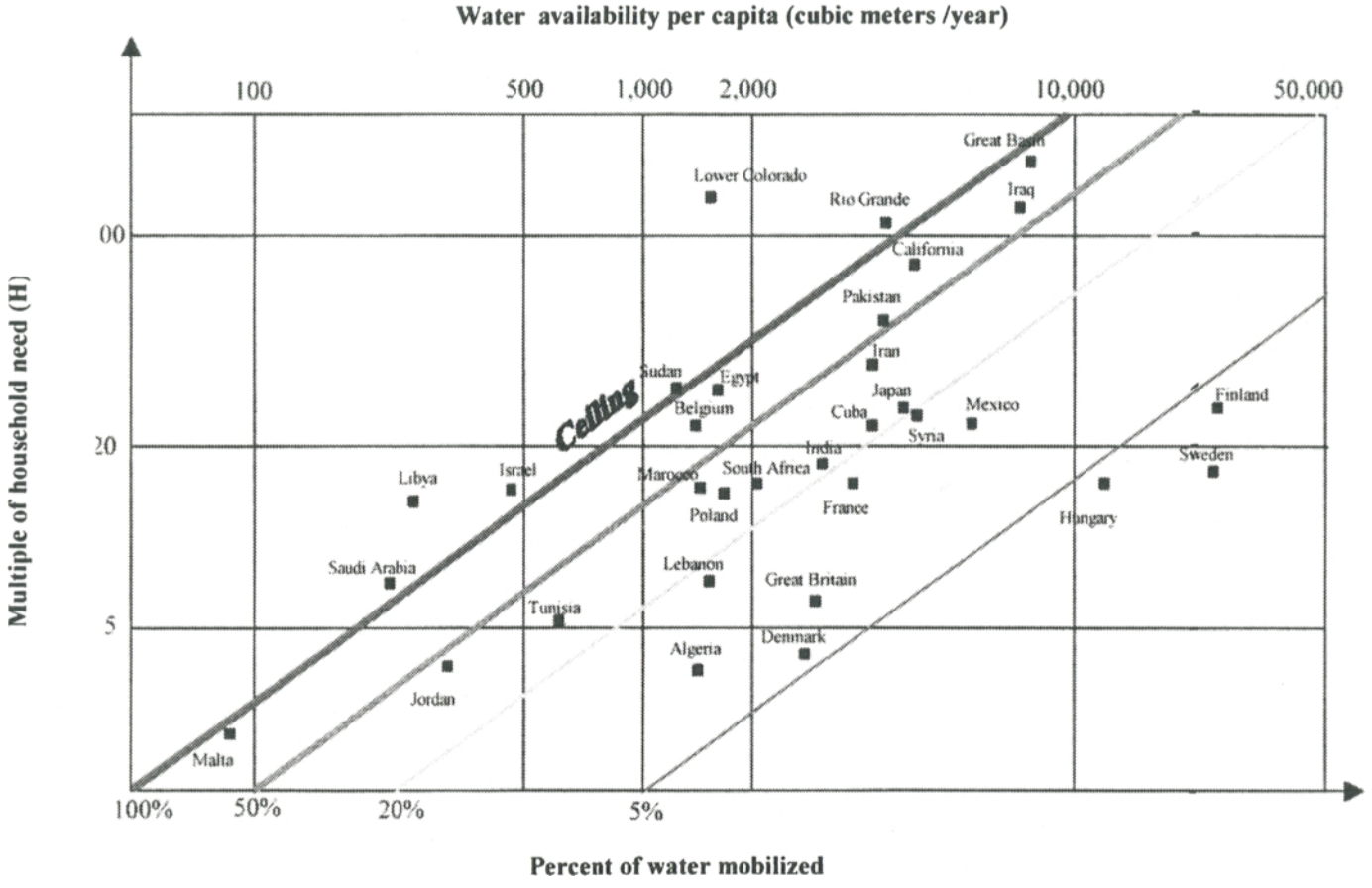



Fig. 1.3. Water availability per capita ($m^3yr^{-1}$) and percent of water utilized in different countries (Falkenmark and Widstrand, 1992).

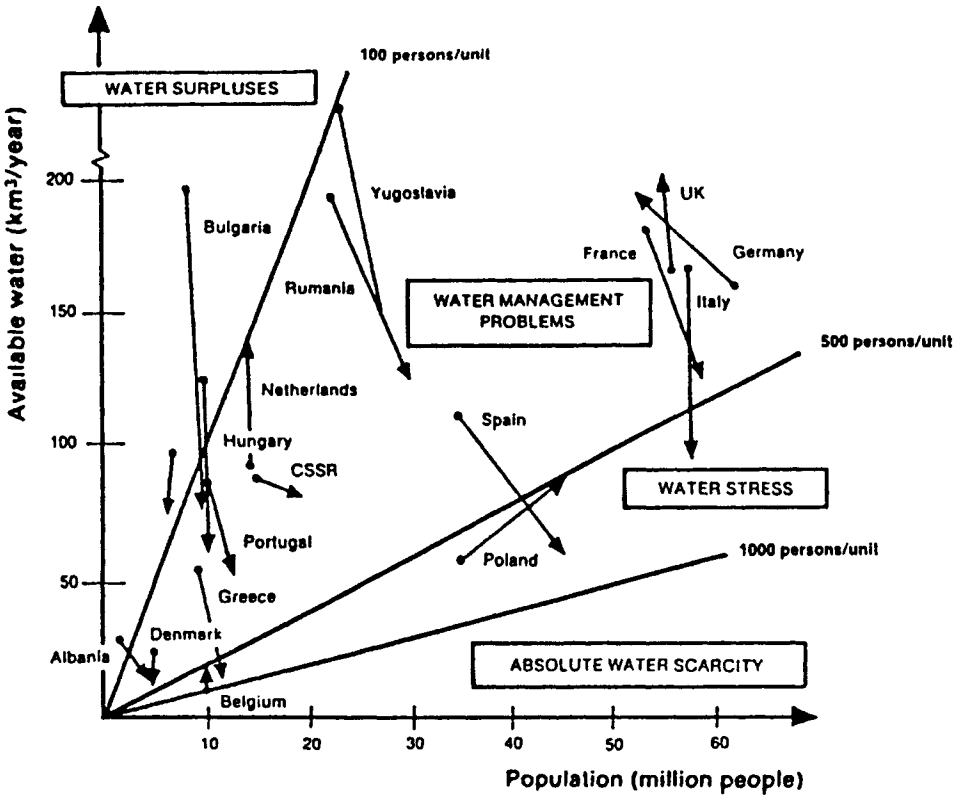



Fig. 1.4. Shifts in relative water availability of European countries in response to climate change and population growth (Falkenmark et al., 1999).

Ghali previously expressed this concern with the statement, "...that future wars in the Near East will not be connected with oil but water".

In the Hamun Basin in eastern Iran, damming activities in the lower Hilmand River section, the only tributary of the large shallow lake and its vast reed areas, have resulted in a critical drought, which may eventually cause the extinction of traditional "Gaw-dar" (cattle herdsmen) culture. With its Southeast Anatolia Project, Turkey has planned for 22 dams and 19 hydroelectric power plants in the upper Euphrates and Tigris Basin. With this ambitious plan, about 20,000 $km^2$ of dry country will be irrigated. This possibility has resulted in critical reactions by Syria and Iraq. Turkey is trying to mitigate this negative reaction with such promises as the construction of aqueducts from the Seyhan and Ceyhan Rivers, which enter the Mediterranean near Adana. In addition to the Jordan River concerns of Israel, Jordan and Syria, the Nile River Basin (3.1 million $km^2$) is the source of large-scale regional conflicts involving the annual discharge of the Nile (84 $km^3$) into the Mediterranean, with about 85% of the discharge arising from the Blue Nile, whose headwaters

are in Ethiopia. Other than fossil groundwater, Egypt, with its more than 60 million people and a fast-growing population, depends almost entirely on Nile River water. Egypt has been storing Nile River water in the up to 6000 $km^2$ Aswân Dam since 1970. This large reservoir not only provides 60% of Egypt's electricity, but has also increased the country's irrigation capacity. Nevertheless, Egypt is facing critical water problems in the near future. Moreover, the sustainable water supply of the Nile River depends on Ethiopia and Sudan, countries also in need of water for an expanding irrigation system. In 1983, several Nile River Basin states attempted to form an economic grouping that could have smoothed the path for a basin-wide Nile Water Authority. Kenya and Ethiopia refused to join, however, stating their governments would only negotiate such an agreement on a bilateral basis. Despite their reluctance, these two nations may eventually participate in a basin-wide authority. At the same time, in order to guarantee itself a certain portion of Nile water, Egypt also will be required to acknowledge the water rights of all the upstream riparian nations (Anonymous, 1994).

The 4200 km Mekong is the tenth largest river in the world, carrying about 5000 $km^3$ of water each year into the sea. The river flows from the Tibetan Himalayas southward through China, passes north of Burma, its watershed encompassing nearly all of Laos, northeast Thailand, most of Cambodia and the South Vietnamese delta. Thailand is interested in allocations of Mekong River water for irrigation in its northeast. Vietnam is concerned, however, that such diversions will further reduce the river's low flows between March and May. Laos is interested in hydropower development. The Mekong River Commission might be best served by emphasizing smaller-scale development alternatives, putting aside plans for large, main stem dams that are part of the development designs of the 1950s and 1960s.

### *1.2.3 Economic Value of Water*

Water represents a renewable resource whose economic value began to be recognized only after water shortages materialized. Economists have only recently recognized that one major reason for inefficient use of water resources is that many considered them to be "free" resources with no monetary value. This has subsequently led to the establishment of an "ecological economy" which attempts to overcome this deficiency, and which establishes the economic value of natural resources. Costanza et al. (1997) recently estimated the value of the "ecosystem services" provided to humanity by and rivers, considering such functions as water regulation, water supply, natural waste treatment, food production, recreation, etc. The estimated values of these services, provided free-of-charge by nature to humans, are given in Table 1.1. It is noted that in calculating the total world economic value for lakes and rivers, Costanza and coworkers assumed a total area of only $2 \times 10^8$ hectares, a value that may be highly underestimated, based on new data provided by Meybeck (1995) for lakes. This latter estimate of total lake area corresponds to $2400 \times 10^8$ hectares, about three orders of magnitude higher than that used by Costanza and co-workers. And this latter estimates includes only lakes, not reservoirs or rivers. If a raw estimate of the total area of lakes + reservoirs + rivers is taken to be only 1.2 times higher than the estimated

Table 1.1. The economic value of selected ecosystem services provided by lakes and rivers per hectare (according to Costanza et al., 1997) and the total global value (according to the estimate by Meybeck, 1995)

| Value of ecosystem service (1994 US $ $ha^{-1}yr^{-1}$) | | | | | Total value per hectare (US $ $ha^{-1}yr^{-1}$) | Total global value (US $ $10^{15}yr^{-1}$) |
|---|---|---|---|---|---|---|
| Water regulation | Water supply | Waste treatment | Food production | Recreation | | |
| 5445 | 2117 | 665 | 41 | 230 | 8498 | 4.9 |

lake area, the estimated value of the ecosystem services provided to humans by these water resources is 1440 times higher than the estimate given by Costanza et al. (1997). The recalculated value of US $\$2.4 \times 10^{15}$ per year for the whole world using these latter figures is in sharp contrast to the much lower value of US $\$1.7 \times 10^{12}$ per year estimated by Constanza and co-workers.

It is becoming evident in developed countries that increased water prices, covering not only the costs of treating and distributing water, but also the cost of the resource itself, has led to a considerable lower of water consumption in many cities. Both the quantity and quality of lake and reservoir water is conserved in this way. This is because increased water usage also leads to more water to be treated. Any treatment is only partially effective, however, and complete treatment of all effluents is not feasible in virtually any country. In many regions of the world, only a small part of water is treated, if at all. And both partially treated and untreated water reaches the waterbodies, including lakes and reservoir, with their polluting consequences.

The economic value of water is both direct and indirect. Direct values relate to the sale of water for human consumption and production. In such situations, water is a commodity. It is a phenomenon of our time that bottled water, which is commonplace in the developing countries because of inadequate water treatment and delivery systems, has become a major industry in developed countries. In part, this reflects trends within society, but also the reality that much of the freshwater of good quality previously available in large quantities is now less abundant. In the United States, for example, outbreaks of waterborne diseases in recent years has increased alarmingly, suggesting that potable water treatment techniques have failed to keep pace with the rate of decline in the quality of surface and groundwater. As a result, consumers have indicated a willingness to pay a higher cost per unit volume for water, with the cost differential reflecting their belief that bottled water is safer than public supplies. In contrast, the water consumed for agricultural services is often heavily subsidized by society, in order to minimize the cost of foodstuffs produced in irrigation schemes. These subsidies are often in the form of reduced water prices that allow agricultural producers to purchase water at rates below market value. The allocation of water between consumer groups, specifically the agricultural producers and urban dwellers of the American west, also has resulted in tensions that, at times, have taken on violent overtones (MacDonnell et al., 1995; Krannich et al., 1995; Coggins and Glicksman, 1998; Smith and Rast, 1998).

In addition to the direct costs, consumers also purchase water indirectly. Examples of such indirect costs include the nonconsumptive uses of water for hydroelectric power generation, where the cost of using water is hidden in the cost of the electric power produced, and for recreational or aesthetic uses. In the former case, the price paid for the use of the water does not involve the transfer of water to the consumer, but rather the transmission of electric power. Contained within the nonconsumptive use of water for power generation is the hidden cost of the environmental impacts of aseasonal water flows and habitat modification associated with dam operations. In the latter case, the consumer pays the price for the largely intangible benefits associated with the ambiance of lakes and rivers. In some cases, the incremental costs can be significant. In Maine (USA), a recent study indicated that a premium of approximately 10–15% of the total value of lakefront property was associated with perceived water clarity benefits (Michael et al., 1996). For clear water lakes, this premium was positive, increasing the purchase price that consumers were willing to pay. In contrast, for turbid or humic lakes, this premium was negative, decreasing the purchase price. Public perceptions of water quality, therefore, can have a major role in public economic activities.

An increasing number of industrialized countries are now using the "polluter-pays-principle", although the principle is unfortunately not used consistently or completely. An example of a complete application of the polluter-pays-principle is seen with the discharge of wastewater in Denmark. Based on analysis of the wastewater, a factor ($F$) is calculated. Although the equation used is dependent on the local conditions, the following equation is representative for the considerations:

$$F = BOD_5/(400 + BOD_5) + P/(10 + P) + N/(30 + N),$$

where: $BOD_5$ = biological oxygen demand (5 days) at 20°C;
$P$ = phosphorus concentration (mg $l^{-1}$) in wastewater; and
$N$ = nitrogen concentration in wastewater (mg $l^{-1}$).

The factor $F$ identifies the extra charge that industry must pay to the community to discharge wastewater. Thus, if the normal discharge price per cubic meter for municipal wastewater is $PD$, industry must pay $(F + 1)PD$. As can be seen, the equation considers the nutrient and BOD content of the discharged wastewater. This extra charge has encouraged many industries to at least partly treat their wastewater, because it was a less costly solution to treat it where it is most concentrated than to discharge highly polluted wastewater to the municipal treatment plants, where it will be considerably diluted. The introduction of the extra charge implied that the wastewater is treated at the site, where it is most cost efficient, and by the cheapest method. The discharge of toxic wastewater, on the other hand, should be completely prohibited, as it is harmful to the biological treatment processes at the wastewater treatment plants and to the waterbodies receiving the effluents.

The requirement for deposits for reusable bottles and tins is another economic means of reducing pollution. Recycling is in many ways a very attractive method to abate pollution, because it imitates the way nature handles the combined discharge and resource problem.

The use of so-called "green taxes" attempts to direct resource consumption toward more environmentally-friendly processes and products. For example, the most polluting energy sources, coal and other fossil fuels, are subjected to taxes in many industrialized countries. The heavy tax on gasoline in European countries is another example. If a polluter must pay more than the actual treatment cost at a municipal waste water treatment plant, for example, this may be considered a green tax.

It is expected that the use of green taxes will increase rapidly in coming years, particularly in industrialized countries. They will likely be applied increasingly as an instrument in resource management, as they are able to reduce the utilization of depleted resources. The renewable resources can be utilized in a sustainable manner by adjusting the rate of application to the rate of recovery. The application of green taxes can be a very powerful tool to obtain this balance.

A possible trend in the industrialized world toward replacement of income taxes by green taxes may be facilitated by the recognition that green taxes may be better able to adjust consumption patterns in society toward more sustainable resource management. Developing countries also can learn from this experience, and consider introducing green taxes, rather than income taxes, to cover the increasing income needs of the state.

Other recent developments of significant economic impact are the application of "green auditing" (Curran, 1996), "life cycle analysis" and "cleaner technology" (Section 4.2.2). It has been possible to demonstrate in numerous cases that production costs can be reduced simultaneously with environmental considerations, using these technologies. Application of such environmental technological changes in industry often results from the use of environmental certification in accordance with the ISO-standard 14000. Of particular interest for lake management is the possibility of upgrading existing wastewater treatment facilities at relatively moderate costs (Henze and Ødegaard, 1995).

### *1.2.4 The Role of Society in Lake and Reservoir Management*

The public response to the perceived degradation of rivers, lakes and reservoirs, like their perceptions of water quality, vary more as a matter of degree than as a matter of fact. Most countries of the world, for example, have some form of water resources legislation that includes elements regarding the prevention of pollution (Schlickman et al., 1995; United Nations Environment Programme, 1997, 1999). To a degree, such statements reflect the relative availability of water resources, and the range of uses to which the water resources are put, with countries that experience more severe water shortages generally having stricter water quality standards governing the discharge of materials to rivers and lakes (e.g., McKendrick, 1982).

In addition to this regulatory approach of governments on behalf of all its citizens, communities and individuals have adopted other strategies for managing lakes and reservoirs. Community actions are generally undertaken by governmental or other corporate organizations. In Wisconsin (USA), for example, citizen organizations include public "inland lake protection and rehabilitation districts", which are special-purpose governmental units formed by petition of the landowners. They have specific, but limited, authority to

raise funds through taxation and other charges for lake water quality management (Dresen and Korth, 1994). Such districts undertake water quality monitoring activities, lake management planning, aquatic plant management, and informational programming. Citizen organizations in Wisconsin include lake associations, or nongovernmental organizations with a waterbody focus (Dresen and Korth, 1994). These organizations have a voluntary membership, and may or may not be incorporated as a body corporate under state law. Funding of these organizations is also voluntary, except in situations where membership in the association is a requirement of residence within certain planned communities (membership in this situation is a requirement outlined in specific clauses in the property's title deed). Incorporated organizations have access to the same state-funded grant programs as lake districts, although they lack the taxation ability. These NGO's undertake many of the same activities as lake management districts.

Similar organizational bases exist in other locations. In many cases, the same concept of public–private cooperation has proven to be a successful mechanism for engaging and directing public involvement in lake management projects. This model is reflected in actions taken by the City of Cape Town in the management of Zandvlei (Thornton et al., 1995), and in the formulation of the recently adopted water law of Brazil (Anonymous, 1997). In the former case, the City of Cape Town, for example, worked cooperatively with the Zandvlei Trust, a citizen-based nongovernmental organization (NGO), to define the perceived problems to be addressed and to refine proposed management actions. Watershed-based management measures, especially those involving substantial informational programming, also were implemented in cooperation with other NGO's within the metropolitan Cape Town area. Their efforts included such activities as anti-littering initiatives sponsored by The Fairest Cape Association. In the latter case, the Government of Brazil recognized the need for broadly based representation and public "buy-in" as the basis for successful river basin management. A key feature of this law is the creation of river basin committees, comprised of federal, state, municipal and NGO representatives, for the purposes of promoting the discussion of water resource issues, arbitrating water resource conflicts, approving water resource plans, monitoring plan implementation, proposing exemptions from water-use rights awards, establishing water charges, and promoting the equitable funding of multiple-use water resource projects within specific river basins. Thanh and Biswas (1990) and Rast and Holland (2003) summarize similar public, private and public–private water resources management approaches used elsewhere in the world.

In addition to corporate or community level actions, individuals can participate in the water resource management process. In the United States and elsewhere, citizen participation ranges from volunteer lake monitoring (Ely, 1997) to watershed-based clean-up activities (Harding, 1997). Many multiple purpose options also exist, an example being the Wisconsin Adopt-A-Lake program (McCann, 1997). In this program, schools are encouraged to take an active role in their communities by learning about, monitoring, and helping clean a waterbody within their community. These programs are designed to be fully integrated into the classroom experience and can provide a vehicle for articulating and re-enforcing many previously identified societal values embodied in rivers, lakes and reser-

voirs. Individual participation within such programs also can lead to a more informed and knowledgeable citizenry which supports community-based decision-making and citizen-participation within water resources management projects. Such public involvement is a key to successful environmental solutions (Davis, 1997).

## 1.3 GLOBAL REVIEW OF LAKE AND RESERVOIR DISTRIBUTION

The distribution of lakes and reservoirs around the world is irregular. Regions with water surpluses often have many natural lakes, while those with water scarcity typically have fewer natural lakes. Reservoirs, on the other hand, are build extensively in regions of water scarcity. Nevertheless, countries with many natural lakes also may build reservoirs, mainly for power generation, water regulation and navigation purposes.

### *1.3.1 Volume and Area of Lakes and Reservoirs*

Estimates of the total volume of water contained in lakes and reservoirs depends on the definition of lakes and reservoirs, and the size taken as the lowest limit considered in the estimates. Thus, the estimates vary on the basis of these differences and how they are interpreted. The total water volume of lakes and reservoirs existing prior to approximately 1975 was estimated by Margalef (1983) to be approximately 229,000 $km^3$. Another estimate made at approximately the same time was 280,000 $km^3$, which has been confirmed most recently (The Lakes Handbook, 2003). The time period of the estimate is important, since new reservoirs are continuously being constructed, while the volumes of some lakes is diminishing significantly because of mismanagement and unsustainable use (e.g., Aral Sea, Lake Chad). With the further shrinking of Lake Aral, the number of lakes exceeding 1000 $km^3$ drops from 13 to 12 (still including 4 lakes with more than 10,000 $km^3$).

The most recent estimate of the area and volume of natural lakes by Meybeck (1995) is based on a revision of earlier estimates for the 253 lakes whose surface areas exceed 500 $km^2$. The total water volume of natural lakes with areas exceeding 0.1 $km^2$ is estimated as 178,000 $km^3$, with the values for lakes with areas less than 1 $km^2$ and the values for some regions with lakes with areas between 1–10 $km^2$ and between 10–100 $km^2$ being the least-confident components of this estimate. About half (82,500 $km^3$) of the total lake area is represented by saline lakes, with the Caspian Sea comprising 95% of the latter. The distribution of lakes based on their surface areas and the number of lakes comprising the total lake areas and volumes in different size categories is illustrated in Figure 1.5.

The figure indicates there is only one lake of the size class $10^5$–$10^6$ $km^2$, but over one million lakes less than 0.1 $km^2$ in surface area. The smallest lakes represent a very small fraction, however, of the total lake area and total lake volume. The total area of lakes in the regions for which direct estimates are available is estimated to be about 1,610,000 $km^2$, with one-half being in Canada and one-quarter being in the territory of the former Soviet Union. The total area of natural lakes on the globe, however, is estimated to be about 3 million $km^2$, which represents approximately 2% of the Earth's land surface. There are large

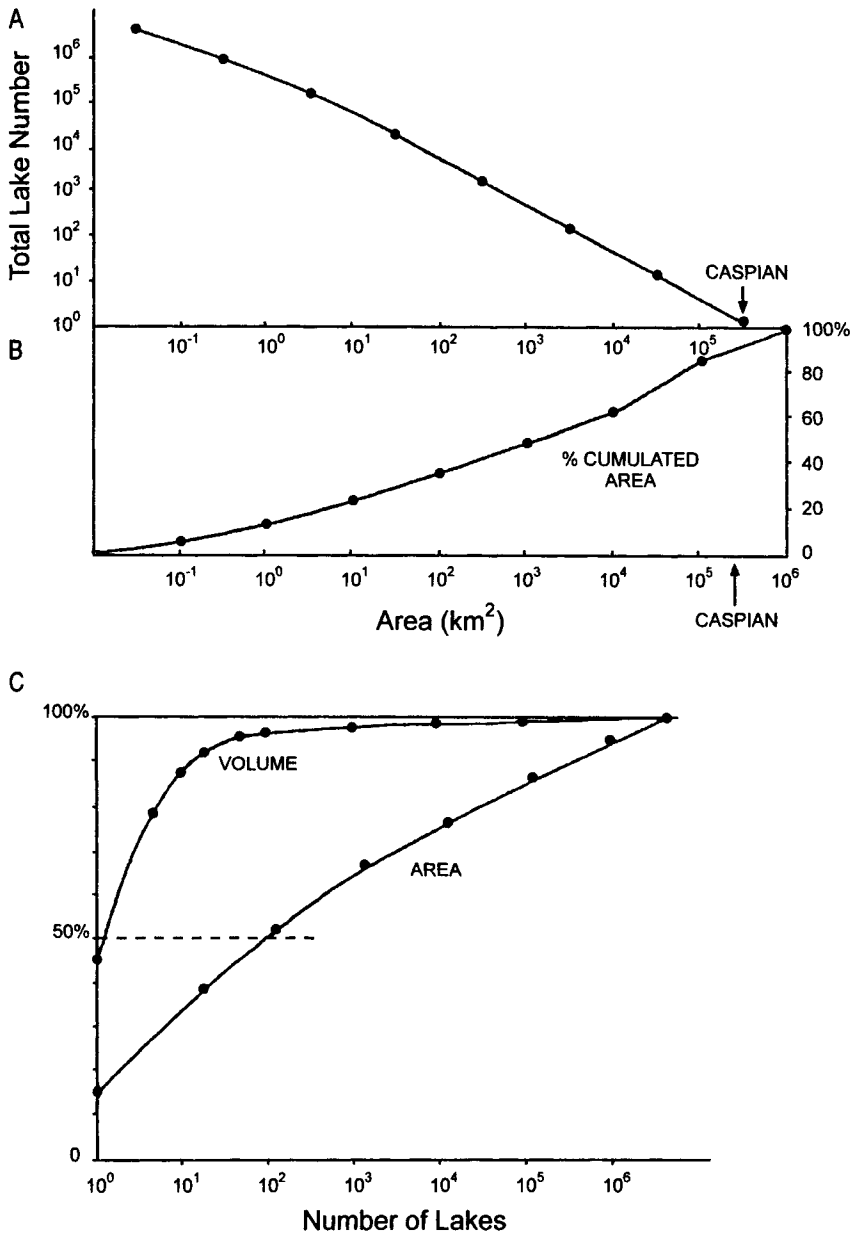



Fig. 1.5. Cumulative lake volumes and area with increasing lake numbers and cumulative area and total lake numbers, based on sizes of natural lakes (Meybeck, 1995).

differences between countries (e.g., 9.4% in Scandinavia, 0.88% in China and Argentina, 0.09% in France). The number of lakes exceeding 10,000 $km^2$ has been reduced to 17 from 18 due to the shrinkage of the Aral Sea. Geographically, most lakes are located in the deglaciated climatic zone (Table 1.2). The total number of lakes is estimated to be about 6.1 million. Lakes with surface areas between 0.1–1 $km^2$ represent about 5 million of the total.

From the perspective of water quantity management, the larger lakes would be the most logical choices for addressing large-scale human water needs. The hydrological reality, however, is that this use is constrained by their limited numbers and irregular distribution.

Reservoirs (artificial lakes) have existed for more than 6000 years (e.g., Kosheish Dam during the first Pharaoh Menes, 4900 BC) and have rapidly increased in number and size in the 19th Century, mainly in response to increased water demands for irrigation and energy. The most accurate reservoir data exists only for the larger ones. The World Register of Dams (International Commission on Large Dams, 1998) classifies large reservoirs as those (i) with dam heights of lists reservoirs with dam height of at least 15 meters and any volume

Table 1.2. Lakes of different size classes in major geographic regions (modified from Meybeck, 1995)

| Climatic zone | Lake class based on surface area | | | | Total lake area ($10^3$ $km^2$) |
|---|---|---|---|---|---|
| | 0.01–0.1 | 1–10 | 100–$10^3$ | $10^4$–$10^5$ | |
| Deglaciated regions | | | | | |
| *A* | 229 | 340 | 333 | 345 | 1247 |
| *n* | 3805 | 69.8 | 0.76 | 0.009 | |
| *dt* | 210 | 388 | 0.04 | 0.005 | |
| Temperate regions | | | | | |
| *A* | 20 | 34 | 103 | 31.5 | 188 |
| *n* | 310 | 5.8 | 0.17 | 1 | |
| *dt* | 12 | 0.23 | 0.008 | 0.004 | |
| Dry and arid regions | | | | | |
| *A* | 15 | 27.5 | 85 | 475 | 602 |
| *n* | 283 | 4650 | 0.147 | 6 | |
| *dt* | 67 | 0.115 | 0.09 | 0.0012 | |
| Wet tropical regions | | | | | |
| *A* | 39 | 52 | 50 | 81.47 | 223 |
| *n* | 600 | 11 | 0.11 | 0.002 | |
| *dt* | 25 | 0.44 | 0.084 | 0.00008 | |
| Desert regions | | | | | |
| *A* | | 7 | 55.6 | 64 | 127 |
| *n* | | 0.96 | 0.049 | 0.001 | |
| *dt* | | 0.04 | 0.02 | 0.0004 | |

Explanation: The temperate region is divided into two subregions, based on the extent of glaciation. Each value represents the center of the class in $km^2$.

*A* = area ($10^3$ $km^2$); *n* = number of lakes (thousands); *dt* = ratio of total area of region covered by lakes.

or (ii) a minimum dam height 10 meters and a volume of at least one million $m^3$. Some regions have many reservoirs, an example being Spain with more than 1000 reservoirs with volumes exceeding 40 $km^3$ on a territory of about 500,000 $km^2$, primarily because of its arid climate. Countries with many natural lakes, however, also may build reservoirs. An example is Finland, with about 10% of its land surface covered with natural lakes, but which also has built many reservoirs, some of considerable size (e.g., Porttipahta, containing about 1.35 million $m^3$ of water). Another example is Canada, with a high proportion of natural lakes, relative to its land surface area. Nevertheless, its Southern Indian Reservoir Complex, built mainly for hydropower generation, covers 2391 $km^2$ and contains a water volume of more than 23 $km^3$. As shown in Figure 1.6, the historic trends of reservoir construction are relatively similar in different countries. As seen from the comparison of the number and volumes of reservoirs in a few countries, smaller reservoirs were built in earlier periods, while the construction of larger ones has mainly been since the 1970s. Although reservoir construction continues in various locations around the world, during the last periods covered by these surveys, both the number and volumes of reservoirs have leveled off, and the rate of reservoir construction is decreasing dramatically, when one compares reservoir construction patterns between 1940–1950 and 1980–1990.

The geographic differentiation of reservoirs also is informative. Data for single regions indicate that, while there are few natural lakes in regions of water scarcity, many reservoirs have been built in such regions to compensate for water shortages. Far fewer reservoirs are generally found in regions with many natural lakes, and vice versa, as seen with the situation in the United States (Fig. 1.7). Intensive reservoir construction began only after 1900, with the two world wars interrupting this trend. After 1945, another boom or reservoir construction took place. After 1985, however, reservoir construction showed a marked decline, and very few new large reservoirs are now being planned. Virtually all the large reservoirs being constructed now are in developing countries (Rast, 2003; Rast and Thornton, 1999).

### *1.3.2 Geographical Conditions and Their Effects on Lakes and Reservoirs*

As shown in the previous section, the geographical distribution of lakes and reservoirs is uneven. This is due to major geographic differences in physical characteristics, first of all being their hydrological budgets. The consequences of hydrological budgets and other geographically conditioned variables are significant, not just for the existence of lakes and reservoirs, but also for other water quality processes. This is reflected in lake thermal structure and water mixing via the distribution of radiation, temperatures and winds, in aquatic chemistry via changes in soil and vegetation, and in the aquatic biology via temperature and other influences. The distribution of human populations also is an indirect effect, although is becoming a very strong influence in many places.

Another consequence of the geographical differentiation is the varying species composition of local flora and fauna from region to region, with more detailed differentiation than that based on the ecoregions defined below. This has consequence for biological monitoring, in that any system based on the presence of individual species is only local, and should not be used in other regions.

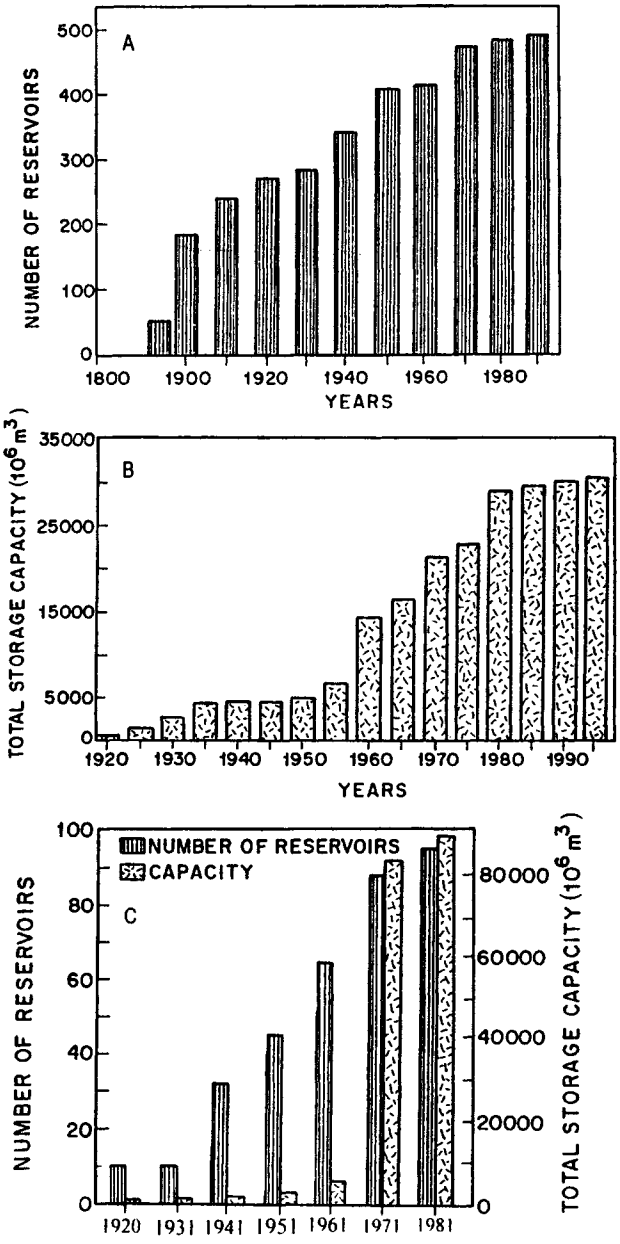



Fig. 1.6. Historical development of number and volumes of reservoirs in different countries. A—Number of reservoirs in United Kingdom (Stanners and Bourdeau, 1995); B—Reservoir storage capacity in Murray–Darling Basin of Australia (Blackmoore, 1995); C—Numbers and/or storage capacity of reservoirs.

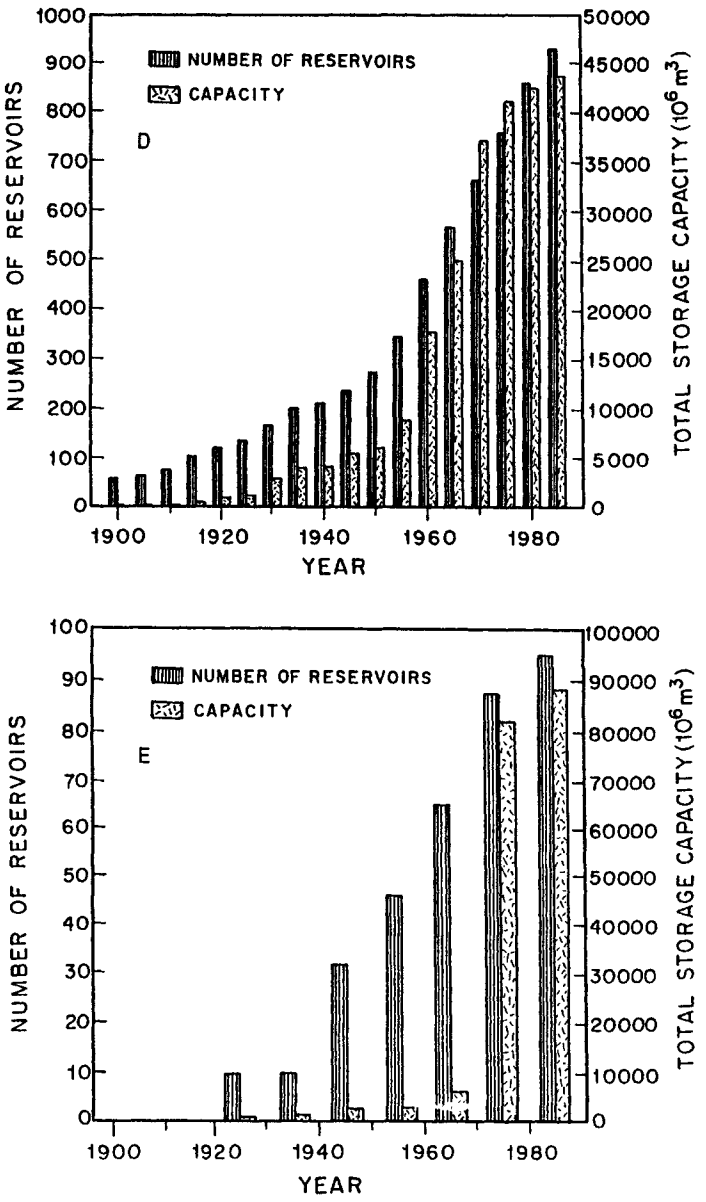



Fig. 1.6 (*continued*). D—Number and volume of reservoirs in Spain (Riera et al., 1992); E—Number and capacity of reservoirs in Germany.

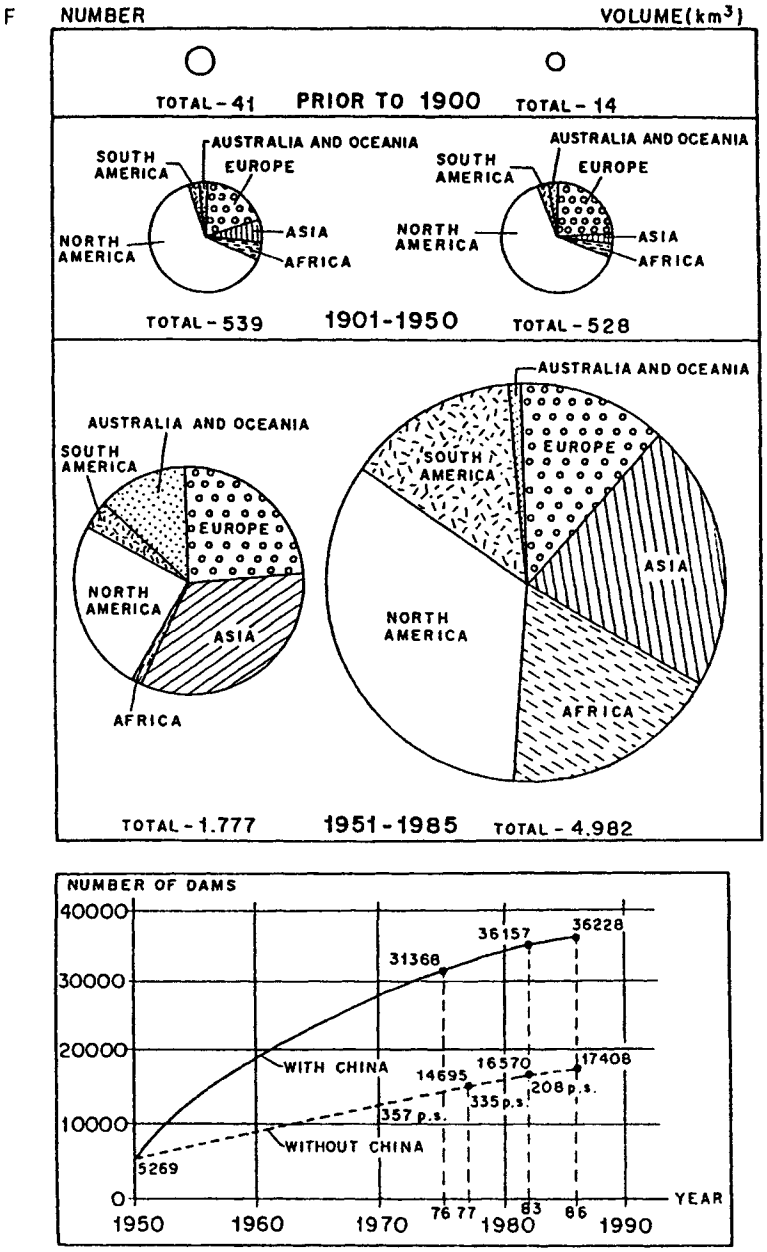



Fig. 1.6 (*continued*). F—Comparison of reservoir developments throughout the world (Mermel, 1991).

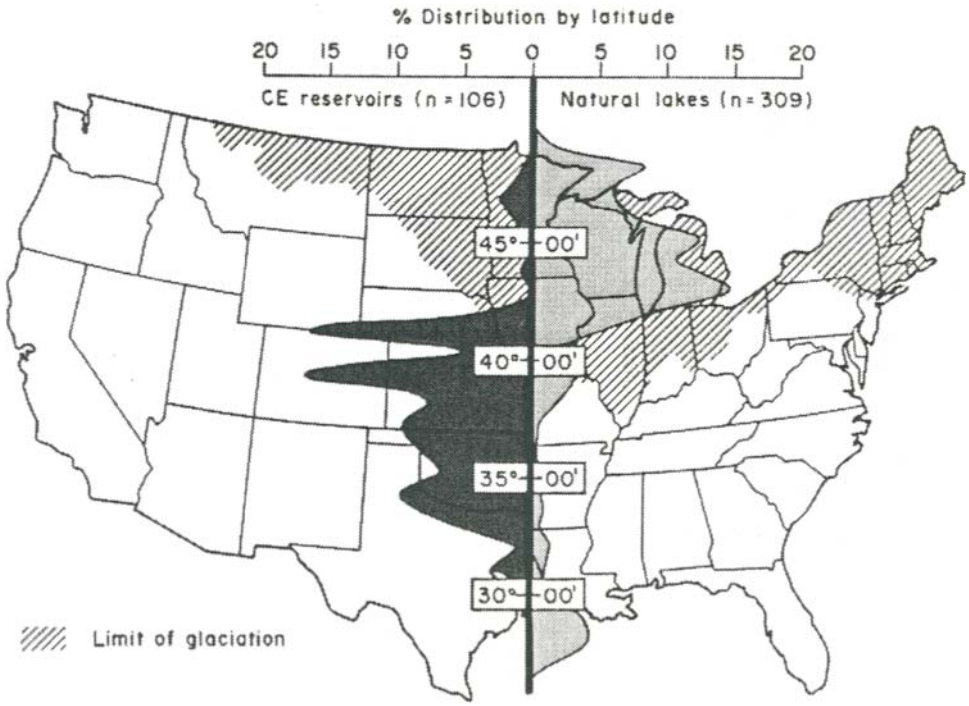



Fig. 1.7. Distribution of natural lakes and reservoirs in the United States (Walker, 1980).

A short description of the basic ecological zonation around the world is given in the following section, along with a discussion on how it is reflected in lake and reservoir water quality variables. Effective water quality management efforts must consider and reflect these differences.

*Major ecoclimatic regions*

Figure 1.8 illustrates a gross differentiation of the world's major ecoregions. The physical conditions of the major global ecoregions relevant to a water quality perspective are based on patterns of precipitation, evaporation and the resulting flow, solar radiation, wind speed and direction, cloudiness and air temperatures. These are summarized in Table 1.3, along with the resulting major characteristics of lakes and reservoirs of given regions. It is evident from Figure 1.8 that, although the zonation is largely latitudinal, major modifications also exist because of the distribution of continents, mountain ridges, and the effects of the oceans. The zones are located approximately symmetrically in respect to the Equator. Differences between the Northern and Southern Hemisphere, however, are due largely to the much smaller land mass in the South (except for Antarctic). Thus, the values given in the table and following figures are only approximations.

Table 1.3. Four major geographical zones and resulting lake and reservoir characteristics

*Humid tropics* (0–15° latitude)
- Radiation—Highest annual average radiation with very low seasonal and stochastic component
- Photoperiodicity—Constant
- Mixing—Deepest from all regions

*Dry regions* (15–35° latitude)
- Hydrological budget—Negative, leading to extreme stochastic variability of precipitation and flow rates
- Mineral turbidity—Extremely variable, related to above
- Chemical composition—Extremely variable

*Temperate regions* (35–60° latitude)
- Radiation—Largest seasonal component of incident solar radiation variability and radiation available to lake phytoplankton
- Air and surface temperature—Largest seasonal and stochastic components of variability
- Minimum surface temperature—Reaching to freezing point

*Subarctic and polar regions* (> 60° latitude)
- Zone of continuously-frozen lakes

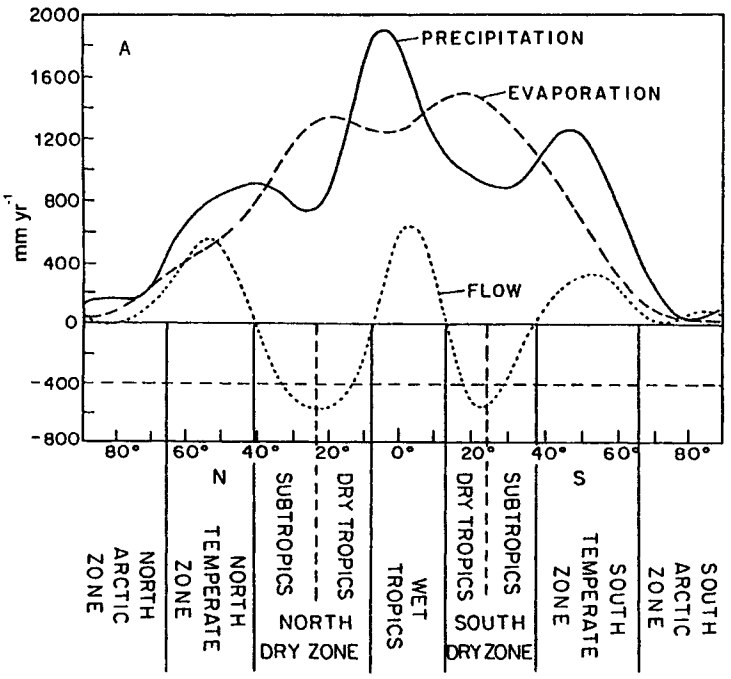



Fig. 1.8. The distribution of water budgets and soil characteristics. A—Annual average precipitation, evapotranspiration and runoff, based on latitudinal zones (Straškraba and Tundisi, 1999).

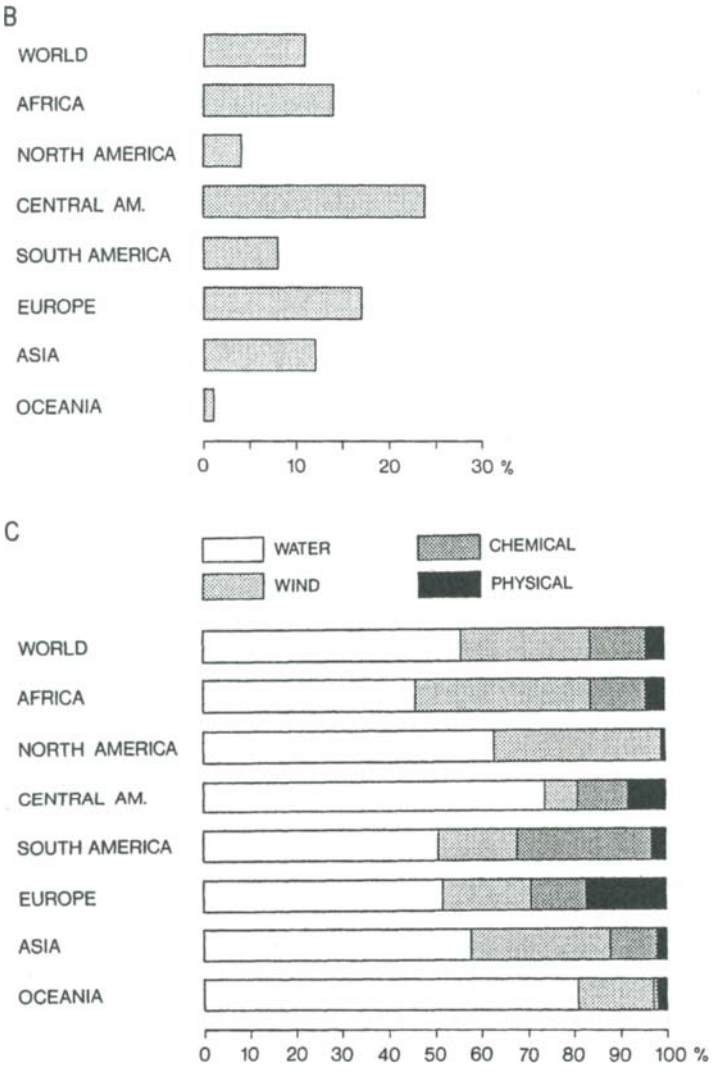



Fig. 1.8 (*continued*). B—Soil degradation as percent of vegetated land, 1945–1990 (Hebel, 1995); C—Types of soil degradation by region, 1965.

The distribution of the average annual water budget around the globe shows an approximate regularity on both sides of the Equator. Extremely dry regions are found mainly at latitudes around 20°C (the trade winds zone), with highly negative water budgets where evaporation greatly exceeds precipitation. Negative water budgets occur at latitudes between about 10 and 35 degrees. There is a smooth transition of water budgets along the latitudinal axis, and exact boundaries between zones do not exist, except where

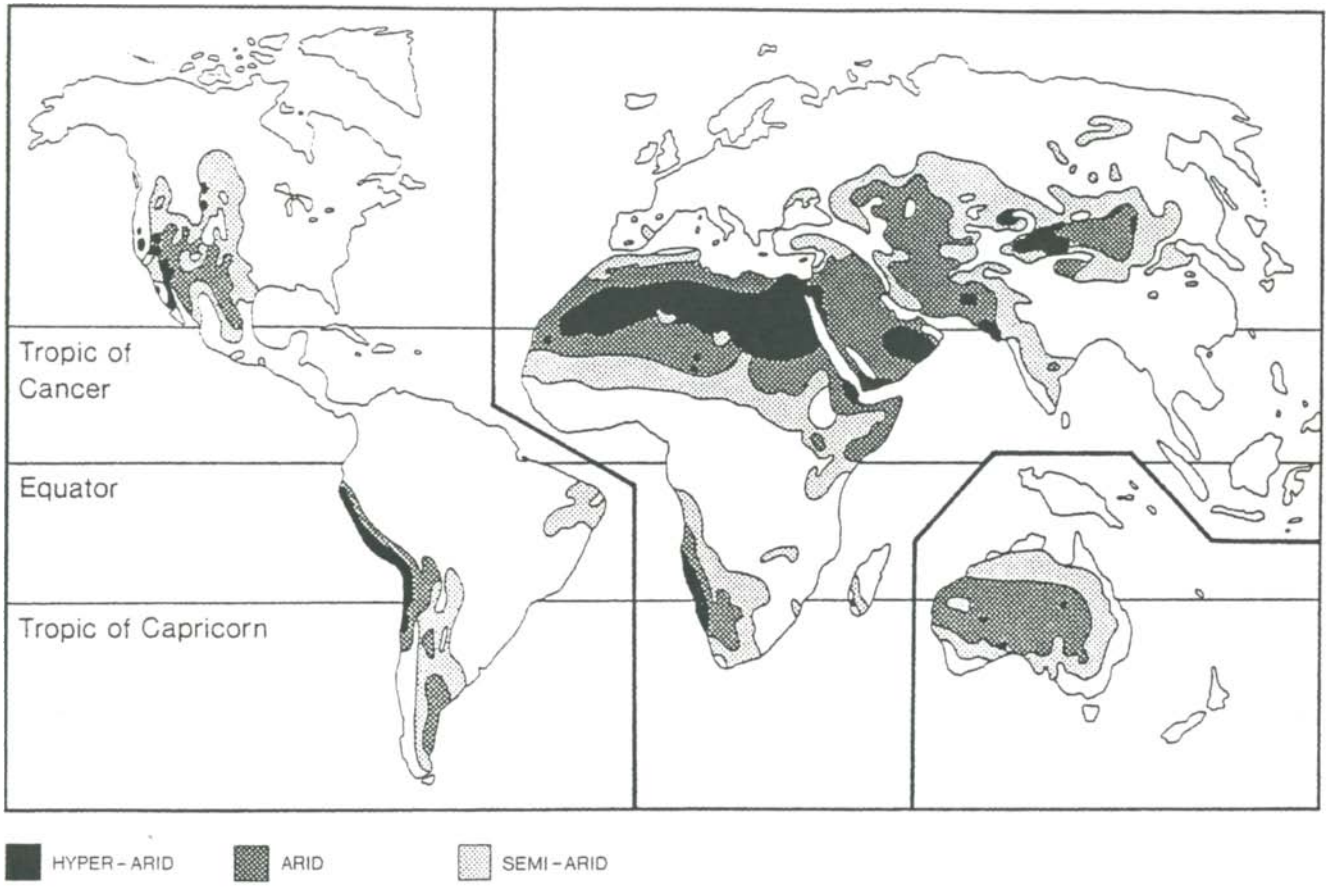



Fig. 1.9. Major arid regions of the world (Bailey, 1996).

supplemented by more clear-cut boundaries among vegetation types and mountains, such as in East Asia or along the Americas.

The same approximate regularity is seen for other variables. Thus, in the following text, latitude is considered independently of its position on the Northern or Southern Hemisphere. The half-year time shift in the position of seasonal maximum and minimum as shown schematically for temperatures in Figure 1.10, however, should be kept in mind. Some differences between the Southern and Northern Hemisphere are caused by different proportions of land and sea; in the North there is more land area, while the sea is more extensive in the South.

In addition to the major latitudinal zones for the existence of lakes, there are more subtle differences within individual countries that may be of great interest to water managers.

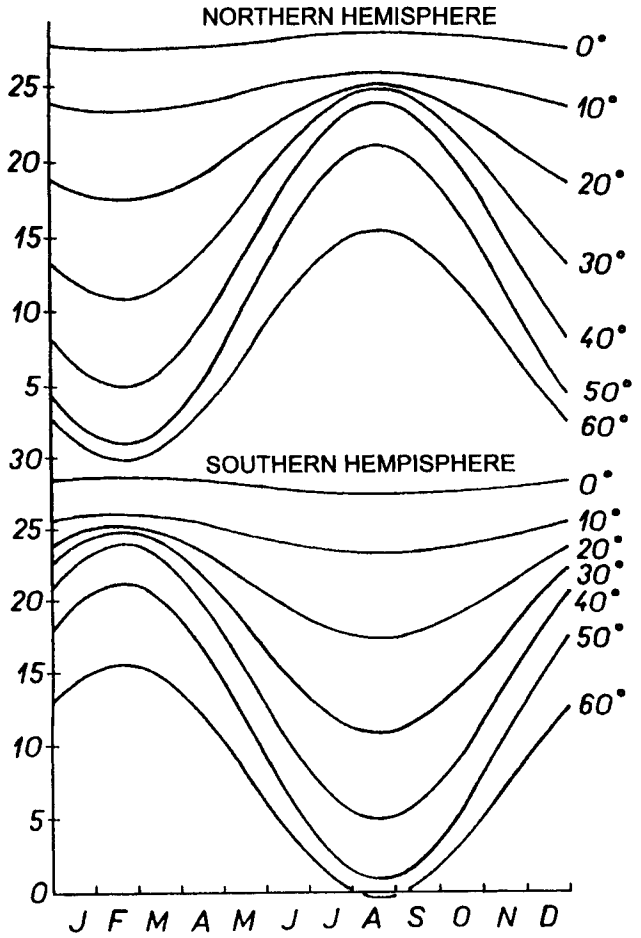



Fig. 1.10. Average global temperature distribution by continent and globally (Straškraba, 1980).

These are due primarily to the effects of continentality, elevation, orography and geology, with the associated soil and vegetation types.

Continentality expresses the distance of the land area from the sea. Its effect on the extremity of temperature fluctuations is well known, with nearshore areas being known for milder temperatures, and inland areas for harsher temperatures. This difference is reflected in air temperatures, with the annual temperature amplitude increasing about 1.6 times in the first 10 km from the shores, but progressively rising to 2.6 times at a distance of 100 km and to 4.6 times at 1000 km from the shores (Straškraba, 1993). Lake temperatures change correspondingly, with increased summer temperatures. There also is a decrease of lake salinity inland, although this effect is restricted to a several kilometers range along the shoreline.

Altitudinal differences are to some degree equivalent to latitudinal differences. The annual average water temperatures seem to decrease with altitude, at a rate of about 0.4–0.8°C per 100 meter elevation. Precipitation, solar radiation intensity and wind speeds generally increase with altitude. The effect of higher wind speeds on lakes are often compensated by more complex orography in mountainous regions. Some differences in population densities and land use also will be associated with altitude, with less people and less intensive land use generally occurring at higher altitudes.

In addition to factors that are closely related geographically, other factors also show geographic trends. The relation to geography, however, is indirect and less pronounced. The trends are loosely related geographically (e.g., water turbidity is much higher in drier regions and regions with intensive rains). Variables unrelated to geography, such as the nutrient loads resulting from human activities, or differences in biological composition due to fisheries and pollution, may override the geographical dependencies.

A recent development in a number of countries is to distinguish "ecoregions", characterized by general ecological conditions (Omernik, 1992, 1995). A regionalization based on total phosphorus concentrations in waterbodies in the United States, for example, was made by Rahm et al. (1995).

*Relation of geography to water quality*

In Figure 1.11, the macro-geographical differences in physical forcings relevant for the water quality of lakes and reservoirs are diagrammed for locations at given latitudes, irrespective of their North or South location. Three components are distinguished, including (i) annual average values; (ii) seasonal variability; and (iii) degree of stochastic, irregular component. The three components together best characterize conditions at any given latitude.

The aquatic chemistry of lakes and reservoirs also is dependent on geography, although the differentiation is less clear because of the interference of local geological formations. The ratio of precipitation to evaporation plays a role not only in diluting or concentrating chemical species, but also in determining the dominant minerals in the water.

The sediment load to lakes and reservoirs is typically determined by runoff conditions in their drainage basins, and is an important variable affecting the operational life of a

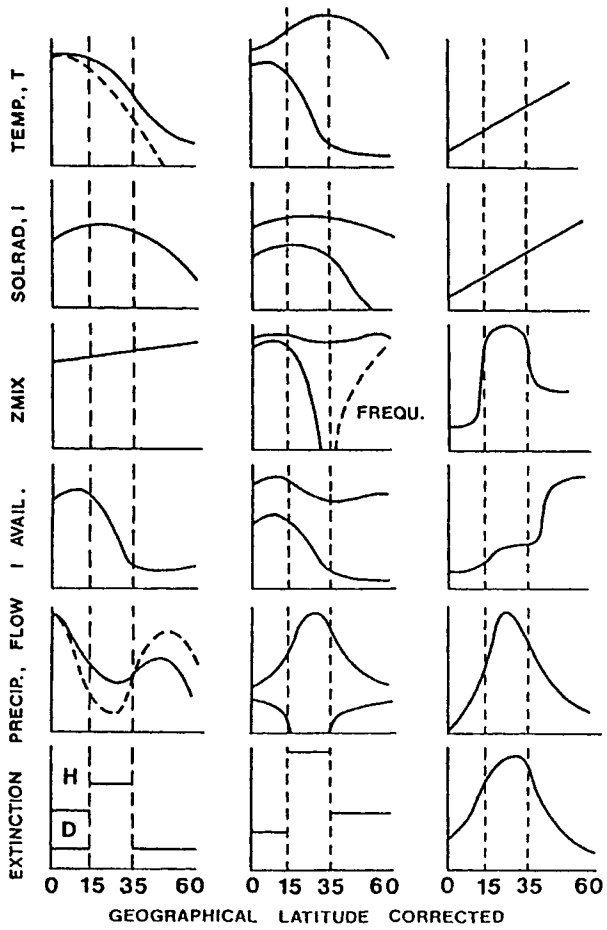



Fig. 1.11. Macro-geographical differences in physical forcings: Average, seasonal and stochastic components of latitudinal variations in solar radiation, air and water temperature, precipitation, water flow, turbidity and light availability for phytoplankton in lakes and reservoirs (Straškraba, 1993).

reservoir. The development turbidity is parallel to the ecoclimatic regions, with highest turbidity in dry regions.

A lake located high above sea level, therefore, will on average have conditions similar to a lower latitude lake situated at higher latitudes. Differences of only several hundreds of meters in elevation can play a significant role in influencing these conditions.

A characteristic feature for lakes in any geographical position is the annual water mixing cycle, a very important property influencing the development of water quality over the annual cycle. Mixing types have been found to be related to geographical latitude and altitude (Hutchinson and Löffler, 1956). The effect of altitude reflects altitudinal changes in

temperature and other previously-described variables. The overall effect is that the geographical boundaries for different mixing types are related to altitude as one moves toward the higher latitudes.

The only mixing type almost independent of geography, and which occurs in most regions of the world, is the polymixis, or transitory complete mixing of shallower waterbodies. The delineation of the depth of polymictic lakes is due to their relative, rather than absolute, depth. Thus, it cannot be unequivocally said, for example, that lakes with depths greater than a certain number of meters will mix, while lakes deeper than this depth will not. The mixing is basically due to two forces; namely, the force of the wind, and its counteraction by buoyancy related to the potential kinetic energy of the water mass in the lake or reservoir corresponding to its temperature structure. The wind starts to act on the water surface essentially on the basis of the 1 : 8 rule, which means that the wind blowing over a landscape reaches the water surface approximately in the distance from the shore that is 8 times the height of obstacles on the shore. The wind fetch, the force with which wind creates water movement and mixing, is related nonlinearly to the distance the wind travels over the lake surface. The wind gains speed over flat, smooth water surfaces. Thus, lake mixing depends on the size of a lake, rather than its absolute depth. Because wind-induced mixing penetrates just to a certain depth, it is the ratio between the depth of wind mixing for the particular size and wind exposure of the lake that is decisive for its hydrological shallowness and polymixis. The second defining variable for wind mixing is the thermal structure of the waterbody, mainly the air–water temperature difference and the development of internal density gradients. The same wind fetch will cause much deeper mixing when the gradients are low. This is why polymixis will be restricted to waters only a few meters deep in temperate regions, while polymixis also is common for very deep lakes in high altitude tropical regions. Polymictic waterbodies are intermittently mixed and stratified, depending on the existing weather conditions. On calm days, particularly with nights when the air temperature remains higher than the surface water temperature, and no wind stirs the water surface, stratification of the water column begins and can persist for a few days, until the wind starts to blow and the temperature grows colder. When this happens, the surface water cools down, sinks because of its greater gravity, and mixes the waterbody via convective mixing. Anoxic conditions in polymictic lakes are more detrimental to water quality than for lakes of other mixing types because it developed very rapidly (within a few days), unlike other mixing types. The onset of thermal stratification prevents mixing between the deeper and shallower water layers in lakes and reservoirs, and the ratio of the water–sediment contact zone to the water volume of the layers that are isolated from contact with air is much lower. Thus, the oxygen consumption in this small volume can result in rapid oxygen depletion.

Outside the tropical belt, where seasonal climate dominates, three mixing types with respect to full circulation of the waterbody may be distinguished. Cold monomictic lakes are connected with high latitude and altitude, and are frozen over most of the year. They are fully mixing only during summer if their surface temperatures do not exceed 10°C, which implies only small density differences. Inflow from glaciers or frequent cloudiness are most likely essential to provide for the low temperature. Otherwise, the twenty-four-hour period

at high latitude and the strong insolation at dry high altitude areas may easily heat the upper layer of lakes and thus suppress full circulation. Due to the ongoing climatic change, cold monomictic lakes are obviously extremely rare and most likely absent from the Southern Hemisphere. There are some indications that during certain periods of the (early) Holocene they might have been common (Löffler, 1998). Lakes with two circulations—after cooling during autumn followed by the break down of the thermal stratification and after the melting of the ice during spring or at least the inverse winter stratification—are designated dimictic and most common throughout the Northern Hemisphere. Most likely among deep lakes they even present the majority on our globe. However, they are almost missing in the Southern Hemisphere where they obviously are limited to the southern Andean region. This is mainly due to the oceanic predominance, which is also responsible for their absence in most of Japan and in Ireland. Finally, warm monomictic lakes undergo circulation during winter at temperatures well above 4°C. They are typical of the Mediterranean type of climate, as well as regions with strong oceanic influence. Ireland and most of Japan are examples where this climatic type, which dominates in the Southern Hemisphere, prevails.

Within the tropics, distinguished by preponderating diurnal climate, lakes at great altitudes (mostly well above 4000 m) lack any persistent thermal stratification. Cooling during the night (typical of altitude ranges with regular change of frost and thaw) results most often in complete mixing, followed by only feeble stratification during the day. This kind of lake has been labeled cold polymictic (Hutchinson and Löffler, 1956). The majority of this type is known from the tropical Andes (Löffler, 1960), high mountains in East Africa and New Guinea. In contrast to the proposition made in 1956 (Hutchinson and Löffler, 1956), tropical lakes at low altitude have a tendency to mix completely once a year (instead of several times), most often during the dry season. In East Africa, Lake Victoria is a remarkable example of this phenomenon. Therefore, they are denoted monomictic lakes which must be distinguished from temperate warm monomictic lakes. Warm polymictic lakes as described by Hutchinson and Löffler (1956) may be connected with regions where sudden cold rain events or even hailstorms are frequent events. With their high density gradient and only a few degrees difference between surface and maximum depth, such climatic impacts can easily release complete mixing of a lake. Another rather hypothetical group suggested by Hutchinson and Löffler (1956) comprises an oligomictic type characterized by very infrequent circulations at irregular—not even annual—intervals.

Finally, it should be mentioned that the globally distributed amictic lakes, which are sealed by ice from any climatic influence since the onset of climatic warming, have become increasingly rare. They were known from glaciated high mountain areas, such as the Alps, the Andes and Mt. Kenya (Curling Pond, at least until 1960). At present, only a few examples exist in the polaric regions. The most prominent one, Lake Wostok in Antarctica, may be frozen over since the late Pleistocene Era. It should be stressed that, along with climatic warming in our time amictic lakes within the tropical belt turn into cold polymictic ones, and outside the tropical climate turn into cold monomictic, if not immediately into dimictic types. The latter can then turn into warm monomictic ones. Within the tropical belt the cold polymictic lakes will be shifted increasingly to higher regions, whereas the other types mentioned will probably be more subjected to regional climatic changes.

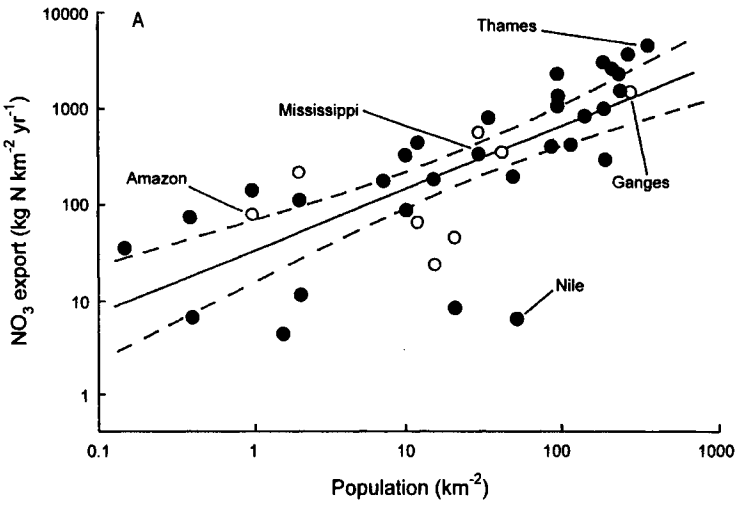
Among these climatic types of fully circulating (holomictic) lakes, the warm monomictic, monomictic and dimictic ones are most likely to undergo oxygen deficiencies even under natural conditions. This is much more so in lakes with a permanent density gradient, which results in partial circulation (meromictic lakes) and in anoxic conditions in the deeper water layers. The most prominent example of this phenomenon is Lake Tanganyika, with a maximum depth of 1471 m, but devoid of oxygen below about 110 m. In small lakes the anoxic deep water ("monimolimnium") below the upper circulating layer ("mixolimnium ") can be, if desirable and feasible, reduced or even removed by means of the Olszewski-tube (a siphon technique).

Most of the described climatic types of deep lakes—although less so for the reservoirs—are located in exorheic regions connected to the sea. Within endorheic basins, any connection with the sea, they are most often confined to the boundary region of such regions, with their main distribution in Australia, Asia and Africa. They are much less presented in the Americas and Europe. Endorheic regions are recognized by their great diversity of hydrological and chemical features, and need significant attention with respect to their water management. The catastrophe of the Aral Sea, and the rapidly increasing problems of the endorheic region in Iran are dramatic examples of mismanagement of lakes which can hardly be restored. In both cases, large scale agricultural projects were responsible for the observed disastrous conditions. Similarly, agricultural activities in dry regions (both, endorheic and exorheic) require the use of lakes for irrigation, causing hydrological and pollution problems. Examples of such impacts are numerous. In Europe, they are well known in the Mediterranean region (e.g., Lake Prespa, shared by Albania, Greece and Macedonia). Some regions are also characterized by large scale water transfer projects (e.g., South Africa, Mexico, Greece, etc.) which often are combined with the construction of reservoirs, but also contribute to the devastation of lakes and wetlands (e.g., Jonglai-Canel, Sudan-section of the Nile).

*Consequences of geographic differences for water quality management*

The geographical differences of a number of water quality variables are great. It would be erroneous to use experiences from one region for managing lakes and reservoirs in other regions, without taking such differences into account. The material below describes some of the relevant rules to be considered in this context.

*Regions with low and high population densities.* Although directly and indirectly related to geography, there are major regional differences in population density and its economic wealth. Summarizing investigations show that a strong correlation exists between pollution and population density for a number of water quality variables. Figure 1.12 illustrates the relations between some water quality variables and population densities in different regions. It would be erroneous, therefore, to use estimates made in a region with low population density to another region with high population densities. The differences are not just related to the population density itself, but also to the differing practices of peoples living in the different regions.



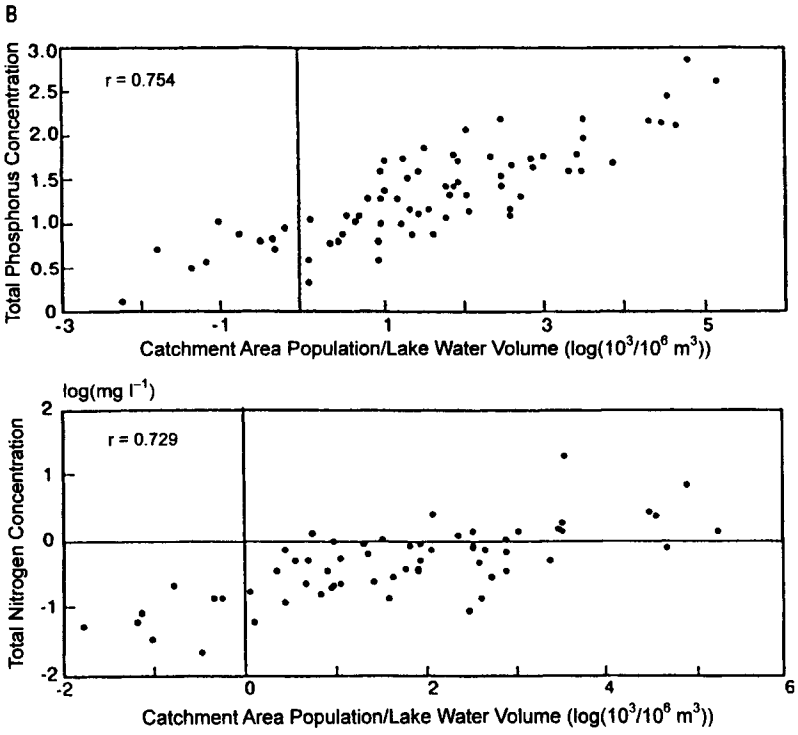




Fig. 1.12. Population density and water quality variables. A—Average annual export of nitrate as a function of human population density in the catchments (open circles—tropical rivers; filled circles—other rivers; Howarth et al., 1996); B—Relation between total phosphorus and total nitrogen concentrations in catchment areas and lake volumes (Kira, 1993).

*Temperate and tropical regions.* Experiences from temperate regions, from which most lake and reservoir knowledge has been gained, are often applied to tropical conditions. This error can happen both to experienced temperate zone limnologists and water quality managers with no experience in tropical regions, and to tropical specialists trained in a temperate country. The domination of the daily cycle over the annual cycle, typical of tropical conditions, is often ignored since nothing similar occurs in the temperate regions. The domination of the annual cycle in temperate lakes also is reflected in phytoplankton production as the primary source of eutrophication difficulties (Fig. 1.13). Inaccurate conclusions are easily made from water quality measurements taken in shallow tropical waters during morning hours when the water is mixed, while it may become stratified and anoxic during the day. Lake and reservoir water quality monitoring must consider these daily differences much more strictly than in the temperate regions, where differences in some variables such as pH, oxygen and nutrient concentrations are not very pronounced over a period of a few hours. Anoxic conditions can occur in tropical regions at lower external organic loads because of higher temperatures and higher production of allochthonous

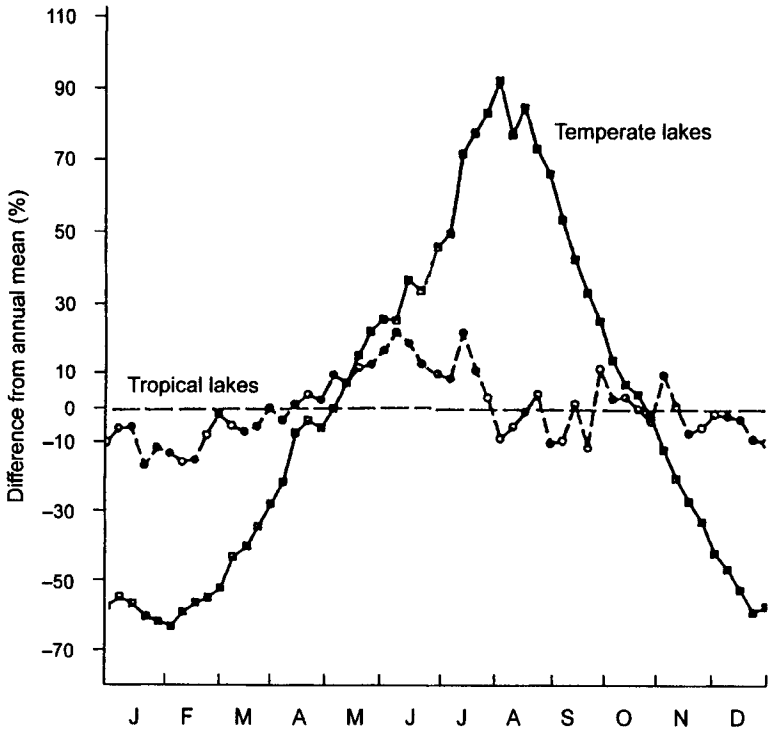



Fig. 1.13. Seasonal variation of primary production by phytoplankton in temperate and tropical lakes (mean percentage difference from annual mean values; modified from Alvarez Cobelas and Rojo, 1994).

organic matter by phytoplankton. One feature easily overlooked because of similar effects on lake and reservoir water quality, but with completely different consequences for their management, is the differing nature of the problem of general pollution and eutrophication (Rast and Thornton, 1996). While in-lake methods may be effective in treating eutrophication symptom in some cases, efforts to treat pollution typically must concentrate on the drainage basin.

*Dry and wet regions.* Major differences also exist between the wet, near-Equator part of the tropics and the dry belts around it. Similar, but less pronounced, differences exist between regions with more balanced hydrological budgets (i.e., evapotranspiration approximately equals precipitation) and more arid and semiarid regions. Much greater variability in water quality can be expected in a dry region than in regions with balanced water budgets (Thornton and Rast, 1993; also see Glantz, 2001). Multiyear cycles of drought can cause decreased water quality during periods of reduced flows and reduced lake volumes. On the other hand, more intensive short-term flushing by strong precipitation events brings more silt and pollution from the landscape to lake waters. Major differences also can occur because of population density, which is naturally reduced in a dry country compared to regions with greater precipitation. The situation in certain regions of India with very high population densities, which are also dry because of water mismanagement, does not appear to follow this observation. The consequences of flow irregularities, resulting in higher pollutant loads, can lead to underestimates of load coefficients valid for hydrologically-balanced regions, which usually also have well-developed vegetation.

*Forested and agricultural regions.* Major differences in water quality are to be expected in forested and deforested regions. The reason is twofold:

- Forest conditions are usually closer to natural conditions, in which the least disturbance of vegetation takes place, and the population density is low; and
- No fertilizers or pesticides are usually applied, and no agricultural practices that increase erosion are used.

*Economically developed and rapidly-developing countries.* The natural geographical situation in the present world is largely modified by human activities, and socioeconomic development plays a very important role for the landscape and waterbodies. Figure 1.12, for example, illustrates the role of population density. Population density is not only directly related to various components of water pollution, but also to other socioeconomic changes that can indirectly affect waterbodies. Other major differences in water use and water quality are related to different kinds of industrial activities, in regard to both the extent and magnitude of sophistication. From this perspective, the strongest differences exist between the developed part of the world and the rapidly-developing so-called "Third World". The politico-economic differentiation into "NORTH" and "SOUTH" (Fig. 1.14) characterizes these differences, which are strongly reflected in the waterbodies of the two regions, making the understanding of complex water problems more difficult.

*Global changes.* To further complicate the tasks of water managers, the present physical world is not static. It undergoes relatively slow, but continuous, changes, usually summarized under the term of "global changes". One component is global climate change,

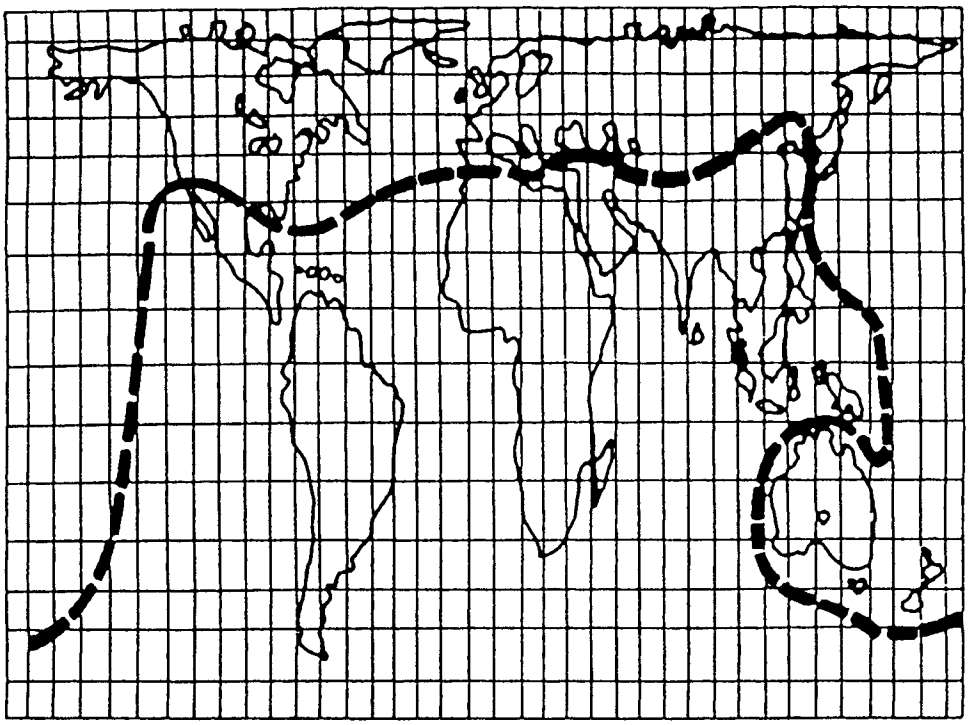


Fig. 1.14. Global political-economic distinction of North and South (Goldemberg et al., 1987).

with global warming being the most broadly-known element. It is misleading, however, to reduce the observed changes just to the issue of global warming. Temperature rises are observed on average throughout the world (Fig. 1.15A), but it is not regularly realized in all locations. In fact, although warming prevails on average for the whole globe, in some regions, there is to be expected a significant cooling rather than warming. Even more misleading than the local differentiation is the increase of short-time temperature irregularities. Some years can be rather cold, while others are much warmer. For waterbodies, therefore, it is not just the solar radiation and resulting temperature regime. The changed hydrological budget also is of great importance. In this latter case, we are speaking about global climate change. Observations of these phenomena in places remote from direct human interference (Fig. 1.15B) demonstrate how significantly waterbodies are already changing. And, in addition to temperature and climate changes, there are other global changes associated with the manufacture and spread of different products and processes that can change the environment. Some may represent environmental "time bombs", with direct effects of climate on human cultures, on vegetation in general and on crops in particular.

For water managers, therefore, it is necessary to take ongoing changes into account, considering the world to be in a state of dynamic development (Smith and Rast, 1998).

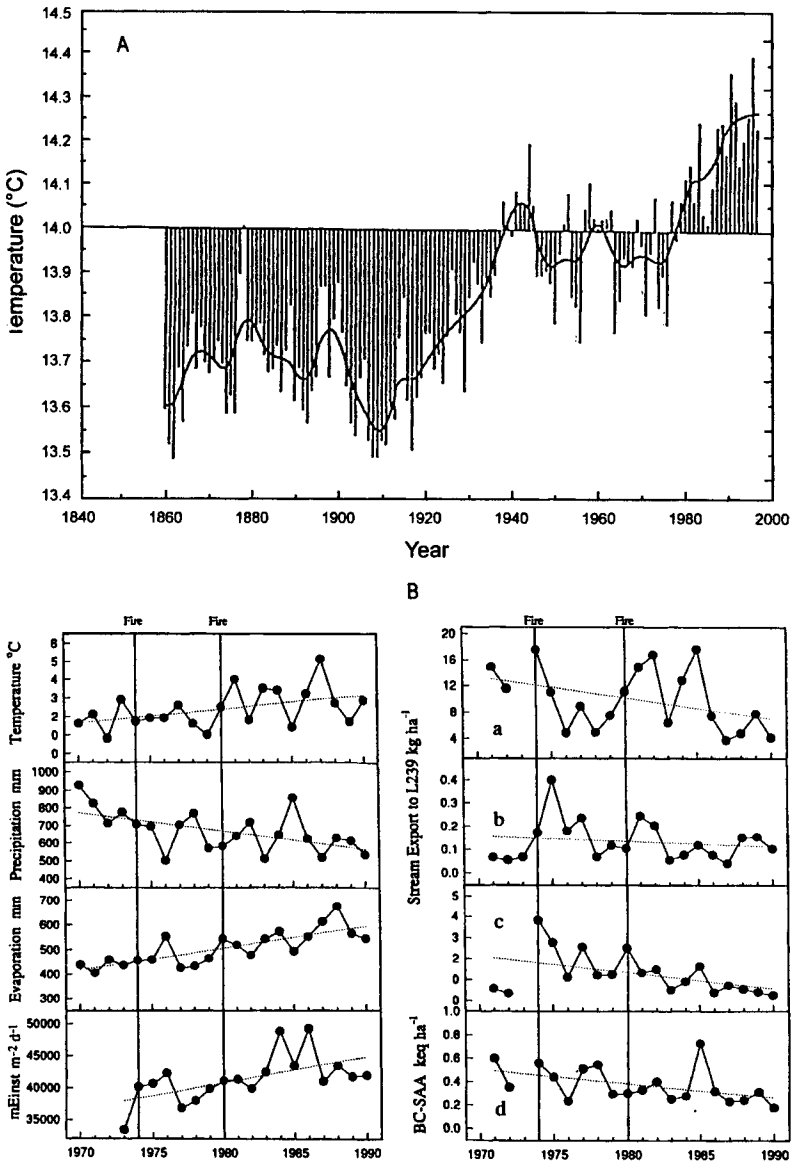



Fig. 1.15. Global changes and their consequences for lakes. A—Average global air temperatures near the ground (land and sea) (horizontal line—average determined from measurements on the whole globe. The horizontal line represents average from period 1961–1990; B—Climatic conditions (left) and loads to a lake in Northern Canada (right). (Other changes—increase of thermal capacity of lakes; high increase of retention time (from about 4 years in 1970 to about 20 years in 1990 in one lake); decrease of runoff and increase of the number of days without flow; Schindler, personal comm.)

Water resource development projects typically have rather long planning and construction phases. Thus, what is valid now may be very different from what actually happens after they have been realized.

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*Chapter 2*

# LAKE AND RESERVOIR WATER USES AND ABUSES

Attempting to manage anything is impossible if the usage of the object(s) to be managed, and the causes of possible management difficulties, are not clarified. Thus, it is useful to begin with the identification of lake and reservoir water uses, and the possible causes of deterioration of their water quality. Reservoirs are constructed for beneficial human water needs, sometimes for one particular purpose, but more recently for multiple simultaneous purposes (i.e., multipurpose use). In contrast, lakes are natural waterbodies, often without designated human water uses. However, the use of their water is recently becoming more intensive and multipurpose, particularly for lakes in heavily-populated countries and intensively-utilized regions. This multipurpose and extensive use can often lead to abuse and conflicts, to a reduced ability to supply water of good quality, aesthetic and safe for human consumption. This chapter distinguishes twelve types of lake and reservoir functions. The deterioration of lakes and reservoirs is more difficult to classify, and two major groups are distinguished. One is based on their improper usage—abuses of the waterbodies, classified according to sources or reasons of the deterioration. The other is based on the agents and compounds causing the deterioration—pollution. As with any classification, however, it can be more or less arbitrary, not considering the many sources and agents acting simultaneously and often with catastrophic consequences for humans and nature. This chapter concludes with some warning examples of lake and reservoir mismanagement.

## 2.1 LAKE AND RESERVOIR USES AND FUNCTIONS

Before considering a global overview of uses and functions of lakes and reservoirs, the items in this section are summarized in Table 2.1. Many reservoirs, from the smallest to the largest ones, have been constructed with a single purpose in mind and have later been turned into multipurpose reservoirs. As an example, Lake Kariba was originally constructed for the production of hydropower, but is now also used for irrigation, fisheries, transport and recreation. Similar things can be said of many other reservoirs around the world.

### *2.1.1 Drinking Water*

The supply and storage of drinking water for humans and livestock is of fundamental importance in arid and semiarid regions. This reality is a causative factor for the onset of settlements and cultures along large river basins in Asia and Egypt. Generally, shallow groundwater was readily available. Thus, the establishment and use of wells (apart from

Table 2.1. Major uses and functions of lakes and reservoirs

- Drinking water
- Irrigation
- Flood control
- Fish production and production of other useful organisms
- Mining
- Fire- and ice-ponds
- Urban reservoirs
- Energy
- Industry
- Low energy purifiers
- Traffic on lakes and along lake shoreline and tributaries
- Recreation
- Conservation and biodiversity
- Training and education

rivers and springs) must have been the oldest source of water in such regions. However, in regard to irrigation and flood control, artificial lakes (such as in the Egyptian Saba Empire, in areas in the Middle East and Mediterranean regions, in Sri Lanka, etc.) have become very important. Likewise, dams in Central Europe became essential drinking water sources when the population increased during the Middle Ages (e.g., Harz, northern Germany; Pless, 1957). In the 20th Century, artificial lakes as drinking water supplies prevails in areas of low precipitation, or where low quality water from underground aquifers or rivers allows no alternative. Even in countries like Austria, which can provide high quality water for more than 160 million people, some lakes at convenient sites are sometimes used for drinking-water supply. Examples of natural lakes which represent the dominant water source in industrialized countries include Lake Constance (which has supported the city of Stuttgart, Germany, for decades), Lake Ladoga (which supplies St. Petersburg, Russia), and Lake Biwa in Japan (which provides drinking water for 13 million people). Prominent examples in the developing world include Lake Victoria and Lake Tanganyika in Africa. In addition to these examples, hundreds of reservoirs in industrial countries provide the water supply of cities (e.g., the Sosetal Reservoir, which supplies Bremen, Germany) and industrial areas, as well as in the developing world. Sophisticated technologies and careful consideration of the drainage basins of drinking water reservoirs has greatly contributed to their safe use.

### *2.1.2 Irrigation*

Agricultural irrigation is one of the oldest water uses originally associated with river systems. The annual water volume needed for this purpose has increased by almost 300% since 1900. At present, it totals more than 2700 $km^3$ (approximately equal to the volume of Lake Victoria), with the irrigated areas increasing five-fold since about 1900. Irrigated crops represent 17% of the total global agricultural area (Table 2.2).

Table 2.2. Irrigated areas and agricultural water demands between 1900–2000 (from UNEP/UNESCO, 1990)

| | 1900 | 1950 | 1990 | 2000 |
|---|---|---|---|---|
| Irrigated areas ($10^3$ km$^2$) | 473 | 1010 | 2720 | 3470 |
| Agricultural water demand (km$^3$ yr$^{-1}$) | 925 | 1130 | 2680 | 3250 |

Although dams and artificial lakes (apart from groundwater) were used to a large extent for irrigation in early culture, only much later did the exploitation of natural and artificial lakes became significant for regional irrigation. One of the most outstanding irrigation systems in Sri Lanka is presented by the "Jaya-Ganga", an 80-km long irrigation canal which connects two river systems and supplies a large number of reservoirs. The latter provide water for about 460 km$^2$ of paddy-fields. Another example of a natural lake with a much shorter tradition is Lake Biwa in Japan. Rice fields in the vicinity of this lake, Japan's largest, are irrigated from canals, with the assistance of water wheels (the "Jasha" and "Ryukotsusha") operated by human power. Since about 1950, large dams have contributed to the exponential increase in irrigated land. In semiarid and arid zones of Asia, India's demand of 800 km$^3$ of water annually for irrigation for an area of 550,000 km$^2$ amounts to almost 30% of the global irrigation, of which a major share is provided by dams. Similarly, in China and in the Middle East, artificial lakes and dams play an important role in expanding irrigated areas (e.g., about 460,000 km$^2$ in China). Many other countries (e.g., Spain) can be added to this list. In regard to cost-benefit evaluation, however, it should be mentioned that each case must be considered on its own merits.

More recently, irrigation has become a dramatic threat to large (mostly shallow) natural lakes. The diversion of their main inflows for irrigation has resulted in either total desiccation, destruction of traditional lake cultures (e.g., Lake Human-e Hilmand, Iran), or in dramatic changes in lake area, volume and salinity. Lake Aral is a globally-recognized case. In spite of repeated warnings by limnologists during the early 1980s, the main tributaries of the Aral Sea (Amu Darya, Syr Darya) became increasingly used for ambitious irrigation projects, mainly to increase cotton production in the region. Since that time, the lake's water level has dropped 15 m, causing a reduction in its surface area of more than 50%. The salinity has increased from 10‰ to almost 34‰, and continues to increase (Williams and Aladin, 1991; Ptichnikov and Nauber, 1994). Because of this salinity increase, and increased pesticide loadings to the lake, the fish harvest decreased from 44,000 tons/year to zero. About 60,000 people involved in the fishing industry lost their jobs, and the population in the vicinity of the lake has become exposed to adverse health conditions.

### 2.1.3 *Flood Control*

Along with the development of the earliest cultures and civilization in large Afro–Asia river valleys, humans learned not only to make careful hydrographic observations, but also to adapt technologies to help mitigate catastrophic flooding. Lake Möris, Egypt, provides a remarkable example of this marvelous technical adaptation. Many other retention basins in this country, Mesopotamia, Syria, and the Middle and Far East also should be mentioned.

Because of the prevalent climatic conditions in these regions, flood control was almost always combined with irrigation and drinking-water supply needs. In contrast, multiple-purpose dams from early Cretic–Mykenian and Greek periods were used exclusively for flood control. As early as the Second Millennium BC, dams in Boiotia, in combination with polders, were used to divert the rivers Kephissos and Melos and to protect the plain of Kopais, which could be claimed for settlement and agriculture. This design was remarkably similar to the current drainage of the Ijsselmeer in The Netherlands. In this regard, it is stressed that the Greek construction of dams had reached a most sophisticated stage. Careful consideration of the water pressure from one side, and possible erosion on the "air side", are among the remarkable achievements of that time.

At present, retention basins, reservoirs and polder systems, inundation plains, discharge channels and river regulation are still the main methods used for flood control. There are, however, still many unsolved problems with respect to flood control. Among these are the maintenance of valuable floodplain areas and their biodiversity in association with hydroelectric power stations, as well as the flood control of complex river sections affected by excessive regulation. The upper sections of the Rhine, with the application of retention basins, presents a paradigmatic example (Dister, 1985). From 1817 to 1880, radical regulation of the Rhine between Basle and Mannheim was carried out. Groynes were built to provide better facilities for the shipping traffic. This regulation caused greater erosion than had been foreseen, with a drop in the groundwater level and associated harm to agriculture being the inevitable consequences. Further, due to the reclamation of areas in the flood plain, the protecting dams of the flood plain were gradually moved toward the river, with the result that the final inundation space (reduced by about 130 $km^2$) no longer had sufficient capacity. Further detrimental measures included the construction of a Rhine canal for hydroelectric power between Basle and Breisach, exacerbating the groundwater situation, and the construction of "river loops" between Breisach and Strasbourg. In order to meet the risks of flooding, retention basins for almost 240 million $m^3$ of water were considered an appropriate strategy and were implemented. The final solution, however, will be the restoration of the former retention areas which, in combination with adapted dam systems and retention basins, could overcome the present problems (Dister, 1985). The disaster of the Gabčíkovo basin (Hungary, Slovakia) is similar for a large portion of the Danube floodplain.

### *2.1.4 Aquatic Production, Fisheries and Aquaculture*

Fishing has been practiced not only throughout the *Homo sapiens* period, but also most likely by pre-hominids. During the Middle Ages (since approximately 800 AD), the establishment of fish ponds in Europe was mainly associated with cloisters. It was only at a later stage that farmers were also permitted to build and possess fish ponds. As in early China, the common carp (*Cyprinus carpio*) was the principal fish kept in numerous ponds. Perch (*Perca fluviatilis*) and pike (*Esox lucius*) also became important and, with the onset of the Tang Dynasty in China, the common carp was replaced by the Chinese carps *Ctenopharyngodon idella, Mylopharyngodon piceus, Hypophthalmichthys molitrix*

and *Hypophthalmichthys nobilis*. In this regard, it is interesting that different cultures developed distinct preferences or aversions with respect to fish. In early Egypt, marine fish were considered unclean, and in the Jewish culture fish without scales (e.g., catfish, eel, lamprey) were condemned. Even in current times, carp is avoided by former British colonies, whereas it is preferred by most other nations and their former colonies.

The present global marine and freshwater fish harvest totals more than 100 million tons/year, to which rapidly-increasing aquaculture (at present about 20 million tons) contributes greatly. With Japan still leading (12 million tons), followed by China which may soon become the number one practitioner of aquaculture (10.5 million tons), and by Russia (still more than 10 million tons), a third of the global harvest is consumed. The United States, Chile, Norway, Peru, Denmark and Canada can be listed among other important fishing nations (1 million tons).

With respect to inland water fishery and aquaculture, China is clearly in the lead. Fishing harvest from natural waters totaled more than 680,000 tons/year in 1953. Because of land reclamation from lakes and increasing damming of rivers, however, this figure has dropped to about 300,000 tons. On the other hand, the production of cultivated freshwater fish in a total area of almost 30,000 $km^2$ has easily compensated for this loss (800,000 tons/year in the 1970s). There is no doubt that aquaculture will surpass the inland water fishery, if not the total fishing harvests, in the near future not only in China but also globally.

The adverse or even catastrophic consequences of the introduction of exotic fish to lakes also must be stressed. In Central Europe, the stocking of high alpine lakes, where fish were missing, started during the late Middle Ages and is still practiced (mainly with the alpine trout *Salvelinus alpinus*). This activity greatly affects the species diversity of invertebrates and should be prohibited. In fact, fish introductions are often carried out without any preliminary limnological surveys, which could have provided some predictions regarding the potential impacts of fish introductions. Moreover, during the late-19th and 20th Century, fish stocking increased to such a degree that evaluation studies became essentially impossible. In Saskatchewan, for example, 1.6 billion fish comprising 30 species were introduced to fresh and saline inland waters between 1900 and 1970 (Marshall and Johnson, 1971). The Neusiedlersee in Austria was not only stocked with eel (about 5–7 million fingerlings/year) for several decades after World War II, but also illegally with grass carp during the 1960s. Because of the grass carp, the submerged vegetation, which represents an important spawning site for several fish species, was almost completely destroyed. It is only slowly starting to recover. The damage done by eel to fish, amphibians, etc., in the lake was documented only at a very late stage, but nevertheless was stopped for several reasons some years ago.

In 1940, five salmoniform fish species were introduced into Lake Titicaca, although only the rainbow trout (*Oncorhynchus mykiss*) survived. Ten years later, the *Basilichthys bonariensis* from Argentina eventually entered the lake via the Rio Desaguadero. The endemic genera of Lake Titicaca (*Orestias*, *Trichomycterus*) were greatly affected by these exotic species, although the extent of the damage is still not fully understood. Similarly, the introduction of brown trout in Southern Victoria (Australia) has deleteriously affected endemic

salmoniform *Galaxias* species (Jackson and Williams, 1980). The best known and paradigmatic case of catastrophic effects from exotic species is Lake Victoria. Entering it from Lake Kyogo (where *L. niloticus* was introduced in 1957), the Nile perch (*Lates niloticus*) rapidly colonized the lake, being a major predator on the plankton and detritus-feeding endemic cichlids. Moreover, four alien *Tilapia* species previously introduced to Lake Victoria in 1951–1953 also compete with the endemic *T. variabilis* for nursery grounds (Serruya and Pollingher, 1983).

Commercial fishing in the North American Great Lakes started during the 18th Century, flourishing until the 1930s with highly esteemed species as American lake trout (*Salvelinus namaycush*), white fish (*Coregonus* spp.) and lake herring (*Leucichthys artedi*). Less valuable species such as yellow perch (*Perca flavescens*), alewife (*Alosa pseudoharengus*) and common carp (*Cyprinus carpio*; introduced in 1870) predominated thereafter. This change was related to the gradual immigration of fish species through the Welland Canal, constructed in 1929 to bypass Niagara Falls (Sarhage and Lundbeck, 1992). One of the immigrating fish species, the sea lamprey (*Petromyzon marinus*), damaged the stocks of the lake trout (*Salvelinus namaycush*), white fish (*Coregonus* spp.) and other valuable species. The lamprey plague was subsequently successfully controlled by electric fences and selective chemicals. Fish stocks, however, are presently endangered by pollution.

In Brazil, rather than cultivating local, easy-to-cultivate species, *Tilapia* have been introduced into small reservoirs since the 1930s. Although the annual fish production increased, it resulted in a setback on studies in the biology, reproduction and other features of local species. An Amazonian species, tucunare (*Cichla ocellaris*) was introduced into reservoirs of South Brazil, and to Lake Gatun in Panama (Zaret, 1980). This predatory species caused complete disruption of the local food chains. An introduction of a fish species with positive effects also was noted. Corvina (*Plagioscion squamosissimus*) was successfully introduced to reservoirs of Southern Brazil (e.g., Itaipu, Barra Bonita) (Agostinho et al., 1994). The reason for its success is that the species predominantly inhabits the pelagic region, which was not previously utilized before in these reservoirs.

In contrast to the adverse effects of fish stocking, positive examples also can be mentioned. One of the most spectacular successes was achieved in Lake Kariba. In 1967–1968, the freshwater sardine (*Limnothrissa miodon*), endemic to Lake Tanganyika, was introduced into Lake Kariba. This small, planktivorous fish reaches adult size after only three months, producing a harvest of 66 tons in Zimbabwe as early as 1973. By 1980, the annual harvest had increased to a maximum yield of 8000 tons. More recently, there have been decreasing, but still considerable harvests. *Limnothrissa miodon* also was successfully introduced in Lake Volta.

Macrophytes comprise another useful product from lakes. The common reed (*Phragmites communis*), papyrus (*Cyperus papyrus*) and tatora (*Scirpus tatora*) in Lake Titicaca have played a significant role for millennia in the construction of houses (especially in the Shatt el Arab area), boats, as thatching, production of paper, baskets, mats, etc. It also partly served as cattle fodder (e.g., Lake Hamun in Iran/Afghanistan) and for human nutrition (e.g., rhizomes of *Phragmites communis* in Central Asia and *Colocasia esculenta*). Further, submerged vegetation may be used to feed cattle and pigs (e.g., Lake Titicaca

area). In many tropical and subtropical countries, *Trapa* spp. is recognized for its nutritional value. In Indonesia, *Myriophyllum* spp. is used in salads, while it is considered a pest in Canada. In China and Japan, *Eichhornia crassipes* serves as small sewage treatment plants, for gas production and as pig fodder. Certain other plants (e.g., *Scirpus* spp., *Phragmites communis* and *Typha* spp.) have also generally become important as biological sewage treatment plants (Section 4.2.4).

Apart from fairy shrimps (*Artemia salina*, *Streptocephalus* and *Dendrocephalus*), and insects (*Dytiscus* spp., *Belostoma* spp., which are used for nourishment in North Africa and East Asia, respectively), invertebrates from lakes and reservoirs are generally only of secondary importance as food sources, compared to their marine counterparts. Among these are mussels, crabs, prawns and shrimps (*Macrobrachium*). Compared to marine aquaculture (e.g., 800,000 tons/year of the mussel *Mytilus edulis* in Norway) freshwater invertebrates only play a minor role, although promising activities have started in certain regions. The construction of Kariba Dam resulted in an increase of mussels, amounting to an estimated biomass of almost 170,000 tons. Certain species were selected and tinned. However, because of nonacceptance by the local population, this industry failed (Machena and Kautsky, 1988). In regard to mussels, the pearl industry at Lake Biwa, which still is economically feasible and flourishing, should be mentioned.

Finally, mention must be made of waterfowl, frogs and water-bound reptiles (snakes, crocodiles, etc.). The latter, however, are only locally cultivated and acceptable as food.

In Brazil, cultivation of alligators (*Caiman latirostris*) is expanding, being supported by the government. Both the meat and skin is used, the meat being sold to fashionable restaurants around the country. Cultivation is also used to re-populate regions originally inhabited by this species (e.g., the region of Pantanal). In Venezuela and Brazil, the large rodent of wetlands, capybara (*Hydrochoeris hydrochaeris*), is cultivated in great numbers (J.G. Tundisi, personal communication).

### 2.1.5 *Mining*

Mines, including abandoned mines, are nonpoint sources of pollution. An estimate of the magnitude of this type of pollution can be obtained by using a watershed-based approach (Caruso and Ward, 1998). More examples of the environmental effects of mining are given in Section 2.2.2—*Metals*.

*Iron and coal.* In some regions (e.g., Southern Sweden (Småland)), easily-soluble iron minerals can cause considerable formation of limonit (lake ore) in many of the lakes that contain low conductivity water. This iron ore was exploited and used for iron production until the 19th Century, when this kind of iron work became no longer economically feasible (Naumann, 1922; Montén, 1939; Thunmark, 1948). The lake ore was collected by rakes and dredging from rafts or, during the winter when the lakes were frozen, from ice. Only slight disturbances of the collecting sites occurred. On the other hand, strip mining (open-cast working) of iron or coal eventually resulted in the formation of lakes which, because of the presence of pyrites, most often became acidic, sometimes exhibiting pH values

below 3. Lakes with pH values below 3.5 sometimes lack both photosynthetic production and zooplankton or benthic animals. Moderately-acidic strip-mining lakes can be managed in the same manner as lakes affected by acid rain. The use of cyanides in gold mining is an environmental time bomb, as shown by the poisoning of the Tisza River from a Romanian mine in the year 2000, reaching up to the Danube River reservoirs.

*Minerals and salts.* The exploitation of minerals like diatomite can result in more serious impacts than the above-noted iron collecting activity. The diatomite operation in Lake Myvatn (Iceland) increased the rate of deposition in this lake area by 32% by the end of the 1970s (Jónasson, 1979). The diatomite excavation also has increased the phosphorus load by 20% and the nitrogen load by more than 90%. Thus, phosphate precipitation from mining effluents was encouraged, and may eventually save this lake area from unwanted *Anabaena* blooms. A Conservation Act for the Myvatn area was signed by the Icelandic President in 1974.

*Sand mining.* Exploitation of other minerals, such as sand and gravel, mainly results in the formation of new lake basins. It extensively affects reservoirs and rivers in South America. The suspended material is highly increased, causing decreased light penetration, destroying shallow areas in reservoirs representing fish nursery grounds, and decreasing oxygen concentrations and invertebrate populations decrease.

*Gold.* Gold mining also represents a potential hazard to lakes. In certain regions of South America (specifically Brazil), gold mining by hydraulic removal of bottom sediments of rivers has caused considerable damage to entire river ecosystems. Rio Madeira, for example, suffered extensive loss of fisheries. Further, mercury used for amalgamation of gold has contaminated several Amazonian rivers.

*Salts.* The use of salt by humans for spicing and conservation has occurred since the Neolithic period. Not so long ago, caravans crossing the Sahara from the Mediterranean coast to Timbuktu sold salt for gold. Salt mining was concentrated mainly in marine coastal areas. Continental deposits, such as the outstanding ones in the Salzkammerkut (Austria), were more likely to offer high quality salt than many of the saline lakes. There are, however, a few examples of traditional use of other salts. One example is the collection of soda from lakelets east of the Neusiedlersee (Austria) during periods of desiccation. Throughout the 19th Century, this soda was used for soap production. Salt mining in saline lakes most often became important in conjunction with the modern chemical industry. Examples include the extraction of soda from Lake Magadi (Kenya) for making glass, and from the Dead Sea for chemical plants in Israel. Most recently, industrial activities near large saline lakes south of Tehran (Iran) have been developed. Obviously, salt extraction from saline lakes causes relatively little disturbance (e.g., salt gardens in the Lake Urmia basin, Iran) if such lakes are not affected in regard to their salt and water budgets.

*Peat.* The exploitation of peat (as fuel, for horticulture or for medicinal purposes) is mainly restricted to mires, especially peat bogs. If such excavations occur in basins, they may become ponds or small lakes. Peat collection from peat bogs within sensitive watersheds can greatly influence the water budget and water quality of the relevant waterbodies.

*Gas.* Gas may accumulate in considerable quantities in the monimolimnium of meromictic lakes. If reasonable amounts of methane exist, in addition to $CO_2$, the latter may be used as an energy source. In the Lake Kivu basin, the most remarkable site of its kind, methane is used for the energy supply of a brewery.

*Mineral oil.* In a few natural (e.g., Lake Maracaibo, Venezuela) and artificial (e.g., Lake Kama near Perm, Russia) lakes, oil exploitation is performed, with a high risk of water quality degradation. The most spectacular case may become like Lake Tanganyika, where sediments up to 6000 m have accumulated since the Miocene. Based on more recent investigations, they contain more mineral oil than the presently-known reserves of the United States. All efforts should be undertaken to save Lake Tanganyika with its tremendous biodiversity, which has only been partly explored. Similarly, it should be mentioned that, in spite of heavy protests, a mineral oil pipe-line along the eastern shore of Lake Constance was established during the 1960's. Because of the tectonic activities in this area, the pipeline represents a permanent danger.

The Caspian Sea, Laguna Maracaibo, Venezuela, and Lake Kama near Perm, Russia, also create high risk situations with respect to water quality. In the Caspian Sea basin, where mineral oil exploitation has occurred in the Baku area since 1871 and has spread over many areas along the western and eastern coast with large numbers of open sea oil platforms. In areas of such offshore drilling activities, oil films can cover more than 800 $km^2$. In addition, the sea receives annually more than 100,000 tons of oil and derivates from the Volga (Ural–Volga oilfields) (IUCN, 1980; Koptyug and Uppenbrink, 1994).

Together with large quantities of other adverse substances (e.g., heavy metals) oil pollution in particular has contributed to great damage to coastal, benthic and pelagic communities. The extent of the dramatic impacts has been restricted so far mainly to the fishing industry; but even those evaluations are only of regional concern and often unrealistic. The establishment of the succession states of the former Soviet Union also has seriously affected the monitoring of the Caspian Sea.

### *2.1.6 Fire and Ice Ponds*

Until the early 19th Century, many villages in industrial countries were dependent on wells for their water supply. If not located close to a lake or river, in case of fire they were in need of fire ponds—which otherwise provided sites for domestic ducks and geese, especially in the Pannonian Region. These ponds were usually located in the center of villages ("village greens"). Since the Second World War, most have disappeared, being replaced by settlements, small parks, or even industrial entertainment. Hundreds, if not thousands, of ponds have been lost in this manner.

Before household cooling facilities were generally available in temperate regions with frost-period and warm summers, ice was collected during the winter from special ponds, mostly located outside the villages. This ice was preserved with sawdust and used in household ice boxes. One remarkable example is presented by a still existing ice pond in Lower Austria (Zwingendorf), which was built in an area of salty soils (i.e., containing $MgSO_4$,

$Na_2SO_4$, NaCl). Because of its remarkable saline flora and fauna, which developed more than hundred years ago, it was recently declared a national monument.

### *2.1.7 Urban Lakes*

Urban reservoirs are usually restricted in size, ranging from very small, pond-like waterbodies to impoundments of several million cubic meters. They are subject to very high anthropogenic pressures from surrounding communities, with the water quality in the reservoir closely tied to the hygienic and economic conditions of the surrounding populations. Even if most of the pollution is removed from the water, there often is a high level of eutrophication, due to the use of polyphosphate-containing detergents in the urban area. These nutrients eventually can reach a lake, particularly during heavy rain and storm events. Thus, the use of detergents associated with car washing, and similar activities around a lake, should be restricted. The use of chemicals in garden and lawn care also can be a source of toxins that can poison aquatic life.

The only management option specifically recommended for urban reservoirs is prevention of stormwater runoff (Novotny and Olem, 1994). Urban runoff collection is usually separated from standard effluent collection systems that are conveyed to the purification plant. This is because urban runoff consists of huge water masses with low organic matter content, but containing high turbidity and garbage. If this material was all conveyed to the purification plant, the capacity of the plant would quickly be exceeded. Increasing treatment plant capacities to meet the demands of storm events can be quite expensive. When stormwater is collected and treated separately, water quality is improved at lower costs. The response of phytoplankton to runoff consists of a phase of decreased biomass, due to decreased photosynthesis under low light conditions caused by a high input of turbidity, followed by a phytoplankton bloom stimulated by increased nutrient inputs and improved light conditions following sedimentation of the turbidity.

Most of the management options described in Chapter 4 are useful in managing urban reservoirs. Planting and protection of shore vegetation is highly recommended, since it serves as a natural pollution barrier. Fisheries that are regulated with respect to biomanipulation principles are feasible, particularly in small waterbodies primarily serving aesthetic purposes. Unregulated fisheries favoring extraction of predatory fish can result in water quality degradation. The presence of numerous aquatic birds is a sign of good water quality and proper regional management. However, if large flock of ducks or geese are fed by people, considerable pollution can result. A decision tree for urban lake management was developed by Birch and McCaskie (1999).

In larger urban reservoirs located in densely populated developing countries, the major problem is multiple-use conflicts. For example, the Xuanwu Reservoir in Nanjing, China, which has a volume of 3.32 $km^2$, but an average depth of only 2 m and an average theoretical water retention time of 54 days, is used as a tourist resort with swimming activities. However, it also serves as an aquatic farm, as an industrial and domestic water supply, and for agricultural irrigation. Consequently, pollution and siltation levels are very

high, with dredging activities becoming necessary by 1954. Without dredging, the reservoir would become filled with sediment in just a few decades.

### 2.1.8 Energy

Hydropower is one of the oldest energy sources used by humans. During ancient time, it played a significant role not only with respect to water-driven engines (e.g., for irrigation), but also for cultural purposes (e.g., the remarkable water-organ).

During the Middle Age, however, and especially from the 8th and 9th Century onward, the establishment of water mills of different kinds (e.g., for cereals, oil, textiles) increased exponentially.

Due to limited engineering capacity, the damming of rivers or valleys remained an activity of limited size until the 20th Century, when the onset of the construction of large dams (e.g., Lac Loutre, Canada, with a surface area 1295 $km^2$, opened in 1917) began.

From observations of meromictic lakes with a high-salinity monimolimnion, which sometimes became heated to more than 60°C, the idea of establishing artificially-stratified ponds ("*solar ponds*") as an energy source originated in Israel (Tabor, 1963, 1980). To date, however, the major problem of controlling the halocline has evaded solution. Even the reduction of water mixing, by the use of nets of floating windbreaks, did not succeed. However, there is some hope that solar lakes may eventually become important energy sources, especially in arid and semiarid regions (Serruya and Pollingher, 1983).

River and lake water is used for the cooling process for power plants. In some cases, this may improve desirable production in a waterbody. It seems more advantageous, however, to use the heated water from power plants for aquaculture and greenhouses before it is returned to rivers or lakes. This kind of application also provides more predictable results than the direct return of the heated water to rivers or lakes. In contrast to the more common impacts of heating to inland waters, the drastic cooling of lakes in connection with hydroelectric activities has been reported in a few cases. Lago di Cavazzo in northern Italy originally had summer surface temperatures above 20°C. Since the 1950s, however, when flushing by a pipe was started (15–40 $m^3\ s^{-1}$), the surface temperature has rarely exceeded 15°C. Because Lago di Cavazzo presents an important recreational site, the negative impacts on tourism has resulted in considerable economic loss for the region (Löffler, 1987).

Finally mention should be made of ponds and lakes heated by volcanic activity. Generally, however, volcanic energy from hot springs or vapor is used (e.g., Iceland, New Zealand). Lakes themselves tend to be only slightly affected. Rotamahana, New Zealand, for example, has raised temperatures by about 3°C, compared to noninfluenced lakes in the same vicinity (Hutchinson, 1957).

### 2.1.9 Transportation

Apart from continental channels, such as the Suez and Panama canals, which make use of intermediary lakes only for traffic convenience and economic reasons, the 3770 km shipping system of the Great Lakes in North America (started as early as 1783 and finalized

in 1959) presents the most important inland waterway, with an altitude difference of 182 m. Its traffic is largely concentrated on the lakes and the economically-most important harbors. More than 130 million tons/year passed through the "Soo" sluices alone between Lake Superior and Lake Michigan during the 1960s. Along the St. Lawrence Seaway, passage of 40 million tons was exceeded during the late-1970s. Due to the increasingly-competing capacity of road and rail traffic, a significant drop in shipping transport has occurred more recently. This also is evidenced by Chicago Harbor where, instead of the previous 1000 ships/year, only about 500 are unloaded in more recent years (see also Section 2.1.4 on fish in the Great Lakes).

Commercial shipping traffic also has a long tradition on other large lakes (e.g., Lake Ladoga in Russia; Lakes Vänern and Vettern in Sweden). In many European countries, the early onset of ship traffic was mainly related to transport and recreation.

From about 1840 onward, lake traffic was increasingly carried out by steamboats. This steamboat period, in some cases carried on until now because of recreation, is evidenced by the presence of soot particles in relevant sediment core sections.

Along with a switch to motorboats, speedboats and often excessive motor-boat oil spills have become a menace to water and fish quality in many lakes. This adverse situation is not confined only to industrial countries but also has been reported for lakes in developing countries (e.g., Lake Toba in Indonesia, personal observation H. Löffler). Thus, motor boating has been restricted to commercial activities in many lakes.

Traffic along lakeshores, or even crossing lakes, is always related to such impacts as the loss of shore sections, eutrophication and pollution. Because of technical limitations, the construction of traffic ways, especially in mountain regions, was most often carried out along lakeshores. Eventually, trails and paths evolved into roads for post coaches, and finally into highways for car traffic, with much more space being occupied. In contrast to the frequently-induced lake eutrophication caused by road construction, oligotrophication of lakes may occur under certain conditions. For example, the dumping of excavated rock material in Mondsee (Austria), in connection with road and highway constructions during the late-1950s and 1960s, resulted in high turbidity and the in-lake sedimentation of nutrients and algae. Consequently, oligotrophication of the lake, with a remarkable increase in transparency, occurred for a period of several years. The pollution impact from the erosion of tires and asphalt by traffic was documented for the first time during the "Lake Constance Project" between 1960–1969 (Borneff, 1962, 1963, 1974). For a daily rate of 4000 cars, it was calculated that about 1.5–2.0 kg of cancerogenic 3,4-benzypyrene would become available from a road section of 10 km (6 m broad). Similarly tear-products used for the isolation of buildings released 3,4-benzpyrene (see also Section 2.2.2—*Persistent Organic Pollutants*). Wherever lead-containing petrol is used, serious contamination of small waterbodies or sections of lakes located close to roads have been observed. Lead is converted into toxic organic compounds by bacterial activity. It should be mentioned that train traffic also may affect lakes if steam-driven (soot) or oil-driven trains are used.

Two case studies demonstrate the importance of traffic on lake conditions. Lago di Cavazzo, one of the largest lakes of the province of Friulana (Italy), and one of the

foremost recreation sites in this area, was not only heavily influenced by hydroelectric power activities, but also by the construction of the major highway from Austria to Trieste across this lake. The latter involved not only the severe destruction of the landscape, but most likely also caused water pollution impacts. Similarly, a pillar bridge project in 1972 across Neusiedlersee, 5 m above the mean water level of Austria's largest lake, and which since 1977 has become one of the countries most prominent Biosphere Reserves, threatened this site with heavy traffic and its polluting consequences. This project resulted in the first large-scale public intervention, which became a successful environmental initiative.

### *2.1.10 Recreation*

In Europe, lake-based recreation has a long tradition, comprising a variety of sports and games (on ice), as well as bird watching. In addition, salt lakes and lakes with mineral springs (e.g., Dead Sea) have become sites used for the sake of human health.

Since the early-19th Century, tourism and recreation in lake areas has increased exponentially (especially after the Second World War). Even lakes of modest size (e.g., a few square kilometers) may experience more than a million overnight bookings annually. Lakes in developing countries also have become affected more recently (e.g., Lake Toba, Indonesia; Lake Atitlan, Guatemala). Like wetlands, lakes are particularly vulnerable to mass tourism, which generally results in lakeshore destruction for housing, camping, car parks and traffic. The types of recreational activities, and their accompanying impacts on water quality, are summarized in Table 2.3.

In order to effectively address the unwanted impacts of recreation sewer systems and other problems, technologies had to be applied that eventually improved the quality of lakes. In Austria, the costs of these kinds of sanitation systems totaled more than US $1 billion annually between 1955–1990. However, there is no doubt that most of the destroyed lake shores will never be restored. Thus, the Austrian Economic Water Association proposed a resolution as early as in 1961, stressing the danger of further reclamation of lakeshores for housing and tourist colonies, and promoting their protection against destruction by technical projects. Thus far, however, these efforts have generally been in vain. Only a few countries have succeeded in passing legislation, according to which a landside fringe (50 m in Denmark) is fully protected.

### *2.1.11 Conservation and Biodiversity*

Conservation is understood and defined with a large variety of interpretations and perspectives, often adapted to regional conditions and need. Several authors (e.g., Williams, 1993) prefer the following definition, "Conservation is the management of human use of the biosphere so that it many yield the greatest sustainable benefit to present generations while maintaining its potential to meet the needs and aspirations of future generations". However, this is not necessarily the case for all parts of the biosphere. For many watersheds and their lakes, located in high-mountains, polar and arid regions, where human activities are unlikely for the foreseeable future, no management efforts seem necessary. This means

Table 2.3. Types of recreational activities on and around lakes, and their potential water quality impacts

| |
|---|
| *Recreation in the lake drainage basin* |
| Pollution from recreational facilities |
| The use of chemicals in gardens |
| Erosion connected with deforestation and road building |
| *Recreation at the lake shoreline* |
| Housing, hotels, kiosks—As in the drainage basin |
| Camping—Cause of serious impacts if sanitation and waste is not carefully controlled |
| Walking, picnicking, sunbathing, bird watching—Depends on the numbers of people and the sanitation facilities |
| Fishing—Possible destruction of shoreline vegetation |
| *Recreation on the lake surface* |
| Sport fishery—Interference with biomanipulation procedures; pollution caused by improper disposal of fish remains and fishing supplies; excess fish feeding; introduction of nonnative fish |
| Commercial fisheries—Impacts cause by commercial fisheries and aquaculture vary greatly, depending on specific methods. Organic loads associated with fish farming in cages and mussel farming can be greater than the amount of organic matter removed from the lake during fish or mussel harvest |
| Swimming—Stirring of sediments can cause turbidity and increases in proliferation of *E. coli* bacteria, hygienic impurities, danger of infections |
| Scuba diving—Rarely a source of pollution |
| Canoeing, rowing, windsurfing—Negligible problems within the lake; potential impacts by associated shoreline activities |
| Sailing—Large sailboats can cause a major pollution source (see Houseboats) |
| Houseboats—Input from sanitary systems which can include sewage, detergents, excess organics and trash |
| Motor boating and water skiing—Shore erosion caused by waves, oil and fuel pollution |
| Cruisers and large boat traffic—Danger is minimized by hygienic arrangements (dry toilets, preservation of wastes and wastewater) |
| Ice skating—No known impacts |

that such lakes should be left to their natural physiographic conditions which, at least for long periods, facilitate the persistence of their biological communities including migrants such as birds and mammal.

In 1959, the Societas Internationalis Limnologiae (SIL) initiated "Project Aqua", the first international attempt at lake conservation (Luther and Rzóska, 1969, 1971). The goal of the project was "to get international recognition for a list of fresh-water and brackish water areas which are of agreed international importance for research, education or training, and where therefore the countries should accept national responsibility for

conservation". At the time, this was an important first step directed to the protection of lakes and at present, more than 25 years later, most of the selected 600 inland waters (including running waters, springs and wetlands) are still of relevance. More than 150 lakes and wetlands are represented among the 411 Biosphere Reserves of UNESCO (year 2002) distributed over 94 countries.

Since this list was designed for the activities of the International Biological Program, it is desirable that categories such as lakes not affected by humans and, as a high priority, old lakes originating during the Tertiary or during the early Pleistocene, be included. In fact, many of the old lakes are already listed in the Project Aqua.

Since "Project Aqua", the Ramsar Convention and the Biosphere Reserve activity of UNESCO, the World Cultural and Natural Heritage and many other international protection efforts have stimulated the selection of important lakes and their possible conservation.

Thus far, however, the protection of lakes is still far from satisfactory. In spite of the great number of lakes which, after careful assessment of their values have been proposed as relevant sites, corresponding legislation and its execution are most often not sufficient. Such legislation, which would be simple for national lakes that have not been significantly affected by human activities (e.g., Torneträsk in Sweden), becomes almost unpractical in the case of multinational lakes with conflicting interests, especially where existing traditional uses are difficult to control.

Accordingly, efficient legislation should comprise:

- The control of the hydrological regime (water retention time),
- The protection of littoral zones and their vegetation (building activities and traffic should be avoided to the maximum extent),
- The sustainable exploitation of salt and other minerals or organic matter,
- The mitigation of ongoing eutrophication or pollution which can eventually decrease diversity,
- The sustainable management of fisheries and, in most cases, avoidance of the introduction of exotic species or even exotic genetic material, and
- The sustainable management of tourism and recreation.

With regard to exotic species, it should be stressed that for a few exceptional lake cases, the biodiversity has actually increased because of the introduction of exotic species, without harm to native or even endemic fish. The introduction of tilapia (*Tilapia grahami*) into highly-alkaline Lake Nakuru (Kenya), where fish had been absent, resulted in an abundance of fish-feeding birds (e.g., pelicans, cormorants). Lake Kariba also was stocked with the freshwater sardine (*Limnothrissa*) without any disturbance to the native communities of this dammed lake, as well as a dramatic increase in fish production.

### 2.1.12 *Training and Education*

Aquatic ecosystems offer many possibilities for environmental training on all educational levels. Its comprehensible objects, watersheds, rivers and lakes, facilitate reasonable access to the understanding and functioning of ecosystems, their metabolism and energy flows.

In addition, the relatively easy construction of ponds or artificial lakes offers excellent facilities for the study of different essential properties.

The role of aquariums and small ponds in public- and high-schools has recently found a wide acceptance, although these kinds of activities in teaching basic biology in most countries are still left to the initiative of engaged and enthusiastic teachers. Moreover, wherever possible, children could be invited to protect a small lake, pond or wetland, and also should be trained for their basic management. Such small ecosystems can become "the property" of a school-class. Selected ponds, or any kind of inland waters, can be turned into sites of permanent public information, and may include facilities to measure and observe some of their basic features. At present, very few localities of this type do exist. International and nongovernmental organizations (e.g., UNESCO, IUCN, WWF, ILEC (International Lake Environment Committee), SIL (Societas Internationalis Limnologiae)) should be invited to promote these kinds of public information sites.

Consistent with the title of this book, training of students, engineers, aquaculturists, etc., for lake and reservoir management should be given high priority. Technical measures concerning sanitation, restoration, mitigation of hazards and the establishment of integrated systems for sustainable production will certainly undergo rapid development in the near future.

## 2.2 LAKE AND RESERVOIR MISMANAGEMENT

A difference exists among the many types of lake pollution and other types of human abuses to lakes in different regions, particularly between the Northern and Southern Hemispheres (as characterized in Chapter 1, Fig. 1.14), both in the degree of occurrence of different types and in the time sequence of their appearance. Some of these abuses are treated in Section 2.2.1 and the types of pollution in Section 2.2.2.

### *2.2.1 Abuse of Lakes and Reservoirs*

*Wastewater disposal*

Since the last century, small artificial and natural ponds (e.g., backwaters of riverine areas) have increasingly become sites for waste dumping, eventually becoming completely filled. This water material often contains toxic substances likely to degrade the quality of groundwater. As a result of such (often illegal) deposits, large regions may become heavily polluted, with necessary restoration activities being not only costly, but sometimes (e.g., for large aquifers) even impossible. In spite of strict laws in many countries, this kind of dumping is still occurring, representing a large-scale environment problem (especially in regard to drinking water).

*Industrial impacts*

In some cases, large lakes are still used to receiving industrial wastes, an adverse tradition that became established during the 19th or early-20th Century. One of the most dramatic

examples is represented by Traunsee (Austria's deepest lake) where, since the turn of this century, the alkali works at Ebensee have released up to 30 tons daily of largely insoluble substances (e.g., $CaCO_3$, $MgCO_3$, $CaSO_4$) and about 200 tons of soluble compounds (e.g., $CaCl_2$, NaCl). The latter have resulted in periods of chemical gradients in the lake and, under meromictic conditions, the insoluble material has piled up to more than 40 m close to the maximum depth area and near the mouth of the affluent. Similar accumulations of industrial or mining activities are known from several natural and artificial lakes (Kama Lake near Perm, Ural; personal communication to H. Löffler).

Industry and mining also have contributed significantly to the destruction, pollution, acidification and alkalification (Millstätter See, Austria, magnesite works) of lakes and lake shorelines. Lake Erie (USA–Canada) and Lake Orta (Italy) also can be mentioned as examples. The latter was heavily loaded with copper and ammonium sulfate by a Canadian factory producing rayon during the period 1926–1983. The maximum copper loading exceeded 50 tons/year and 2500 tons/year for nitrogen (ammonia). The consequences included a dramatic decrease in pH (to 4 and less) and the disappearance of the pelagic organisms (although a few uncommon species sometimes appeared for short periods). Because of a treatment plant established in 1981, a first recovery phase has been observed since 1983 (Calderoni et al., 1992).

*Reclamation and destruction of lakes*

Apart from lakes which became dry due to irrigation and eventually were used for agriculture, thousands of shallow lakes, ponds and puddles were filled or drained, and reclaimed for forestry or agriculture during the 20th Century. In China, 543 big and middle-sized natural lakes, and many more smaller ones (with areas less than 1 $km^2$), as well as additional reservoir area and regulating volume (totaling approximately 11,585 $km^2$ in area and more than $570 \times 10^9$ $m^3$ in volume) vanished within 30 years (1950–1980). A dramatic example of reclamation is presented by the shallow Neusiedlersee at the Austrian–Hungarian border (see Section 9.1). Cases of accelerated sedimentation in lakes often are due to activities within the lake drainage basin, including deforestation, mining, road construction, etc., followed by land erosion. The construction of a forest road close to Lunzer Untersee (Austria) caused a landslide of a forested area of more than 1 ha which, for a short period, produced a considerably increased sediment (and nutrient) load in a small effluent draining the eroded area. A greater rate of sedimentation was induced by changing the position of the entrance of the Rhine River into Lake Constance in 1900. Until that time, the Rhine River entered the lake at the so-called "Rheinspitz", when, according to a treaty between Austria and Switzerland, the mouth was shifted to flow into Fussach Bay about 10 km east of the old site. The construction of the new river bed resulted not only in a reduction of the relevant river section by 60% (from 12.4 km to 4.9 km), but also in the protection of a large area of the Rhine River valley from flooding. Because of this technical impact, the river supplies an average annual suspended load of almost 2.6 $km^3$, which has resulted in a decrease of the lake area (539 $km^2$) by approximately 1.2 $km^2$. A rapid

increase of sedimentation caused by man's activities (less so by natural events such as volcanic ash; Edmondson, 1984) has been reported for numerous lakes, and also presents the special feature of many lakes held behind dams (e.g., Lake Nasser) and in river dams (e.g., Danube, Aschach, annually totaling up to 80 cm).

Most pre-alpine lakes lost their natural littoral zones because of rapidly-increasing tourism, becoming occupied either by sport- and other recreation centers, and by summer resorts activities which can cause eutrophication and pollution. One example of shoreline destruction in the developing world is represented by Lake Atitlan in Guatemala, where an endemic coot species incapable of flight has disappeared. Rules for the protection of lakeshores have been established in only a few countries (e.g., Denmark), restricting building activities to a minimum distance from the lake shoreline. Very recently, shore restoration has become a new strategy in other countries (e.g., Austria, Germany, Japan, Thailand). Likewise, lakes which have become dammed for hydroelectric power have lost their natural shores (i.e., loss of macrophytic vegetation) because of regular changes in their water levels. Examples of this kind of impact are common in many parts of the world, especially in mountainous regions such as the Alps. One case of extensive, and fairly recent, lake destruction is represented by the Aral Sea (see Section 2.2.2—*Salinization*).

In arid and semiarid countries, desertification (also called dryland degradation) is a cause of lake destruction. This was reported in one study as a major threat in 27% of the developing countries (Fig. 2.1). The cause is either climatic stress or human mismanagement, or both. Population growth connected with the over-exploitation of water resources, land use changes which expose soil to erosion, overgrazing and deforestation are major causes.

In conclusion, the importance of the conservation of (shallow) lakes and small bodies of water, especially the saline ones (Williams, 1993, 1996), must be stressed. Since the early-19th Century, an area of more than 50 $km^2$ in the natural area of lakes has disappeared in Austria. A tentative estimate of this loss on a global level would total at least 100,000 $km^2$. On the other hand, the fast increasing number of artificial lakes must be considered more than compensation for this loss.

### 2.2.2 *Pollution of Reservoirs and Lakes*

Water pollution—the loading of waterbodies with both inorganic and organic matter, with radioactive substances and with undesirable organisms (weeds, invertebrates and vertebrates)—is generally associated with dense, fast-increasing population levels and associated land use changes within a given watershed (e.g., deforestation, agriculture). The importance of pollution is generally related to drinking water quality, domestic use and water conditions desirable for certain recreation activities, such as bathing and swimming.

This section stresses human demands for drinking water, and for high-quality water associated with recreation and biodiversity maintenance. From this perspective, lake pollution comprises a large variety of different items, including:

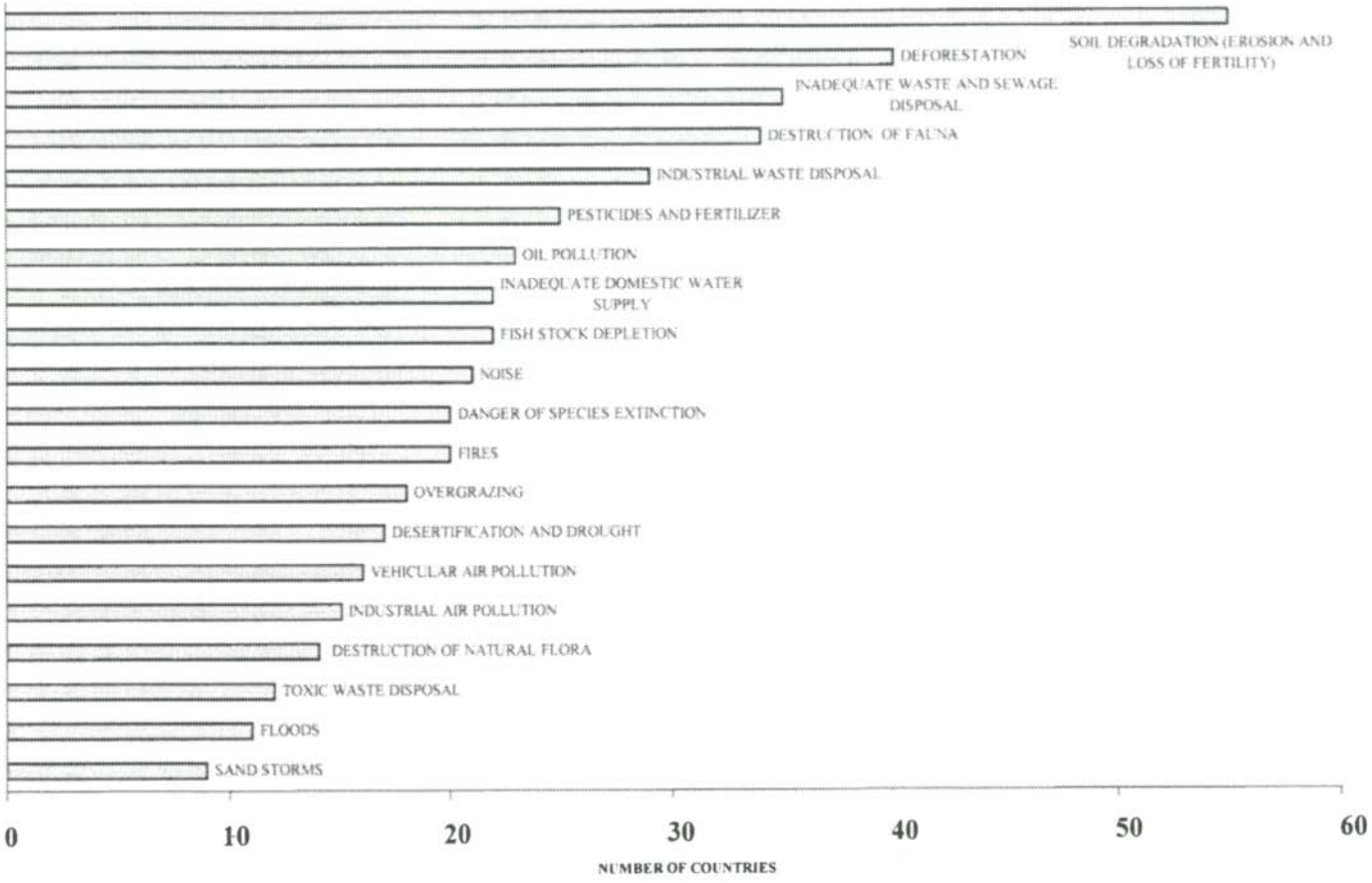



Fig. 2.1. Review of environmental problems in developing countries (modified from Nakamura).

- *Turbidity*—being of natural origin, but strongly enhanced by anthropogenic effects,
- *Sewage pollution*—general domestic pollution from population, the degree of which is lower in rural communities with no sewage service, but highly increasing with increasing urbanization and industrialization (if not decreased by purification),
- *Eutrophication*—a global phenomenon caused by excessive phosphorus and nitrogen loads to waterbodies, and closely related to rapidly-increasing watershed populations, agricultural activities and urban drainage, poor forestry management, deforestation and certain types of industrial activities,
- *Acidification*—a regional phenomenon caused by acid precipitation in areas of igneous rock and poor soil and water acid-buffering capacity (e.g., Eastern USA and Canada, Northern Europe and parts of Central Europe and Asia),
- *Siltation, radioactive matter, thermal pollution, ammonia and nitrate, salts, heavy metals, organic matter (including mineral oil) and pesticides*—local pollution with a great variety of adverse impacts, and
- *Biological pollution*—such as the sporadic immigration and introduction of exotic species and the presence of parasites.

Between the appearance of a new threat and its full impacts, as well as an apparent result of preventive and remedial measures, there is typically a degradation phase. Figure 2.2A illustrates the duration of these phases for countries with advanced sewage treatment. In less-developed countries, the transition periods are not only shifted in time, but may also be highly prolonged. Most severe is the situation in rapidly-developing countries, where many types of pollution appear nearly simultaneously, then becoming difficult to identify the individual causes and their significance in the overall pollution picture (Fig. 2.2B).

*Turbidity*

Soil degradation, including erosion and loss of fertility, is considered the most significant environmental problem in developing countries (Fig. 2.1). Increased erosion, particularly related to strong rains and runoff, causes turbidity with several consequences for lake and reservoir water quality, including:

- Mechanical filling of the lake basin,
- Increased organic matter input (clay particles contain organic matter subject to decay, both during suspension and after sedimentation),
- Increased bacterial production and formation of clay–organic-bacteria aggregates (Lind and Davalos-Lind, 1999),
- Increased phosphorus loads, representing the dominant part of the sum of all phosphorus sources from agriculture (on the one hand, the turbidity contributes to phosphorus concentrations in the water column; on the other hand, clay particles compete with plankton for phosphorus and can transport phosphorus to the sediments),
- Increased heavy metal concentrations, depending on their presence in the soil because of the application of fertilizers and from other sources in the drainage basin,
- Decreased light availability for phytoplankton photosynthesis, leading to water quality improvement, or at least compensating for photosynthesis resulting from increased phosphorus loads (Lind and Davalos-Lind, 1999),

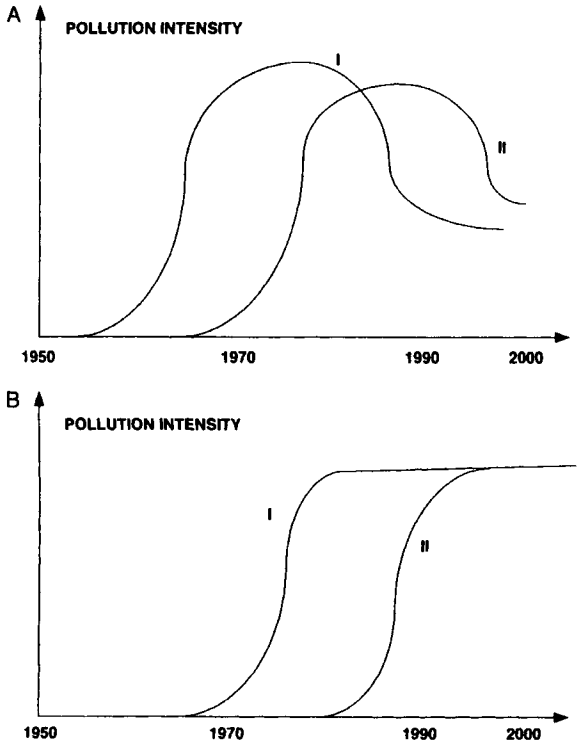



Fig. 2.2. Differences between pollution sequences in industrial countries (A) and developing countries (B). I represents an early starting country and II a later starting country.

- Interference with trophic interactions in lake plankton, particularly decreasing the visual grazers of visual grazers (Cuker, 1987).

The general nature of suspended material in the aquatic environment is described by Eisma (1992).

Summer temperature and primary production profiles of clear and turbid lakes can be quite different. The surface temperature of a turbid lake is higher than that of a clear lake because the particles causing turbidity can adsorb the incoming solar radiation in the uppermost layers. Clean water allows a deeper penetration of solar radiation into the water column, resulting in a heat gain in the deeper water strata. In a turbid lake, oxygen is produced by photosynthesis within shallower water strata than in a clear lake. Water strata exhibiting no photosynthesis, therefore, are more extensive, extending the oxygen deficit. Oxygen conditions are worse in turbid waters because of the increased nutrient load, and the production of organic matter associated with particles, as well as decreased re-oxygenation associated with photosynthesis due to light limitation of primary production. Light penetration also is important for the distribution of macrophytes in lakes (Duarte

Table 2.4. Management consequences of turbidity related to inorganic particles (modified from Hart and Allanson, 1984)

| |
|---|
| Positive effects: |
| • Costs of producing high quality water are low, since particles are easily flocculated |
| • Reduction of the potential for algal and submerged macrophyte growth |
| • Increased fishery potential due to associated nutrient inputs |
| Negative effects: |
| • Increased capacity to transport nutrients and toxic substances |
| • Reduced storage capacity of reservoirs and impoundments, as a consequence of settling out of riverborne inorganic suspended material |
| • Reduced fishery potential, due to high sustained levels of turbidity, which cause large zooplankton species to be grazed and restrict the littoral to a small vertical depth |

and Kalff, 1987, 1990). The influence of turbidity on lake water quality is summarized in Table 2.4.

*Sewage pollution*

This represents the classic pollution historically created with the advent of water canalization. Before canalization, most organic matter ejected by humans was recirculated back to fields, going then straight back into receiving waters. With the development of urbanization and modernization, the organic matter became accompanied by many plastic materials that are not easily decomposable. This is creating major water quality problems, particularly in the megacities of developing countries.

The effect of sewage pollution on a waterbody is illustrated in Figure 2.3. In addition to increased concentrations of organic matter in the inflows to lakes and reservoirs, there also is a lower oxygen concentration associated with the decomposition of organic matter. The concentrations of nutrients also increase, unless the influent water receives tertiary treatment directed to the extraction of nutrients. Up to the present time, this treatment is more easily applied to phosphorus than to nitrogen. The load of pathogenic bacteria is increased. The organic matter decomposes further in the lake, particularly in the sediments where it can accumulate. Oxygen consumption by the sediments can lead to anoxia which, in reservoirs, can extend from the inflow region where most sedimentation usually occurs. Increased releases of nitrites, ammonia, iron, manganese and phosphorus are consequence of the changed chemical redox potential associated with anoxia. The presence of gases, including carbon dioxide, methane and hydrogen sulphate, increases in the hypolimnetic water. Refractory organic matter is a consequence of the partial mineralization of organic matter from sewage. The passage of the water through the lake, however, also markedly increases the quality of the water, due to self-purification processes in the lake.

The consequences of high levels of sewage pollution on water quality are extensive. Human health problems of various kinds are created. Adequate drinking water treatment is made more difficult, costly or even impossible. Fish meat can become of lower quality and even inedible, with resistant fish becoming dominant. Corrosion of structures also can occur, with a strong ebullition of gases in extreme cases.

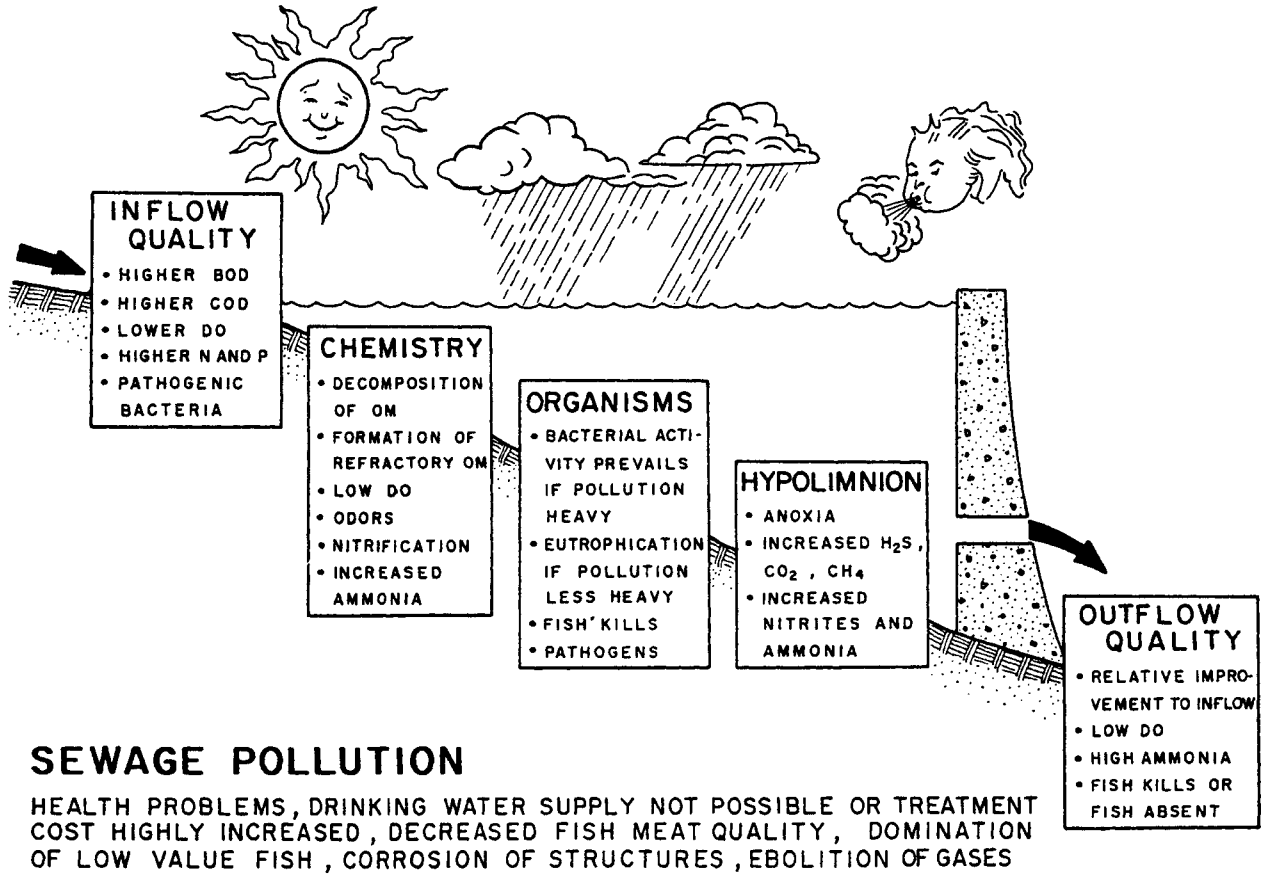



Fig. 2.3. Schematic representation of the effects of sewage pollution on lakes (modified from Straškraba and Tundisi, 1999).

*Eutrophication*

*Trophy* refers to the quantities of nutrients entering a lake (i.e., the nutrient load). Higher nutrient loads typically produce higher primary production by phytoplankton and macrophytes. Thus, *eutrophication* equates to both high nutrient loads and high aquatic production. High nutrient loads are the dominant cause, and high production the dominant consequence, creating most water quality problems associated with eutrophication. It is not the concentrations of phosphorus and carbon in a lake *per se* that cause water quality problems; rather, it is the production of phytoplankton and macrophytes resulting from the nutrients that creates the problems. The situation is different with nitrogen, in that high concentrations of nitrogen compounds in water also can have health consequences for humans, in addition to being an aquatic plant nutrient. *Oligotrophic* waterbodies (from oligo = poor) receive less nutrients from their drainage basins and, therefore, exhibit lower phytoplankton production. As a result, their water is typically very clear. Waterbodies with extremely low nutrients loads and levels of primary production are called *ultra-oligotrophic*. In contrast, *eutrophic* (from eu = rich) waterbodies are rich in nutrients, and have high levels of biological production supported by their high nutrient loads. The waterbodies lying between these two nutrient extremes are called *mesotrophic*. *Hypertrophic* refers to waterbodies extremely rich in nutrients and, therefore, also containing high phytoplankton concentrations. Higher nutrients loads are usually (though not necessarily always) associated with higher loads of organic matter from a lake's drainage basin (i.e., *allochthonous* organic matter). Phytoplankton and macrophytes can produce organic matter in a waterbody (i.e., *autochthonous* organic matter), and the consequence of the decomposition of this matter being water quality degradation. Oxygen is consumed by the microbial organisms that decompose the phytoplankton and low oxygen (anoxic) conditions can result, particularly in the hypolimnion of stratified waterbodies. There are several nutrients, or biogenic elements, necessary for phytoplankton and macrophyte growth. However, their relative quantities necessary for optimal growth are very different. The most abundant nutrients are phosphorus, nitrogen and carbon, which are typically utilized by algae and other autotrophs in a proportion corresponding to their relation in algal cells. This is the basis of the so-called Redfield ratio of carbon to nitrogen to phosphorus (i.e., C : N : P = 106 : 16 : 1 by atomic weight). If these nutrients are present in a waterbody in approximately this ratio, the growth of algae is not limited by any of them, but rather depends on the absolute quantities (i.e., concentrations) present in the water column. However, when the in-lake proportion differs from this ratio, the nutrient that exhibiting the most deviation is the one that limits the algal production (i.e., it is the limiting nutrient). The limiting nutrient in most waterbodies is phosphorus, with excess nitrogen typically being present. In contrast, although nitrogen does not limit algal biomass in many freshwater bodies, it is generally the limiting nutrient for the production of marine phytoplankton, and can create eutrophic conditions in closed seas (e.g., Adria, Baltic or Black Sea).

From the management perspective, therefore, reduction of the quantities of the limiting nutrient entering a waterbody, and the associated decrease in the primary production in the waterbody, usually results in a reduced production of organic matter (i.e., phytoplankton) and an associated improvement in water quality. If the load of the limiting nutrient is high,

there is essentially no other logical way to significantly and reasonably improve the water quality of a lake than to reduce the nutrient input. Almost all other management methods can only be successful when the in-lake nutrient concentrations are low. However, once a waterbody becomes loaded with phosphorus over a number of years, large quantities of phosphorus may have accumulated in the lake bottom sediments. Once the external phosphorus load is reduced, therefore, it will still take several years before the phosphorus in the waterbody and its sediments is flushed out and the organic production decreased.

The nutrient load is not the only variable responsible for differing quantities of algal production in lakes and reservoirs. The same load of incoming nutrients can produce different consequences in different waterbodies, depending on their size, depth and water flushing rate. Shallow and/or smaller waterbodies generally are more eutrophic then deeper, larger ones.

Eutrophication as a process of water quality deterioration is most often due to human influences, including nutrient loads from both point and nonpoint sources. The increased nutrient loading is a result mainly of cities, settlements, recreational pressures, industry, deforestation, agriculture, erosion, damming of lakes and rivers, river diversions, poor watershed management and, last but not least, from atmospheric pollution. The latter plays a significant role in remote high-alpine lakes, which typically are not direct impacted by human activities in their drainage basins (Psenner, 1994; Löffler, 1995b). However, eutrophication (Hutchinson, 1973; Vollenweider, 1979) also can be related to natural events in exceptional cases, such as the large erosion processes associated with the retreat of glaciers during the late Pleistocene Era, or the forming of the lakes behind major landslides, which can result in excessive nutrient loads from decaying terrestrial vegetation and other natural organic matter.

The effect of eutrophication on a lake or reservoir is shown schematically in Figure 2.4. Only the qualitative picture is presented, with the quantitative aspects being dealt with in other parts of this book, including Section 3.2.3, and in Section 4.2.1 in regard to nutrient sources. Eutrophication models are discussed in Section 5.5.1. Major eutrophication processes in lakes are related to nutrient inputs from the drainage basin, and the in-lake activity of algae, Cyanobacteria, macrophytes and associated microbes and fauna. The influence of increased nutrient concentrations on algal biomass in lakes is nonlinear. With increasing concentrations of the limiting nutrient, the algal biomass first increases slowly, and then more rapidly, followed by an almost linear rise until some asymptotic value of biomass is reached at high concentrations of the limiting nutrient (Fig. 2.5). Increased algal activity associated with eutrophication increases the oxygen content of the water column, and reduces its pH due to the utilization of carbon dioxide ($CO_2$). However, this sequence is only valid in the upper water strata, and during thermal stratification, when water mixing oxygen concentration values can be very low, and the concentration in the hypolimnion can drop to zero. This can result in an increased release of phosphorus and heavy metals from the sediments back into the water column.

The water quality effects of eutrophication include generally-decreased water quality, increased costs of drinking water treatment, and taste and odor problems in drinking water. Filters used in water treatment can become clogged, requiring frequent cleaning. The

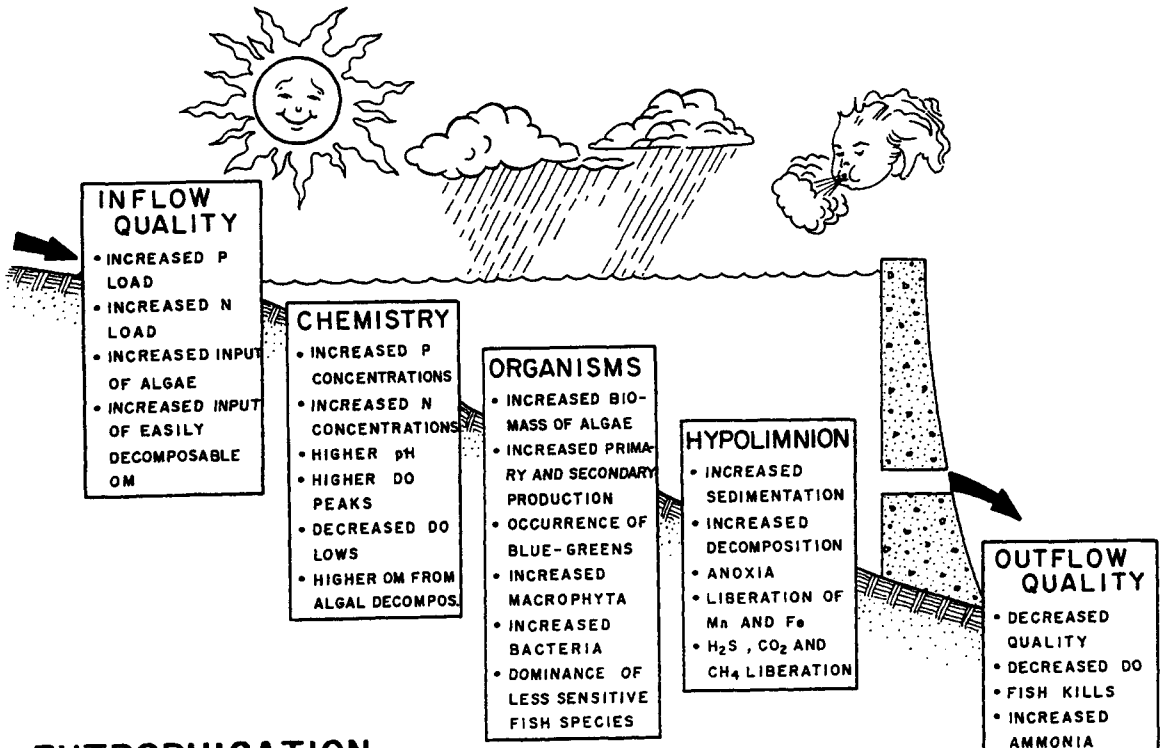



Fig. 2.4. Schematic representation of the effects of eutrophication on lakes (modified from Straškraba and Tundisi, 1999).

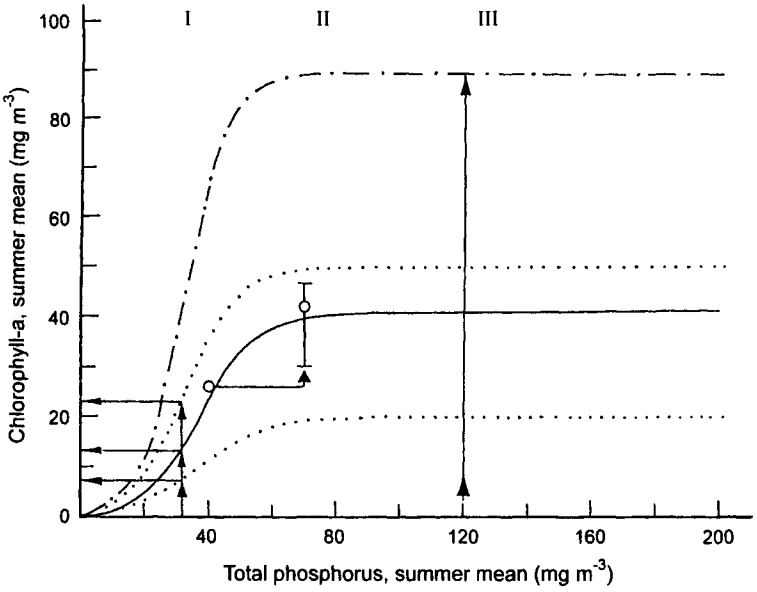



Fig. 2.5. Nonlinear reaction of lake eutrophication to phosphorus reduction (Straškraba et al., 1979).

organic matter produced by algae can create water treatment difficulties, including algal toxins and the formation of trihalomethanes in drinking water, and the presence of mucopolysacharides, causing problems for certain industrial waters. Mechanical difficulties can be created by the clogging of tubing because of organism growths, and by corrosion from high carbon dioxide and methane concentrations in extremely eutrophic waterbodies. Recreation can be threatened by high concentrations of algae, including changes in the algal species composition, and an increased presence of toxic Cyanobacteria creating health problems (including the death of humans and cattle).

Fish productivity also is increased in eutrophic lakes, which is a positive feature from the perspective of increased food production. However, the fish populations frequently change from cold-water to warm-water species that are less desirable from a recreational perspective. With further increases in eutrophication (i.e., hypertrophic waters), fish populations can collapse, due to lack of oxygen and an excess of toxic gases in the waterbody.

In addition to increased nutrient loads, organic matter also increases due to inputs from different diffuse (nonpoint) sources in the drainage basin, including the formation of organic matter in the inflowing rivers by phytoplankton and its subsequent decomposition. If point sources of nutrients are not sufficiently controlled, as is the case in many developing countries, the joint impacts of eutrophication and organic pollution must be seriously considered within the context of lake management efforts.

Most of the above-noted eutrophication symptoms and processes are related to deep lakes. Shallow lakes, however, react differently from deep lakes in some aspects. One important variable for phytoplankton development is the depth of the mixing water layer,

related to the availability of light for the phytoplankton population dispersed in this layer. The narrow mixing layer of shallow lakes provides more light for phytoplankton productivity. As a result, their primary production is higher than for deep lakes receiving the same nutrient load. Thus, the phytoplankton biomass is higher in shallow lakes, and they have a tendency to be more eutrophic and more prone to Cyanobacteria blooms. In the absence of permanent thermal stratification, the oxygen concentration is fairly high at all depths, including the near bottom water layer. More details on the differences between shallow and deep lakes are presented in Section 3.2.1.

*Toxic Cyanobacteria and other noxious algae.* Certain toxic strains of algal species, including *Microcystis*, *Anabaena*, *Aphanizomenon*, *Synechococcus* and *Cylindrospermopsis*, often co-exist with nontoxic strains in the same waterbody. Some of the associated toxic substances (microcystin, anatoxin, saxitocin, cylindrospermopsin) have been analyzed over the last decade. To at least some extent, they are considered to be a means by which algae protect themselves against grazing by zooplankton. It has been observed, for example, that Cladocera unable to select nontoxic algae stop their water filtration in their presence. The occurrence of toxin-producing Cyanobacteria seems to increase with increasing eutrophication in both tropical and temperate regions. Although earlier records were dominantly from warm regions, Willen and Mattson (1997) recently reported that 47% of Cyanobacteria samples from Sweden contained toxins, compared to 44% in Finland and 48% in Norway. The occurrence in France is even higher, with Vezie et al. (1997) detecting toxins in more than 70% of their water samples. A detailed review of the health effects of Cyanobacteria was recently presented by Chorus and Bartram (1999) and Chorus and Cavalieri (2000).

Toxic, or at least allergic, effects also may be caused by some other algae, with cases of skin irritation and allergies reported for forest lakes in Sweden, because of the flagellate *Gonyostomum semen* (Cronberg et al., 1988). A recent plague resulting from intensive eutrophication in Asia is "red tide", caused by the alga *Peridinium bipes*. In Japan, red tide has been observed in reservoirs throughout the country.

Serious health events related to Cyanobacteria have been reported from many countries (Falconer, 1989, 1993). The review of Chorus and Salas (1997) on the health impacts of freshwater algae summarizes 12 case studies and reports on acute intoxication with Cyanobacteria. Human deaths have been reported from Brazil. In the town of Caracau (State of Pernambuco), 55 hemodialysis patients died of liver disease, seizures or acute hemorrhaging after receiving dialysis treatment using chlorinated water from the city reservoir. According to Chorus and Salas (1997), the cause of the deaths was Microcystin. After the closure of the Itaparica Reservoir in Brazil, 87 people died from drinking water from the new reservoir after the development of *Microcystis*. In another case, 32 deaths were reported, following 110 cases of human poisoning. Earlier in another lake (Lago del Parque), 72 cattle deaths were reported in 1973. Kontek et al. (1997) have studied the mutageneticity of toxins from Cyanobacteria.

The observed human health effects (Lawton and Codd, 1991) range from contact dermatitis, eye irritation, fever-like symptoms, gastroenteritis (resulting in diarrhoea and

vomiting), basal pneumonia, hepatitis, endotoxaemia, and one observation of a suspected cause of birth defects being the drinking of water from a water supply lake with *Oscillatoria subbrevis*. Some of the signs on the skin and eyes of swimmers is assumed to be related to allergic reactions, rather then to direct toxic effects. The most extensive cases of poisoning thought to be related to Cyanobacteria involved 6000 U.S. soldiers from a base in the Philippines that used drinking water from a Cyanobacteria-infected river, and about 5000 people becoming mildly ill after drinking from an infested reservoir in Pennsylvania (see Section 4.3.5 for Cyanobacteria treatment). Falconer (1993) provides a good review of algal toxins as they appear in drinking water.

Another negative health effect related to algae and Cyanobacteria is the production of cancer-promoting trihalomethanes, which are formed as byproducts during the chlorination of drinking water. They are considered particularly to be responsible for causing bladder cancer. It appears that different types of algae exhibit similar effects in regard to trihalomethane formation, as evidenced in experimental studies with the Cyanobacteria *Anabaena flos aquae* and diatom *Asterionella formosa* by Graham et al. (1998). Toxic Cyanobacteria may occasionally cause fish kills. The decomposition of blue-green algae also may produce sufficient quantities of hydroxylamine to become lethal to fish.

*Present state of eutrophication.* Since the 1960s, eutrophication of many lakes and reservoirs in most industrial countries has been successfully treated. At the same time, however, eutrophication is increasing in developing countries, where the economy generally does not provide for costly sanitation facilities or lake restoration efforts. It also must be admitted that "re-oligotrophication" of lakes in industrial countries has often not resulted in the same conditions existing in the lakes before the onset of the increased nutrient loadings (e.g., organic materials and nutrient concentrations, water transparency, biodiversity). Rather, the conditions have improved only to the extent deemed desirable for human use of the lakes and reservoirs with respect to drinking water quality and recreational use.

One of the best known examples of lakes in industrial countries that have been successfully restored from eutrophication is the case of Lake Washington (Edmondson, 1991). Based on analysis of long-term data sets, the lake underwent two phases of eutrophication in the 1920s and 1950s. Based on limnological studies and careful monitoring, as well as public outcries lead by Edmondson and his co-workers, a sewage diversion scheme was set up in the late 1950s. After the diversion of the sewage was begun, the lake responded almost immediately, returning in the early-1970s to the conditions existing before 1950 (Edmondson, 1972; Edmondson and Lehman, 1981; Crul, 1995). The situation in Lake Constance (Fig. 2.6) also illustrates the degree of improvement that can be obtained.

Among the Laurentian Great Lakes of North America, only the largest ones (Superior, Huron) are still oligotrophic, exhibiting only local environmental pollution problems. This is due to the climate being too cold for intensive agriculture, and only a small number of large cities on their shorelines. In contrast, Lakes Michigan, Erie and Ontario have numerous industrial cities along their shorelines and in their drainage basins, with their more fertile soils also being subjected to intensive agriculture. In spite of its relative short water retention time, the most serious eutrophication problems have occurred in shallow Lake Erie. Domestic sewage from cities such as Cleveland and Detroit, and effluents from

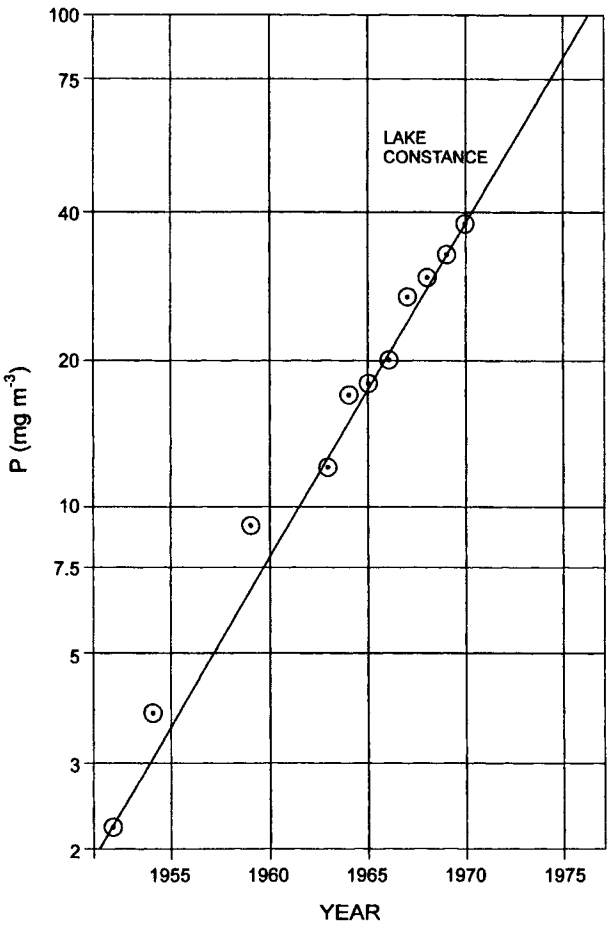



Fig. 2.6. Eutrophication of Lake Constance has shown in the period indicated an exponential trend, presented in the figure in linearized form. The trend was later stopped by the introduction of tertiary treatment, and reversed by the ban on phosphates in detergents (from Stumm and Baccini, 1978).

industries, caused severe eutrophication. A phosphorus restoration program stimulated by limnologists and Great Lakes residents, within the context of the U.S.–Canada Great Lakes Water Quality Agreements of 1972 and 1978, resulted in a sharp decrease in the phosphorus loads and concentrations in Lakes Erie, Ontario and Michigan. Chemical pollution is presently the main pollution problem in these latter lakes (Lee et al., 1978; Crul, 1995; also see Section 9.2).

Lake Tahoe in the Sierra Nevada mountains of the western United States was originally an ultra-oligotrophic lake, with Secchi readings of more than 30 m. Over the three last decades, however, the rate of eutrophication of the lake has accelerated, due to large-scale

tourism and recreational activities. Inflows, groundwater seepage and atmospheric inputs of nutrients (the latter from car emissions, wood-burning stoves and possibly forest fires; Goldman, 1990) have contributed to decreasing transparency and increasing algal growths (Goldman, 1993, see also Crul, 1995). Relevant limnological studies within the lake and its drainage basin again became the basis of a legislative strategy of the Tahoe Regional Planning Agency to address the ongoing eutrophication of this unique lake.

Only little progress has been achieved in Central and Eastern Europe, where increasing recreational pressures (e.g., some Mazurian Lakes in Poland) have most often resulted in increased eutrophication. Related to rapidly-increasing tourism, the shallow Lake Balaton in Hungary is of special interest, since it not only represents a major modelling case study of the International Institute of Applied Systems Analysis (IIASA), but also a site of major restorational activity. Since one of the lake's most important nutrient loading sources is its largest tributary, the River Zala (which enters the western tip of the long-stretched lake at the lower part of its floodplain) was turned into a large wetland. As a result, the water retention time of this river section was increased and the lake's water quality improved. Further, the restoration site became an outstanding bird sanctuary (Istvanovic and Somlyódy, 1999). However, it also is obvious that, in view of the fast-growing recreational activities, this important local measure alone cannot improve the overall nutrient condition of Lake Balaton. In fact, much more effort must be expended to save Central Europe's largest lake.

Lake Baikal (Russia), with its unique plant and animal life, is threatened at least locally by industries and agriculture. Its main pollution (including eutrophication) sources include the inflowing Selenga River, which carries pollutants and nutrients from industrial plants in Ula Ude, the cellulose plant in Baikalsk (southern shore) and the area around Severobaikalsk (northern end of the lake). Since the 1970s, modest efforts have been made to reduce the pollution of the cellulose plant in Baikalsk. So far, however, only limited success has been seen.

In contrast to the ongoing efforts against eutrophication in industrial countries, virtually nothing has been achieved in most developing countries. Even more so than in Eastern Europe, economic shortcomings, as well as lack of interest on the part of decision makers and local or regional people, are among the major reasons. Generally, eutrophication began with increasing population pressures, especially so in Africa during the 1960s. Exceptions are China (e.g., Lake Dianchi) and India, where eutrophication had its onset at least several decades earlier than in Africa. A dramatic example of the first increasing population pressure on lakeshores in Africa is Kampala, the capital of Uganda.

With the exception of some 60 high mountain lakes in East Africa (Löffler, 1964) and Ethiopia, oligotrophic lakes (in terms of water transparency) are represented by some of the deep Rift Valley lakes, especially Lake Tanganyika (although local eutrophication is becoming of increasing concern). Most of the other African lakes have high nutrient loadings related to human activities or naturally-high nutrient concentrations, some examples being the East African shallow alkaline lakes (Talling, 1957; Symoens et al., 1981). Among the large African lakes, Lake Victoria is subject to lake-wide eutrophication, and a general limnological transition which occurred between 1960–1990 (Hecky, 1993). This may be

due partly to higher temperatures throughout the water column in most months of a recent period of observation during 1989–1991 (compared to 1960–1961), but is more likely due to increased point and nonpoint loadings, as well as the dramatic increase of the Nile perch (*Lates niloticus*) in the 1980s, about 25 years after its introduction to the lake (Crul, 1995). This sudden increase led to a dramatic decline of the food-web structure in Lake Victoria. Lake Tai-Hu, the third largest freshwater lake in China, is faced with two conflicting requirements, including increasing the aquatic productivity and improving the water quality to meet the needs of the growing population in this lake basin. A new type of aquatic agriculture has been developed there to optimize the use of the lake's food chain with the establishment of integrated systems. These include growing selected submerged macrophytes in the area surrounding fish pens, with the macrophytes being used as fish food. The areas for plant and fish culture are changed from year to year for water quality conservation. Macrophytes also mitigate wave action and improve the water quality by absorbing nutrients. The annual fish harvest totals between 22–30 tons/ha.

Increased use of this more traditional kind of integrated systems has been made in various countries, especially in Southeast Asia. These countries are threatened by rapidly-increasing population pressures and, therefore, an increased need for more protein resources, resulting in overstocking with fish. Moreover, the use of pesticides, as well as other kinds of pollution, have had adverse impacts. These types of impacts can be observed in the numerous lateral lakes of Chang Jiang (Yangtze River). One example, among many, is the East Lake of Wuhan (Dong Hu) on the northeastern outskirts of Wuchang. This shallow lake serves as a water supply and commercial fishery. With the increased urbanization in its drainage basin over 40 years, a number of changes occurred in the lake, including an increase in the total solids (dissolved and suspended) concentration from 80 mg $l^{-1}$ to 192 mg $l^{-1}$, and in the total phosphorus concentration from 9 mg $l^{-1}$ to 150 mg $l^{-1}$. The previously-dominant submerged macrophytes, such as *Potamogeton maackiansis*, disappeared and others became impoverished, most likely because of overstocking with grass carp. Moreover, Cyanobacteria became increasingly dominant, while Bacillariophyta decreased rapidly. On the whole, the lake has approached "the upper limits of eutrophication" (Liu, 1984). A more general, but extensive, review of the state of a number of Chinese lakes and reservoirs is now available (Jin, 1995). The trophic level of major lakes and reservoirs in China was assessed independently by Shu and Wuang (1993). Similarly, many other lakes which were investigated in 1929 have undergone major changes, especially in regard to eutrophication and siltation.

The destruction of natural habitats by human activities in India has been far more extensive and complete than perhaps anywhere else in the world (Mani, 1974) and should be given special attention. Because of this situation, oligotrophic Indian lakes are restricted to high mountains, whereas lakes of mountainous zones below 2000 m already suffer from population pressures, tourism and, therefore, also from erosion and eutrophication. Very high nutrient loadings and corresponding algal blooms characterize the lakes and tanks (small artificial reservoirs) of most of the lowland areas in India.

In South America, Lake Amatitlan in Guatemala is negatively affected by population growth, deforestation, inadequate land use and industrial development. Although situated

in a unique landscape, no significant lake management program exists. In Nicaragua, Lake Managua (Lago Xolotlán) and Lake Nicaragua suffer from untreated sewage and industrial discharges, including heavy metals. Moreover, they also contain a multitude of pesticides from agricultural and pasture lands. Unfortunately, no rational development plans have been elaborated to date for these lakes. Lake Valencia, the largest and oldest (end of Pliocene Era) lake in Venezuela, presents a disastrous eutrophication (and pollution) case, related to intensive human intervention in its drainage basin and the input of untreated wastewater from domestic, agricultural and industrial activities of about 2 million people. The use of the lake is restricted by its high salt content. Permanent algal blooms, high fish mortality, and odors prevent the use of the lake for drinking water, aquatic sports or tourism. When Gessner visited the lake in 1952 during his Venezuela Expedition, he called it one of the most beautiful lakes of South America (Gessner, 1955a, 1955b). However, sometime between Gessner's last observations in the 1960s and a later census in the 1970s (Infante, 1978), the lake obviously deteriorated completely. A project for the environmental recovery of Lake Valencia and its drainage basin was recently developed, including the proposed construction of at least three sewage plants, to be financed by the Inter-American Development Bank.

Nutrient loads from dry and wet precipitation also endanger oligotrophic lakes in the high alpine regions of Europe (Löffler, 1995b), and perhaps also Lake Victoria (Hecky, 1993). Alleviation of this problem clearly requires regional cooperation. In this connection, it should be stressed that a global monitoring program focusing on the phosphorus contents of precipitation is desirable (Section 3.4.2).

A more global picture of the state of world lakes can be obtained from the ILEC (International Lake Environment Committee Foundation) series of Data Handbooks (Kira, 1994, 1995a, 1995b).

In conclusion, in contrast to the natural nutrient loading of lakes, human-induced eutrophication is the most common and, therefore, most important problem affecting the beneficial use of lakes and reservoirs around the world (Kira, 1993). The increased biological production associated with eutrophication may only be considered desirable from the perspective of food production. However, in regard to domestic water demands, biodiversity, and recreation, eutrophication is considered an undesirable condition, and can often result in human health problems. It became a major problem in industrialized countries more than a half century ago, whereas it became a major problem in developing countries mainly after the 1960s (especially in Latin America and Africa), primarily in relation to the rapidly-increasing population pressure.

*Rate of lake recovery from eutrophication.* The recovery of a lake from eutrophication depends in part on the quantity of phosphorus that has accumulated over time in the lake bottom sediment, and the water volume in contact with the sediment. For most lakes subjected to reduced external phosphorus loads, the in-lake phosphorus concentrations decreased, with corresponding decreases in chlorophyll concentrations (an indicator of algal biomass) and increases in water transparency (Fig. 2.7A, B). However, in cases of high sediment phosphorus accumulation, particularly shallower lakes, the recovery of the lake proceeded at a much slower rate than the reduction in the external phosphorus load.

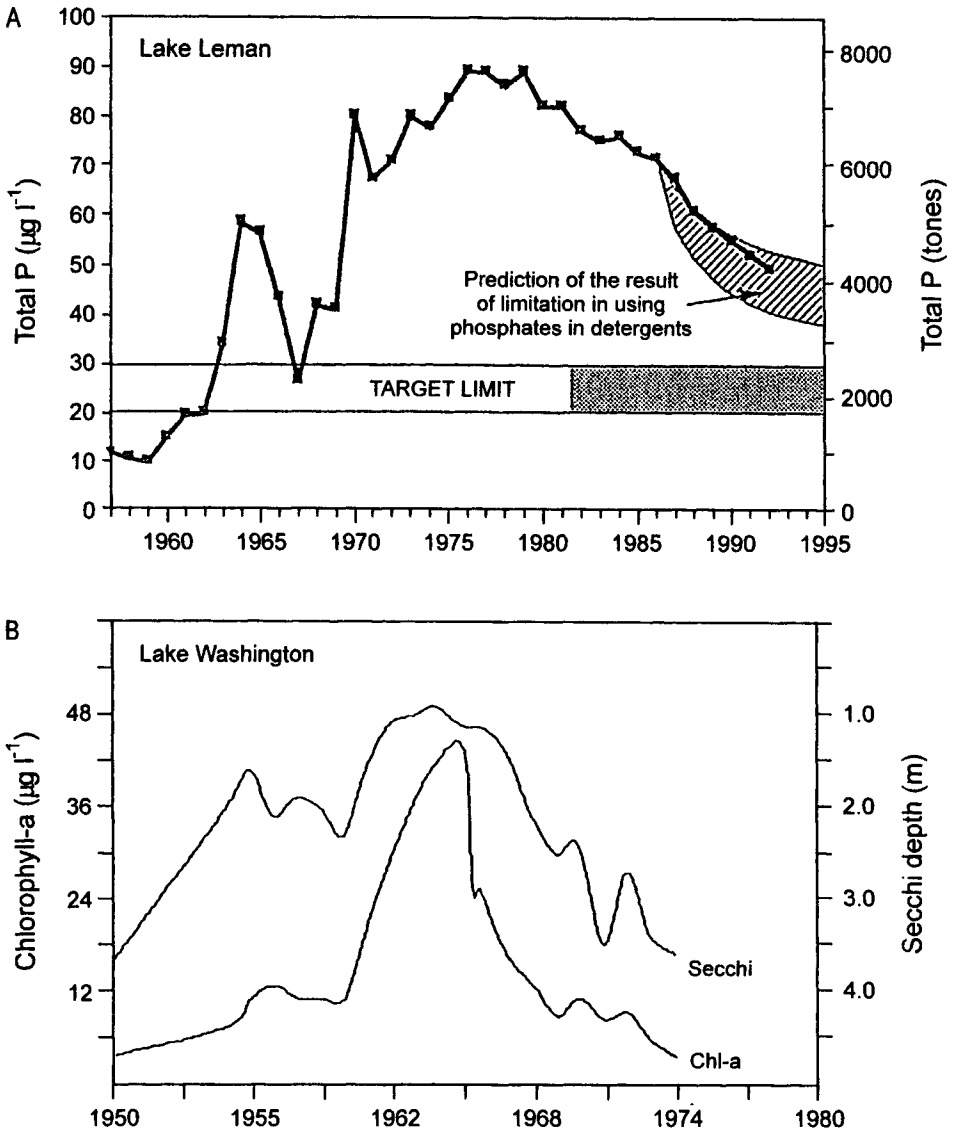



Fig. 2.7. Improvements of water quality after reduction of external phosphorus sources in the drainage basin. A—Lake Leman, Switzerland. Although rapidly approaching the target level marked by the dotted area, it was not achieved in 1995. The first arrow (1972) indicates the start of tertiary treatment, the second (1986) the ban on using phosphates in detergents (from Rapin et al., 1995); B—Lake Washington in the United States.

The accumulated phosphorus in the sediment moved by diffusion into the now more dilute phosphorus content in the hypolimnion, lengthening the phosphorus flushing process by a number of years. The lakes of the Berlin area provide an example, for which Wolf et al. (1988) reported a five-year lag period between the initiation of lake restoration efforts and full lake recovery (cf. Fig. 5.16). Another constraint to immediate lake recovery is the nonlinear, asymptotic character of the relationship between the quantity of algae and the phosphorus concentration (see Section 3.2.2). When the phosphorus concentrations in a lake are very high prior to initiation of external phosphorus load reductions, even a very significant decrease in the phosphorus concentration does not necessarily result in a significant decrease in the chlorophyll-a concentration (Fig. 2.5; also see Ryding and Rast, 1989).

Recent information and guidance on the assessment and management of eutrophication in lakes and reservoirs is summarized by Rast and Thornton (1997) and IETC (1999).

*Acidification*

Changes in lake acidity are generally associated with acid precipitation, which is presently confined mainly to industrial regions of the Northern Hemisphere. However, since lake acidification depends on the degree of susceptibility of a given lake and its drainage basin (i.e., its acid-buffering capacity), its distribution is limited to the northeastern United States and the adjacent parts of Canada, Scandinavia, parts of Great Britain and Ireland and small areas in Central Europe (e.g., Czech Republic, Germany, Poland, Slovakia) and Asia.

The effect of acidification on water quality is illustrated schematically in Figure 2.8. The main impacts are due to the effects of inflows, which can have high concentrations of toxic aluminum (Morris et al., 1989) associated with low pH and modified nutrient loads, compared to nonacidified lakes, because of chemical reactions in the drainage basin that can lead to decreased nitrogen concentrations and increased concentrations of phosphorus compounds. When the pH of a waterbody is low, due not only to the effect of strong acids from acid rain, but also because of humic (fulvic) acids (Hessen and Travnik, 1998), further chemical and physical–chemical reactions can modify the availability of nutrients to phytoplankton. Thus, the eutrophication of these waters is decreased (Mason, 1990; Psenner, 1994; Steinberg and Wright, 1994).

The main water quality problems of acidified lakes are that the potable water supply is endangered due to increased concentrations of heavy metals, particularly aluminium, in the water. The reduction of aquatic life is a major treat for fishery and recreational uses.

It is noted, however, that lake acidification may be a long-term natural event in some cases, if a lake's drainage basin becomes increasingly covered by peat bogs due to climatic changes. Such transitions are known, for example, from the early Holocene Era of Finnish lakes.

Among the human causes of lake acidification, various types of pollution and the treatment of lakes affected by blooms of blue green algae with copper sulfate (e.g., Lake Mendota, Wisconsin) should be mentioned. Further, lakes formed in areas with strip mines (mainly brown coal and iron ore mines) are likely to become acidic. Fonseca (1990) reported a case of stream acidification by acid mine drainage.

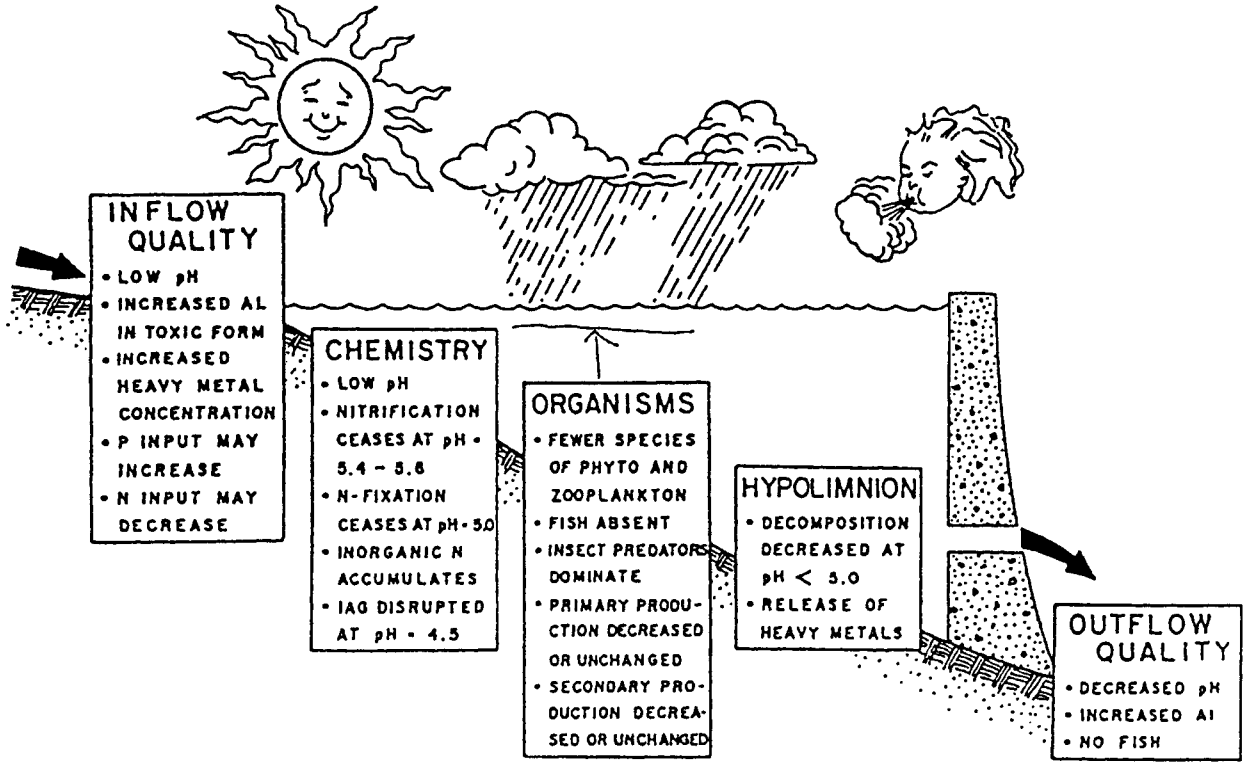



Fig. 2.8. Schematic representation of the effects of acidification on lake water quality (modified from Straškraba and Tundisi, 1999).

The starting point for increased atmospheric loadings of acidifying compounds such as sulphur oxides ($SO_x$), nitrous oxides ($NO_x$), etc., was about 1850, the time when sediment cores from acidified lakes began to exhibit high concentrations of carbonaceous particles derived from fossil combustion. It is of interest that, at some sites, conifer afforestation may contribute to acidification indirectly though, for example, the erosion of peat following land drainage (Battarbee et al., 1988). The onset of lake acidification is variable over the past 150 years. For many lakes, it occurred only after the World War II (e.g., lakes of the High Tatra in Poland and Slovakia). An example of long-term increasing acidification was presented by the lake Cerné Jezero in southern Bohemia, for which more than 120 years of data exist. This small (18 ha) lake was excavated by a glacier during the last glaciation, and presents a typical cirque (Kar) with a maximum depth of 40 m. When it was researched for the first time in 1871, it had a rich zooplankton community, high biomass level, moderate transparency (2–4 m), and a pH of about 6. About twenty years later, the brook trout (*Salvelinus fontinalis*) was introduced to the lake and may have contributed to the disappearance or large zooplankton species, as observed in the early 1890s. However, a further drastic reduction of species was observed about 40 years later, when fish in the lake finally became extinct. Cerné Jezero presently has a pH of 4.5 and a transparency of 10–15 m. The only remaining zooplankton species is the rotifer, *Microcodon clavus*, known to occur in peat bogs and obviously a recent immigrant. As usual, the acidification of this lake has resulted in high aluminium concentrations (0.185 mg $l^{-1}$), which may account for the disappearance of fish, and a relatively high copper concentration of 1.6 mg $l^{-1}$ (Fott et al., 1994).

Lake acidification became dramatically apparent with large fish kills in Norway and Sweden in the early 1980s. In Sweden, about 17,000 lakes (from a total of 85,000 medium and large lakes) are presently acidified, 4000 of them seriously.

Apart from abiotic parameters of non-marine origins (e.g., $SO_4$ concentrations), paleolimnological analyses have become a major focus of relevant research. It is based on diatoms, chrysophytes (*Mallomonas*) and certain other groups of algae which clearly show changes in species composition caused by decreasing pH values over the last decades. In contrast to the above, zooplankton (almost exclusively Cladocera remains) and benthic animals such as mollusks and ostracods started to disappear in proportion to their tolerance to different levels of acidification. The same is true for fish, among which *Umbra pygmaea* and tench *Tinca tinca* have been identified as the most tolerant species thus far in regard to acidity (pH < 4). This acidity tolerance, however, depends on the aluminium concentrations (especially $Al(OH)_2^+$), and a decreased species richness and abundance possibly occurs between a pH value of 4–6. Reproductive failure at a sublethal pH appears to be the main cause for the decrease (Gahnström and Andersson, 1988a, 1988b). These and other results on the physiological, morphological and behavior features have been obtained from lab studies and field observations. Field observations also contributed to the knowledge of the effects of acidification on organisms unlikely to leave remains in the sediment, such as copepods or most of the rotifers. Apart from in-lake bag-enclosures, artificial acidification of a small lake in the Canadian Experimental Lakes Area provided the basis for a better understanding of community reactions to increased acidity. Thus far, however, only little

attention has been given to the littoral fringe, which may act as a refuge for organisms otherwise missing in the rest of the lake. Terrestrial wash-in processes may provide for increased earth alkaline materials and, hence, provide a buffering capacity in this region.

*Radioactive and thermal pollution*

Radioactivity pollution associated with nuclear power plants has received attention in many countries, having now been monitored for thirty years (e.g., about 300 locations in France). The catastrophic Chernobyl event (ROSTE–UNESCO, 1994), which was analyzed specifically in regard to the impacts from the fallout of such radioactive elements as cesium ($^{137}Cs$), strontium ($^{90}Sr$), and phosphorus ($^{239}P$), has demonstrated that drinking water accounts for approximately 5% of the radioactive intake into the human body, whereas the main source of internal irradiation is food (including a not-defined quantity of fish). One of the important results arising from the Chernobyl catastrophe is that it clearly shows that the human response to radionuclide pollution is much more serious than suggested by the nuclear power industry, and that the consequences of exposure to ionizing radiation are of a long-term nature.

Thermal pollution (Langford, 1990), which was discussed recently in connection with the climatic change problem (e.g., since the 1970s Lake Victoria has been subjected to an increase of more than one degree) is otherwise restricted mainly to rivers and lakes, which provide cooling water for atomic and fossil fuel power plants. Although of local importance, and seen mainly in connection with production (e.g., sun energy deliberately used in oyster cultures), thermal pollution has been investigated more carefully than other lake pollutants. A well-studied case of a lake heated by a nuclear power plant is Stechlin Lake in Germany (Casper, 1985).

If global warming is to realistically considered, it is quite clear that one of the most serious impacts would be a switch from dimictic to warm monomictic circulation, which would provide for disastrous anoxia events. Much less harm, although often incorrectly stressed, will occur to shallow lakes, which are already subjected to large thermal changes from season to season and year to year as a result of stochastic meteorological events.

Thermal pollution also may be associated with major cooling events, if a lake is flushed artificially to some extent with cold water from a river outside its drainage basin. Such flushing actions may be desirable for improving lake water quality, but also may result in major recreational impacts. For the latter, Lago di Cavazzo in northern Italy presents a remarkable example of the misuse of an important resort. With summer temperatures well above 20°C, this lake was widely utilized for swimming and bathing, until it began to be utilized as a flushing site for an hydroelectric power plant. Since that time, its temperature has only reached 15°C during the tourist season. Improvement of its fishery (i.e., trout instead of carp) has so far not satisfied the local population, which not only has lost its recreation site, but also an economically important resource.

*Ammonia and nitrate*

Ammonia is produced by heterotrophic bacteria as a product of the decomposition of protein or other nitrogenous inorganic compounds. It is present in water mainly as ammonia

ion ($NH_4^+$); under alkaline conditions, it is present as undissociated $NH_4OH$ ($NH_3$). At a pH value of 9.5, the ratio of $NH_4^+$ to $NH_4OH$ in a waterbody is about 1 : 1, compared to 300 : 1 at a pH value of 7. In contrast, at a pH value of 10.5, only $NH_4OH$ is present. $NH_4OH$ is highly toxic to many organisms, especially to fish. Many cases of sudden fish kills have been reported for eutrophic and poorly-buffered fish ponds which, due to algal assimilation, are likely to shift to high pH conditions. Cases of fish kills due to ammonia also may be observed in regard to sewage loadings or for certain alkaline lakes.

In most surface waters, nitrate is present in concentrations that do not pose human health risks, especially for small children who may suffer from lethal cyanohaemoglobinaemia if they ingest nitrate. The present upper tolerable drinking water nitrate concentration is considered by the European Union to be 25 mg $l^{-1}$, by the World Health Organization to be 50 mg $l^{-1}$, and recently by Austria to be 100 mg $l^{-1}$. The upper limit of 100 mg $l^{-1}$ is only seldom reached in nature. The higher concentrations are most often found in groundwater, sometimes over extended areas, as a result of agricultural activities that facilitate nitrate (and pesticide) concentrations above those considered safe for human health. This may imply large scale drinking water problems of major economic concern.

It is noted that the construction of artificial waterbodies (e.g., fish ponds, ponds for recreation or water storage) often may dramatically influence the quality of groundwater resources. Similar to seepage lakes, artificial basins, especially in gravel areas, tend to have hydrological connections with groundwater aquifers and, therefore, may act as pollution sources. Monitoring of such loading processes often can be difficult. Thus, preventive laws concerning the establishment of artificial ponds in important groundwater areas seem to be essential.

Burt et al. (1993) provided an integrated review of the nitrate problem within a drainage basin context.

*Salinization*

Apart from industrial loadings (e.g., salt works) and the use of road salts for traffic security during winter snow and/or ice events, salinization is restricted mainly to semi-arid and arid zones with endorheic watersheds. In most cases, it is related to the diversion of water for irrigation, which can result in a catastrophic drop in lake levels and an intolerable salinity increase. The best known example of this phenomenon is the Aral Sea (Fig. 2.9) which, since the 1960s, has experienced a hitherto unknown degree of lake degradation. Other examples comprise medium-sized lakes (e.g., Hamun Lake (Iran and Afghanistan); possibly Niriz Lake (Iran); Ichkeul (Tunisia); see Section 9.4). Increased use of water from Lake Sevan for hydroelectric power generation and irrigation resulted in a water level drop of 18 meters, salinization and dramatic eutrophication between 1935 and 1976. Similarly, the exploitation of water from Mono Lake as a drinking water supply for the city of Los Angeles resulted in significantly increased salinity in the lake. The consequences of salinization most often involve a drastic decrease in biodiversity (e.g., almost all endemic species of the Aral Sea have disappeared), loss of natural resources for humans, which eventually forces the local people to emigrate. Most often, political decisions and a desire for profits have been the driving force for such lake-degrading actions.

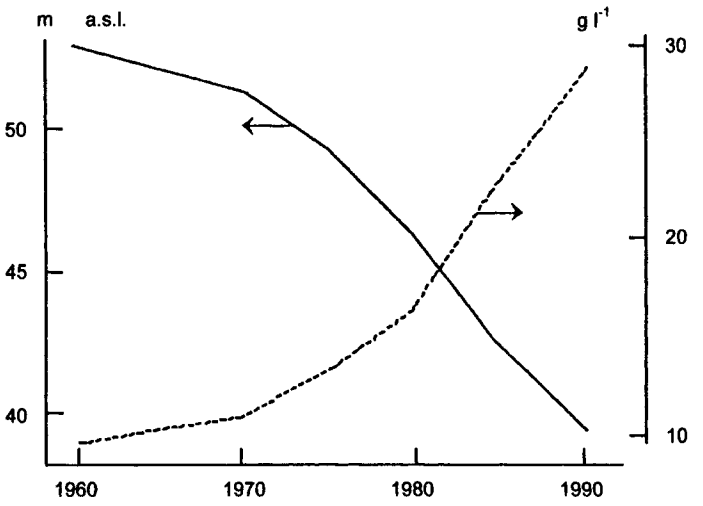



Fig. 2.9. Increases of salinity in the drying Aral Sea: Decreasing water level and increasing chloride contents (from Golubev, 1997).

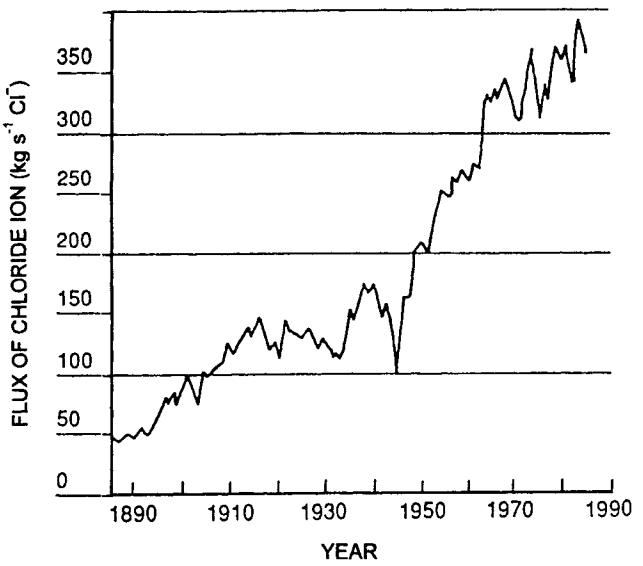



Fig. 2.10. Increase of salinity (expressed as flux of chloride ion) in the River Rhine at the border of Germany and The Netherlands (from Naiman et al., 1995).

Salt concentrations also have recently increased in industrialized areas of the temperate region (Fig. 2.10). In addition to road salting, this is due to the use of mineral fertilizers

which, in addition to the nutritive salts, also contain high chloride and sulphate concentrations, and to the use of high-chloride content detergents.

*Metals*

Two primary groups are conveniently distinguished among the heavy metals. These include (1) those essential for organisms (e.g., iron, chromium) but which, above certain concentrations become toxic, and (2) those not essential for organisms, but which are tolerated at low concentrations and which are toxic above certain concentrations (e.g., cadmium, mercury, lead). In regard to the loading events of lakes and reservoirs, two types again may be considered, including the intentional loading of copper sulphate to lakes to reduce blue–green algal blooms, or to reduce the phosphorus concentration by precipitation with iron or aluminum salts. Although copper treatment is currently outdated for natural lakes and reservoirs, it was a routine method about 40–80 years ago (also see Section 4.3.5—*Algaecide use*). Examples of lakes receiving this potentially degrading treatment include Lakes Mendota, Monoma, Waubesa and Kegonsa in the United States, all of which are linked by one river. As a consequence, these activities have resulted in copper-rich sediment layers, representing an internal loading source for the lakes.

The second type of loading comprises direct adverse additions to lakes and reservoirs by industries and sewage from industrialized cities on the lake shoreline, or loadings from the drainage basin which may release large quantities of heavy metals to rivers and lakes if influenced by acid precipitation. An example of a lake receiving industrial loads is Lago d'Orta in Italy. The lake became heavily polluted in 1926 by a factory producing artificial silk (rayon) via the cupro-ammoniacal method. Only since the early-1980s, when a treatment plant was set up and liming strategies were used to control the pollution, has the lake started to recover from copper pollution and acidification, as demonstrated by the reestablishment of a structurally more-complex biological community. An example of an artificial lake with sufficiently high total copper concentrations to be toxic to algae, is Lake Tjeukemeer in The Netherlands. In this lake, however, the bioavailability and, therefore, the toxicity, is reduced by organic copper complexes (Verweij, 1991).

Water pollution from mining activities has been reported from several sites (also see Section 2.1.5). By far, the largest open-cast mine in China, and one of the largest in the world with an recent annual production over 60,000 tons/day, is the Dexing copper mine in the Jiang Xi province. It has affected China's largest freshwater lake (Lake Poyang) since the early-1960s. Poyang Lake (3350 $km^2$) and its neighboring marshes are ideal sites for white cranes, swans, wild geese and a variety of other species (Hongxiao et al., 1996). Because international investigations have only been undertaken within the past few years, the extent of the mining impacts on the lake has not yet been properly evaluated. The major loading source from the mining area is the nearly Le An River. Primary heavy metals involved in this pollution scenario are copper, zinc and lead.

Man-made lakes constructed for metal strip mining most commonly exhibit a great variety of different chemical features. As such, they can serve as valuable sites for the study of the special impacts imposed by practically all the commercially-extracted metals. Other than for iron and some other commonly-used metals, very little information exists about

lake water quality impacts arising from the mining of less-common ores (e.g., platinum, gold, silver). Strip mining of tin on the Peninsular Malaysia has resulted in the formation of about 4300 bodies of freshwater, ranging in area from between less than one hectare to about 1 $km^2$, and with a total area of approximately 164 $km^2$. These man-made lakes are used as water sources (potable, domestic, irrigation, etc.), for recreation and for conservation (wildlife, native aquatic plants, gene pool for aquatic resources, etc.), and also for waste disposal and dumping (Arumugam, 1994). Thus far, these lakes have been evaluated only on the basis of common physical and chemical features (e.g., transparency, conductivity, pH, phosphate concentrations). The pH values range between 2.7–9.2, with the low values caused mainly by pyritic material. No observations, however, have been made with respect to adverse aquatic impacts by tin. Fifty-five aquatic macrophytes and 42 fish taxons were reported for about 10% of all the strip ponds.

Among the metals present in sufficiently high concentrations in lakes and reservoirs to affect human health, mercury and lead play a more prominent role than others, such as cadmium, chromium, zinc, etc. Organic compounds of mercury and lead enriched in sediment, plankton and macrophytes may contribute to high concentrations in fish and, in this way, also can affect individuals who regularly consume fish. During the 1960s, a number of Swedish lakes were identified as hazardous with regard to mercury, with their use for commercial fishing being prohibited. Few cases of associated diseases were reported. Other than for local gold mining (Brazil, Canada) and industrial activities, the main mercury loading sources are fossil fuels (coal, with about 1 ppm mercury) and fungicides from agricultural activities. The latter was the reason that sea coatings and the use of methymercury as a pesticide were banned in Sweden during the mid-1960s (Hammond, 1971). Since then, many other countries have initiated this action. This general increased awareness of the mercury catastrophes during the 1950s has resulted in a general improvement (i.e., a drastic decrease) in the mercury contents of river and lake sediments. Figure 2.11 illustrates a dramatic decrease in cadmium, consistent with a decline in sludge of a smaller Swedish town. Addressing the long-range transport of metals associated with acid rain requires continental-scale measures. A clear sign is the decline of atmospheric mercury deposition in Sweden after the large chloralkali plants in former East Germany were closed in 1990 (OECD, 1994).

*Mineral oil pollution*

Only a few lakes are presently affected by oil exploitation from sediments. A major example of this phenomenon is Lake Maracaibo in Venezuela, with more than 5000 oil wells. The lake presents a large (about 14,000 $km^2$), shallow (maximum depth of 34 m) and oligohaline (salinity less than 2‰) bay connected to the Gulf of Venezuela (Gessner, 1953, 1955a, 1955b). Lake Maracaibo has been described as heavily eutrophied, with blooms of *Microcystis* (Gessner, 1953). However, information on practically-unavoidable mineral oil pollution is lacking. A similar situation may occur in Lakes Rukwa, Malawi and especially Tanganyika if mineral oil is exploited in these lakes in the near future (Baker, 1989). According to recent reports, Lake Tanganyika sediment (which attains a thickness of 6000 m) may contain more mineral oil resources than the presently-known reserves in the United

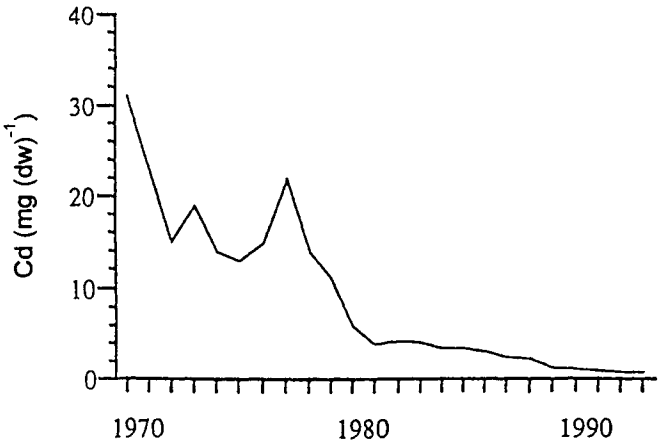



Fig. 2.11. Example of the decreasing heavy metal concentrations in the environment in Europe. Cadmium concentrations in sludge from the purification plant of a small town in Sweden (from Wilson and Jones, 1995).

States. Other problems connected with the risks of oil spills include oil storage tanks on the lake shorelines, pipelines along lakes, mineral oil transfer on lakes, and fuel barges and ships which run on fuel oil (Green and Trett, 1989). During the 1960s, and in spite of protests by prominent organizations such as the German Research Society, a pipeline was established along Lake Constance. Since that time, oil spills related to accidents of fuel barges have become frequent events. Routine boat discharges and pollution from motorboat sporting have resulted in new laws in many countries to help reduce, and even prohibit, motor boat traffic. Nevertheless, there are still many regions where such regulations do not exist. An example of ongoing excessive motorboat sporting is Lake Toba, the largest lake of Indonesia, where fishermen claim that the quality and taste of fish have decreased considerably in parts of this unique caldera lake.

Oil pollution of lakes and reservoirs can result in immediate severe impacts on water quality, since one liter of oil can contaminate one million liters of water. Loss of drinking water resources and fishery are the consequences of oil pollution, and may be irreversible for centuries in large lakes.

*Persistent organic pollutants*

A great variety of organic pollutants (see also mineral oil pollution) exist. Examples include PCBs (Giger, 1992) and 3,4-benzpyrene (a typical compound of tar and tar derivatives, demonstrated as a high risk chemical for Lake Constance by Borneff (1962); also see Section 2.1.9). However, there is no doubt that the most important group of pollutants in the upper layers of sediment in many lakes is pesticides. The first alarm about the serious impacts of pesticides began to surface when DDT was recognized as being harmful to humans and animals. For example, pesticides locally comprise more than 20 ppb. Further, an import of 100 tons DDT was reported for Nepal for 1994 (Shrestha, 1996).

Polychlorinated biphenyls (PCBs) also are recognized as persistent environmental pollutants of high potential danger (Waid, 1986). The most dramatic accidents concerning humans were related to rice oil in Japan ("Yusho" accident) and Taiwan ("Yu-Cheng" accident). PCBs also have frequently been detected in fish (Giger, 1992).

Pesticides affecting lakes and reservoirs arise mainly from:

- Agriculture and forestry,
- Campaigns against aquatic weeds (such as the water fern *Salvinia molesta*, *Eichhornia crassipes*, etc.),
- Campaigns against parasites and waterborne diseases, such as malaria, and schistosomiasis,
- Control of fish populations with rotenone (an active ingredient of derris, known as a fish poison for centuries).

Among the great variety of pesticides used in agriculture, atrazine (used to protect corn from weeds) has been recognized as harmful to humans and, therefore, has recently been banned in several European countries and in Austria since 1995. However, it is still applied in other countries of the European Union. It mainly affects groundwater, but may also be expected to be present in harmful concentrations in lakes and reservoirs within highly-productive agricultural areas. Similarly, bentazone has been identified as a problem, mainly in groundwater. Lindane, an organochlorine insecticide, is poisonous to fish in concentrations as low as one part per million. Moreover, it is a long-lived pesticide.

A rather large variety of pesticides also is applied in forestry activities. In addition to those used in agriculture, they include rodenticides such as zinc phosphide. Only local adverse effects on lakes and reservoirs have been reported for these pesticides.

The application of pesticides to control unwanted (and often introduced) weeds, such as the water fern *Salvinia* spp., water hyacinth *Eichhornia crassipes*, water fern *Azolla* sp. and water lettuce *Pistia stratiotes*, should only be considered if mechanical or biological strategies fail. This is most likely to happen only in very large lakes, or in cases of immediate emergency connected with the blocking of essential harbors or other important sites by weeds. The use of such pesticides should always be considered problematic. Even if their adverse biological consequences are regarded as minor (which generally is unrealistic), the decay of the dead plant material also can influence the water quality.

Since the invasion of Lake Victoria by water hyacinth *Eichhornia crassipes* in 1989, the plant has not only rapidly increased, but also (based on wind-induced currents) choked and blocked areas of important water intakes, as well as becoming a major problem to fisheries. At present, mechanical methods (which imply labor for the poor), biological treatment with South American curculionid beetles (weevils), and chemical strategies are being considered to address this serious problem. The infested area currently (1996) comprises about 4000–6000 ha. Based on first evaluations, it appears that mechanical removal of the weed should be given priority, under the assumption that only the most efficient equipment (e.g., harvesting boats and harvesting from certain shoreline sites) will be made available (also see Section 4.3.4).

Very often, if not always, the use of harvested weeds becomes a problem in countries without relevant tradition. The lack of information on their alternative uses (e.g., as fodder,

fuel, biogas, mulch) does not contribute to the ongoing harvesting activities, and education and training in this field should, therefore, be immediately initiated.

In Lake Kariba (see Section 9.7), the weed problem has a quite different aspect (Ramberg et al., 1987). When finally filled in 1963, the reservoir became affected by a large bloom of the water fern *Salvinia molesta,* which finally covered 22% of the reservoir's surface. Instead of considering this unexpected outburst as the consequence of nutrients from the decaying processes of the flooded terrestrial vegetation and organic matter in general, its management involved control strategies that were both costly and useless. In this case, an assessment of available nutrients within the flooded valley, in relation to the lake's water retention time, would have resulted in much more economic planning. In fact, only two years after the *Salvinia* water fern bloom had its climax, it disappeared because of exhausted nutrient supplies.

*Biological pollution*

Lakes and reservoirs are becoming increasingly affected by biological pollution, comprising four major items:

- Excess growths and spreading of local species,
- Excess growths of exotic species,
- Parasites,
- Waterborne diseases (also see Section 2.2.3—*Health Risks*).

The excessive growth of local species comprises a wide range of Cyanobacteria and algae as well as submerged, floating and emergent macrophytes. Among the latter, only a few species have gained the character of a "weed". An example is the water fern, *Salvinia molesta*, which is the only nuisance species of the genus *Salvinia*. The water hyacinth, *Eichhornia crassipes*, has spread throughout the tropics. Since 1989, it has been a serious problem in Lake Victoria. Both species are of South American origin and belong to item two above (i.e., exotic species). Likewise, the common water lettuce, *Pistia stratiotes*, is most likely a neotropical native. During the late-19th Century, the genus *Elodea* invaded Europe, probably in connection with the introduction of several North American fish species.

Local animal species that have become a nuisance include the European coot, *Fulica atra*, and the mallard, *Anas platyrhynchos*, both of which can affect local water quality. Among exotic birds and mammals, the mute swan in Europe, the Canadian geese in Norway and the musk rat *Ondatra zibethicus* introduced from North America into Europe in 1905 (Czech Republic) can be mentioned. The latter spread throughout Central Europe within decades of their introduction, being introduced deliberately much later in Finland.

Bioinvasions can often result in the extinction of one or more species. Along with eutrophication, new invertebrates may immigrate into a lake and compete with, or even eliminate, other invertebrate species. A large number of intentionally-introduced exotic species in Europe (e.g., the crayfish genera *Pacifastacus*, *Orconectes*, *Cambarus*, the amphipod *Gammarus tigrinus*) have not been reported to have caused the extinction of local species, although they may have influenced the biological community structure. The crayfish (*Pacifastacus leniusculus*) was thought to have replaced the indigenous species,

which almost became extinct due to the introduction of the fungi *Aphanomyces astaci* (introduced first to northern Italy in 1869 from North America and, within three decades, spread throughout Europe). The freshwater crab, *Eriocheir sinensis*, introduced unintentionally from East Asia to Europe, does considerable damage to dams. Most recently, the zebra mussel (*Dreissena polymorpha*), a mussel from the Caspian, Black Sea and Baltic watersheds rapidly expanded throughout Europe during the 1960s, reaching North America during the 1980s. The zebra mussel attaches to all types of solid substrates, becoming widely dispersed by boating and yachting sports. Whenever this species has been detected in a new lake, its population has increased dramatically over several years before it starts to decrease to a level characteristic of its original habitat.

Most recently (Dumont, private communicate), the Caspian Sea was invaded by the comb jelly, a voracious plankton feeder, known already from the Black Sea since the late 1980s. Its presence caused a catastrophic decrease of the mainly endemic zooplankton (*Cladocera*). Fortunately, a melon jelly (*Beroe* sp.) immigrated shortly afterwards which preys mainly on the dangerous comb jelly. Hopefully this event will work to the benefit of the Caspian endemic zooplankton.

With respect to stocking with exotic fish, the two classical examples of catastrophic failures are Lake Victoria and Lake Titicaca. Apart from cichlids introduced to Lake Victoria, which are known to compete with endemic species, the Nile perch (*Lates niloticus*) is a prominent example. During the 1950s, Nile perch "got into the lake". However, from 1987 on, they increased to such an extent that it caused major changes to the biological communities in this largest African lake. One of its major impacts concerns the still unknown degree of extinction of the haplochromine cichlid species. There is no doubt, however, that the Nile perch has since become the most important food and economic resource of the lake. Its present annual harvest amounts to about 500,000 tons. Introduced fish invasions are now dominating in many places of the world, including Europe, with the largest European lake, Lake Balaton in Hungary, being a prominent example (Fig. 2.12). In conclusion, it should be stressed that intended use of exotic species requires not only economic consideration, but also ecological precaution.

Most often, such an evaluation can only be achieved by careful preliminary examination, which should be ordered by law. Such a need is clearly demonstrated by high-alpine lakes where, due to hydrographic features of the watershed, fish are absent. Introduction of char conceded by local authorities (as still going on in the Tyrolian high mountain lakes) can result in the immediate extinction of fairy shrimps or (and) rare daphnids (e.g., *Daphnia middendorffiana*). Therefore, the minimum requirement before any stocking with exotic species is the compilation of an inventory (species diversity at least throughout a period of years) of the relevant lake or reservoir.

### *2.2.3 Risks from Lakes and Reservoirs*

*Health risks*

Between 1870–1900, 450 epidemic events of typhoid fever occurred in Central Europe. Between 1920–1960, the United States and Canada experienced 540 epidemics. In addition to

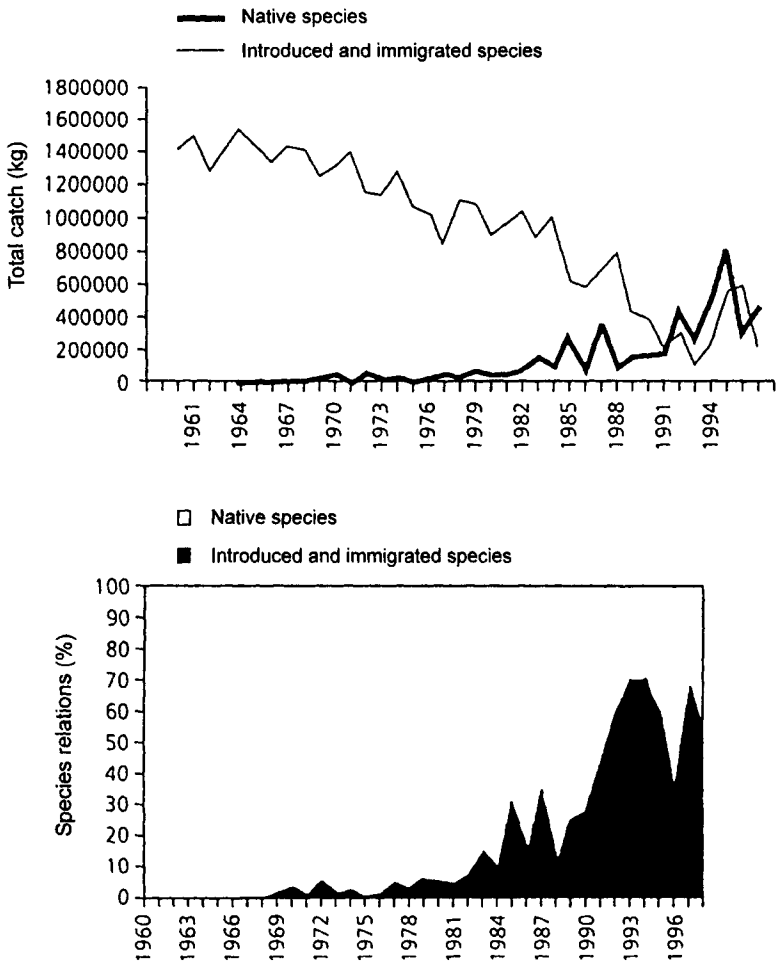



Fig. 2.12. Occurrence of native and introduced species in commercial capture in Lake Balaton, Hungary (from Biró, 1993).

typhoid, paratyphoid fever, dysentery and cholera remained a major catastrophic event during the late-19th Century (e.g., 17,000 causalities in Hamburg in 1892). These epidemics have been greatly reduced in number and extent in industrial countries. In contrast, however, large parts of the developing world and the former Soviet Union are still experiencing not only these waterborne diseases, but also diseases resulting from poor sanitation. Within the topical and subtropical zones, water-based and aquatic insect-related diseases also are major problems (GEMS, 1987).

More than 80% of all the disease in the world is attributed to inadequate water treatment or sanitation facilities. This includes the effects of contaminated drinking water, water

acting as a breeding ground for disease carriers, and diseases caused by the lack of washing water. There are 9.1 million lethal causalities annually caused by waterborne diseases. The 6 million children less than five years old that die each year because of diarrhea (about 100 million people are infected by diarrhea at any given time) demonstrates the serious nature of this continuing situation (UNESCO and UNEP, 1992). Hygienic diseases also are a major problem (e.g., 500 million people suffer from trachoma). This includes diseases resulting from poor sanitation, such as hookworm infections (ancylostomiasis, affecting 800 million people), other parasitic nematodes (1000 million people) and whipworm (trichuriasis, affecting 500 million people), water-based diseases such as schistosomiasis (affecting 60–200 million people), dracunculiasis (affecting about 80 million people), and water insect-related diseases such as malaria (affecting about 160 million people) and yellow fever (onchocerciasis). Unfortunately, there is little hope that the number of infected people from water-related diseases will decrease in the near future, although new research in developed countries is providing guidance for managing these diseases (e.g., Smith and Loyd, 1997, for protozoan parasites).

With respect to lakes, reservoirs and ponds, it should be mentioned that, except for onchocerciasis and hygienic diseases such as trachoma, most water-related infections can be serious problems. This is especially so with dracunculiasis (the small crustacean *Cyclops* is a vector) and schistosomiasis. The crustacean may by swallowed with drinking water from ponds or the littoral zone of lakes (Steib, 1985). Schistomiasis has especially become a problem related to large dams (e.g., Aswan, Kariba, Volta) and is a continuing human risk in large and small tropical lakes and rivers, mainly in Africa.

Waterborne diseases spread by aquatic invertebrates and vertebrate organisms acting as carriers are produced mainly by platyhelminth and nematode worms (Löffler and Malkhazowa, 1990). Estimates of schistosomiasis infections total up to 200 million, a figure reported more than two decades ago and, therefore, not realistic at the present time. This figure implies 200,000 casualties annually. Diseases such as different types of filariasis and dracunculiasis and cestode infections also can be mentioned. In conclusion, it is stressed that, among these listed diseases, aquatic insect related and water-based infections (primarily schistosomiasis) are related to the use of pesticides.

Pollution of medical concern represents a global problem. According to the World Health Organization, about 80% of all diseases are related to inadequate water quality or sanitation, comprising waterborne diseases such as typhoid fever, cholera, dysentery, diarrhoea (gastroenteritis), infectious hepatitis, amoebic meningo-encephalitis, allergic reactions caused by viruses, bacteria, Cyanobacteria, protozoa (*Entamoeba histolytica*, *Naegleria* spp., *Trichomonas* spp., *Giardia* sp., *Cryptosporidium* sp., etc.). About 1.8 billion infected people, and about 3 million casualties (mainly children of less than 5 years age) were reported in 1993 (Brown, 1997). In the coastal waters off the coast of Lima, Peru, a viable form of *Vibrio cholerae* causing "Asiatic cholera" was found associated with several phytoplankton, zooplankton and freshwater macrophyte species (water hyacinths, duckweed) (Epstein, 1993). Similar presence of a viable form of *Vibrio* has not yet been reported from freshwater. However, it is probably possible under favorable conditions, as evidenced by its presence in freshwater macrophytes off the coast of Lima.

Among the aquatic insect-related diseases, malaria has recently increased, restricted mainly to developing nations. Annual estimates for 1993 and 1994 are between 300–500 million, which resulted in an estimated 2 million casualties. In addition, yellow fever and dengue fever, carried primarily by the mosquito *Aedes aegypti*, presently infects about 700,000 people annually. During the mid-1980s, the tiger mosquito (*Aedes albopictus*) from East Asia immigrated into the United States. In addition to being a carrier of encephalitis, it may contribute to the spreading of dengue fever. Onchocerciasis, carried by blackflies, affects an estimated 20 million people in tropical Africa and Latin America. Blackflies also exert heavy impacts on livestock.

Drinking water supplies also are subjected to further risks from pollution by pesticides, mineral oil, allochthonous organic loading, heavy metals, etc. These risks also may be related to lake eutrophication and acidification. Moreover, eutrophication can result in blooms of Cyanobacteria (e.g., *Microcystis*, *Anabaena*, *Aphanizomenon*, *Oscillatoria*, *Nodularia*, *Nostoc*) that are likely to produce neurotoxins and hepatotoxins. Human ingestion can result in illness or even death (see Section 2.2.2—*Eutrophication*). The effects of human ingestion can be gastrointestinal, respiratory and dermatological in nature. The latter are recognizable by initial symptoms which appear after a few hours, comprising erythema and burning pain, followed by the formation of blisters and desquamation which can last for days. Critical toxicity may be caused by hepatotoxins in drinking water, which can affect the liver and even produce liver tumors. It is also stressed that the toxins produced by certain Cyanobacteria may also penetrate into groundwater.

The most common waterborne bacterial pathogens detected in contaminated drinking water supplies in the United States during 1961–1983 were *Shigella*, *Salmonella*, *Campylobacter*, toxigenic *Escherichia coli*, *Vibrio* and *Yersinia* (Craun, 1985). Other waterborne bacterial pathogens were *Mycobacterium*, *Pasteurella*, *Leptospira* and *Legionella*, which also were often detected in swimming waters (Meybeck et al., 1989). The sources of bacterial pathogens for reservoirs are not fully eradicated when human-caused pollution is stopped. Many outbreaks appear to be caused by wildlife living in the drainage basin. Birds and waterfowl can contribute *Salmonella* in their fecal droppings (Anonymous, 1954). Several outbreaks of *Campylobacter* enteritis from drinking water reservoirs had their origin in wildlife, farm animals and animal pets living in the drainage basin. Swimmer illnesses associated with gastrointestinal infections are caused through dispersion of pathogenic agents by fecal wastes discharged in domestic sewage. These pathogenic agents appear to have a better correlation with *Staphylococcus* occurrences in water than with fecal indicators (Seyfried et al., 1985a, 1985b).

Among human viral agents, hepatitis A, Norwalk and *Rotavirus* are reported to have been responsible for 68 outbreaks over 22 years in the United States alone (Lippy and Waltrip, 1984). Other viral agents capable of waterborne transmission include enteroviruses (*Coxsackievirus*, *Echovirus*, *Adenovirus*), *Parovirus* and "gastroenteritis type A". New viral agents capable of waterborne transmission are continuously being found as better methods of detection are developed.

The most significant pathogens debilitating or dangerous to human health from protozoa belong to *Giardia*, *Entamoeba*, *Cryptosporidium* and *Naegleria*. They can produce

dysentery in the human intestinal tracts, dehydration and weight loss. Infection by *Naegleria gruberi* invading the brain and causing meningoencephalitis is almost always fatal. A total of 84 outbreaks of *Giardia* was noted in the United States during 1961–1983, with almost 23,000 people infected. The source of the pathogen was generally found to be the wildlife population, particularly beavers, but also coyotes, muskrats, voles, cattle and animal pests (Meybeck et al., 1989). The density of *Giardia* cysts in reservoirs is often found to be present in the order of magnitude of 1–10 cysts/100 liters of water, thereby being undetectable with common detection methods.

Of the parasitic worms found in heavily-polluted reservoir inflows, *Taenia saginata* (beef tapeworm), *Ascaris lumbricoides* (large intestinal round worm), various species of *Schistosoma* (blood flukes) and *Ancylostoma duodenale* (hookworm) are common.

A recently-detected human health problem is endocrine disruptors, which can cause sexual malfunctions of vertebrate (including human) males. An investigation in the United Kingdom detected estrogenic activity in five rivers by measurement of vitelogenesis in male trout (Harries et al., 1997). Matsui et al. used different chemical methods to detect estrogenic activity in treated sewage effluents in Japan, observing that human estrogens from urine are the major causative substances for their appearance in sewage.

As previously noted by the World Health Organization, about 80% of the diseases in the world are attributable to inadequate water quantity and quality as well as sanitation, including the effects of polluted or contaminated drinking water, water acting as a breeding ground for disease carriers, and diseases caused by inadeguate or no washing.

Apart from lack of water even for drinking, and even less for hygienic activities (see Chapter 1 and this chapter), pollution, such as allochthonous organic loading, mineral oil, heavy metals, and also the effects of eutrophication mainly from Cyanobacteria (producing neuro- and hepatotoxins by certain strains), as described in this chapter is a great risk to stored water.

Waterborne diseases, such as infectious hepatitis, typhoid fever, cholera, dysentery, diarrhoea caused by viruses, bacteria and protozoa (genera *Cryptosporidium*, *Entamoeba*, *Giardia*, *Naegleria*, *Trichomonas*, etc.) contribute greatly (almost two billion infected and annually 3 million casualties have been reported by Brown) to the present worldwide poor health conditions. About 400–500 million people suffer from trachoma (*Chlamydia trachomatis*), about 2 billion are infected by different nematodes (*Ascaris*, *Ancylostoma*, *Trichuris*, etc.) not necessarily related to water themselves. Many other examples of parasites related to poor hygienic could be mentioned.

In addition water-based diseases are a permanent threat to the health of more than 250 to 300 million people in mainly tropical and subtropical regions. These are caused mainly by trematode (schistosomiasis and other diseases such as fasciolasis) and nematode worms (dracunculiasis) and other infections such as angiostrongylosis). The most widespread among these diseases is presented by schistosomiasis or bilharzia (intestinal and urinary). The first larval stage is highly specialized on certain snails, which also are carriers of schistosomiasis of domestic stock, wild mammals and birds. Seventy–Eighty percent of the afflicted persons are African residents. In contrast to *Schistosomiasis*, *Dracunculiasis*

(*Dracunculus medinensis*) is confined to Africa, India and Southwest Asia. The larvae of this nematode are carried by cyclopoid copepodes and swallowed with infected water.

As previously noted, among the aquatic insect related diseases, malaria (spread by the *Anopheles* mosquito) is the most important one, annually affecting 150–200 million people and resulting in more than a million casualties. Yellow and dengue fever carried by the mosquito *Aedes aegypti* also infect approximately 700,000 people each year. Many mosquitos are also vectors of arboviruses (species of *Aedes, Anopheles, Culex, Mansonia*), causing different kinds of diseases, several of which are locally restricted, such as different types of encephalitis, epidemic polyarthritis, etc. In additional to malaria, *Anopheles* species also can carry larval stages of the nematode *Brugia* (*Wocheria*), which may be disseminated by the mosquitoes of the genera *Aedes* and *Mansonia* too.

Another important midge family with respect to diseases is the blackflies (*Simuliidae*), which transmit Onchocerciasis ("Driver blindness") in Africa and Central America, being caused by the nematode *Onchocerca volvulus* (Africa) and *O. caecutiens* (Central America). *Onchocerciasis* affects human skin, eyes and the lymphatic system. The disease is particularly widespread in western and central Africa, where 80–100% of the adult population is infected. Two other Simulium species that are pests to man and livestock are *S. chufferi* and *S. bovis* in central and southern Africa. In the Senar and Roseires Reservoirs in Sudan, *S. damnusum* has spread. Blackflies viciously attack humans and livestock, with severe loss of the latter being reported in various parts of the world. In Rumania, for example, more than 17,000 animals were killed during one season (Rietschel, 1975).

*Catastrophic events from reservoirs and lakes*

*Catastrophic degassing.* Meromictic lakes (recognized by a persistent vertical gradient; see Chapter 1) and volcanic lakes may accumulate considerable amounts of gases ($CO_2$, $CH_4$, $H_2S$) in their monimolimnium or hypolimnium. Among the meromictic lakes, Lake Kivu in East Africa (2370 $km^2$, $z_{max} = 480$ m) is well known for its high methane content which is exploited commercially. The largest meromictic body of water with high, even critical, concentrations of gases, especially hydrogen sulfide ($H_2S$) is represented by the Black Sea, whereas the largest meromictic lake (Lake Tanganyika) is characterized by only a moderate $CO_2$ concentration. More recently, however, sudden degassing from two rather small volcanic lakes in Cameroon resulted in human casualties and loss of animal stock. Both lakes are rather deep (96–208 m) and occupy small explosion craters located within a volcano-tectonic feature that extends about 1400 km from central Africa into the South Atlantic basin.

The first $CO_2$ burst from Lake Monoun in August 1984, although causing 37 human casualties, received only little attention (Sigurdsson, 1987) and was thought to have been released by a landslide. In contrast, a much more catastrophic event occurred in Lake Nyos in August 1986, resulting in 1746 casualties and a loss of more than 8000 livestock, and receiving immediate worldwide attention, was the impetus of an international conference organized by UNESCO in early 1987. The very speculative explanations given (volcanic explosion, etc.) were finally replaced by realistic interpretations of both cases, recognized for the first time by scientists (Sigurdsson, 1987). The gradual influx of volcanic $CO_2$

in the bottom region of the relevant lakes obviously increases the gas concentration in the hypolimnic water continuously to an extent controlled by the pressure of the upper stratified epilimnetic water-mass. Cooling or (and) cold precipitation, which happen during August and September in this part of Cameroon, most likely resulted in a collapse of the high density stratification, allowing a sudden degassing of the lake. As recognized by eroded areas of elevated shore sections, the burst of Lake Nyos caused a giant water fountain more than 80 m in height. The gas cloud then flowed down the Nyos Valley, about 190 m lower than the lake, affecting more than 60 $km^2$. Among preventive measures discussed, the Olszewski syphon tube technique seems to be most appropriate, since it will provide for the continuous draining of bottom water subjected to continuous $CO_2$ influx (see Chapter 1).

*Impacts by hydrologic alterations*

In the valley of Bagnes (Valais, Switzerland), a lake (200 m long; 60 m deep) became dammed behind an avalanche in 1818. Efforts to drain the lake gradually failed, resulting in a disastrous flood (Hutchinson, 1957). In Tyrol, the headwaters of the Ötztal were often blocked by ice barriers during periods of glacier advance (1599–1601, 1677–1682, 1770–1774 and 1845–1848). In association with a sudden increase in temperature and heavy rainfall, a dam formed by the Vernagt Glacier in 1678 and 1680 burst, resulting in the worst disaster observed in the area. Whole villages were destroyed to such an extent they could not be rebuilt (Patzelt, 1994).

The eruption of lakes held behind the terminal moraine of a glacier has been reported for several areas in the Alps, and more especially in the Andes (Löffler, 1988). The most disastrous event happened in 1970 in Peru, when the city of Yungay was completely destroyed, with 18,000 people losing their lives. In this case, however a rock- and glacier-fall from Mt. Huascarán, released by an earthquake, was involved. Part of it fell between two morainic mountain lakes, which contributed to the final dimension of the catastrophe. More recently, Watanabe et al. (1994) reported the prospect for a catastrophic flood in Khumbu Himal, Himalayas.

Catastrophic floods in the Mississippi Basin in 1993, along the Rhine in 1995 and most recently (August 2002) in northern Austria, Bohemia and eastern Germany (along the Elbe River) associated with casualties and tremendous economic losses, have clearly demonstrated that even rich industrial countries, in spite of their relevant knowledge, are not willing to invest in the provision (most often for the return) of adequate retention areas, and for dams and dykes capable of resisting unusual by large floods. Moreover, dam systems along rivers were not prepared (regular removal of sediment mud) for floods of this magnitude, therefore contributing greatly to mud problems during the inundation (see also Section 2.3).

## 2.3 BAD MANAGEMENT

Lack of ecological understanding and inadequate knowledge has contributed greatly to the mismanagement of lakes and reservoirs and their drainage basins around the world. A list of selected cases of lake mismanagement is given in Table 2.5.

Table 2.5. Selected cases of lake mismanagement

Drainage basin:
- Regulation of rivers
- Destruction of wetlands
- Uncontrolled deforestation or adverse afforestation
- Adverse agricultural activities resulting in erosion and pollution with nutrients and pesticides
- Excessive livestock
- The use of inadequate sewage plants for polluting settlements and industries
- Polluting mining activities
- Traffic in sensitive parts of the drainage basin

Lake/reservoir shores:
- Destruction and reclamation for industry, traffic, agriculture and recreation (tourist centers, sport boating, yachting harbors, etc.)
- Destruction (including drainage) of littoral wetlands
- Improper introduction of macrophyte-feeding fish (e.g., grass carp) to lakes
- Chemical treatment of macrophytes
- Rooting of submerged vegetation
- Heavy eutrophication as an impact on macrophytes

Lake/reservoir:
- Point sources of pollution without adequate treatment
- Neglect of nonpoint pollutant sources
- Erradication of algal blooms by pesticides
- Flushing of meromictic lakes, which may result in a massive release of nutrients from the monimolimnion
- Aeration at the wrong time and/or wrong depth
- Stocking with fish species with adverse influences, including increased internal loading, removal of large zooplankton, and general decrease of biodiversity
- Excess use of sport boating, yachting, etc.

Reservoir:
- Point sources of pollution
- Stocking with exotic species, unless careful investigations on possible harm are made prior to introduction

Since the 1950s and 1960s, rapidly-increasing limnological knowledge has significantly aided our ability to provide solutions for sustainable management of lakes and reservoirs, and to address most of the major problems regarding inland waters, provided that public awareness exists, that decision makers are willing to act, and that the economic requirements are not prohibitive. That the mismanagement of lakes and reservoirs still occurs in many regions of the world is due mainly to such factors as:
- Rapidly growing population in the drainage basin,
- Rapidly growing tourism and recreational activities along the shorelines,
- Political pressures and quests for profits (which are often related),
- Poor financial capacity,

- Lack of education and training, resulting in a lack of awareness of the consequences of serious problems, and
- Complex, multinational problems requiring regional cooperation (pollution in multinational drainage basins, acidification, climatic change, etc.).

The most obvious failures, which comprise all of the items mentioned above, are the lack of a holistic perspective of the problem and its solutions, and of long-term planning.

Lakes and reservoirs in developing countries are the most-cited examples of the lack of a holistic perspective. Lake Victoria serves as a prominent example. The rapidly growing population in its drainage basin is evidenced by the capital of Uganda. In the early-1960s, it had a population of only about 90,000 inhabitants, which increased to nearly one million people by 1996. The consequences of this magnitude of population increase (e.g., pollution, deforestation, reclamation of shoreline areas and wetlands within the drainage basin, traffic and rapidly growing agriculture) have resulted in significant eutrophication problems, particularly in the embayments of the lake. Moreover, the introduction of the Nile perch in the 1950s, and the immigration of the water hyacinth in 1989, have caused additional serious problems.

Lack of long-term planning, most prominent in industrial countries during the 1960s and 1970s, has recently received attention and, therefore, this problem seems to have be mitigated to some degree. However, tourist pressure is still increasing in the Lake Tahoe drainage basin in the United States. It also is a major problem in the developing world, with Lake Atitlan in Guatemala, and Dal Lake in India being good examples (Gopal, 1996). In addition to the increasing population (lacking sewerage), industrialization and overgrazing, tourism development also is a significant causative factor in this problem.

Political pressure is clearly recognized as a significant causative factor in the case of the Aral Sea. An international meeting in 1980 included an address by a limnologist of the former Soviet Union, praising the irrigation activities in the Aral Sea drainage basin. However, other participants raised their voices in earnest warnings. Their warnings were, at that time, met with rather harsh criticism from some Soviet participants. Within a relatively few years, however, the magnitude of the Aral Sea catastrophe became obvious to everyone (Glantz, 1998).

Inadequate financial capacity is probably the most important hindrance to sound management of lakes and reservoirs in many countries today, with developing countries forced to give priority to the survival and well-being of its inhabitants. Donor organizations and other agencies, such as the World Bank, United Nations Development Programme (UNDP) and the United Nations Environment Programme (UNEP) continue to try to improve or mitigate ecological conditions, even after they have become catastrophic in nature. An example of this "fire brigade" type of activity is the recent Lake Victoria Environment Management Project. The project is estimated to be able to stabilize, or slightly improve, the present serious problems of this multinational lake involving the five countries of its drainage basin. A total of US $300–600 million is proposed to address the deteriorated fishery, whereas a sum of only US $3.5 million has been provided for the stabilization of its present water quality. The latter sum is less than the amount deemed necessary to attempt to address the water hyacinth problem (US $6–10 million).

Lack of public (and decision maker) awareness of serious problems is a consequence of inadequate or nonexistent education and training. Until the 1960s, limnological knowledge on the part of the public and decision makers was generally poor. As a result, early warnings from limnologists regarding pollution, and the consequences of inadequate lake management, did not find any response. Even less concern was given to those aspects of drainage basin management related to lake and reservoir water quantity and quality. In fact, increasing the awareness of the importance of considering the entire drainage basin of a lake or reservoir remains an ongoing process around the world even today, in both developed and developing countries. In many mountainous areas, for example, a clear-cutting forest strategy is still followed, sometimes contributing to a large nutrient load to waterbodies. Other adverse ongoing activities, such as tourism and holiday village construction on lake shorelines, an example being Lake Tahoe (Goldman et al., 1970; Goldman, 1990, 1993), also must be considered in regard to economic pressures.

The most sensitive, and most frequently affected, habitat of lakes and reservoirs is their littoral zone, which is most often subjected to damage and degradation. At the same time, the littoral zone (with its micro- and macrozonation) represents the most sensitive zone in a lake or reservoir and can react to even moderate eutrophication pressures and other changes within the drainage basin. Apart from tourism pressures, and less so for agriculture and hydrological changes, activities directed to increasing awareness on the part of the public of its importance for the entire ecosystem, with regard to water quality and biodiversity, has only begun in earnest during the last two decades, and still requires considerable additional effort.

Thousands of examples of the lack of awareness and/or inadequate management of the littoral zones of lakes around the world exist. Zürichsee (Switzerland; Thomas, 1972; Schanz, 1993) is a typical case among the pre-alpine lakes in Eastern Europe. Due mainly to land reclamation (e.g., for parking space), a major area of its Phragmites stands disappeared by 1931, when the first census by airplane was conducted. A further reduction of about 70% of the remaining stands took place during the 1950s and 1960s, when eutrophication of the lake contributed greatly to the vanishing of reed stands (Thomas, 1960). Only after 1970, when the nutrient loading of Zürichsee became successfully controlled, coupled with a rapidly growing awareness of the importance of the *Phragmites* remnants, did efforts to facilitate their preservation occur. The total reed area totals 5.14 ha at present, which is probably less than 10% of that existing prior to 1931.

The littoral zones of the artificial Nette lakes (peat mining and mill-ponds, established between the 16th and 19th Century) in northwest Germany were once important breeding sites for birds, such as the little bittern (*Ixobrychus minutus*), the marsh harrier (*Circus aeruginosus*), the bluethroat (*Cyanosilvia suecica*), etc. They subsequently were largely destroyed as a result of using reed stands as dumping sites, by road construction activities, and by other more complex impacts on the macrophytic vegetation (especially reeds), and by increasing eutrophication. Unfortunately, sanitation and restoration measures applied thus far to the lakes have met with little success.

The loss of littoral habitats, demonstrated by the above examples, belongs among worldwide lake impacts, which often do not contain any attempts at restoration, and which often

results in a complex combination of lack of awareness, mismanagement, profit motivation at the expense of environmental protection and conservation, inadequate financial capacity, and rapidly growing populations, especially in cities. Chicago, Detroit, Buffalo, Toronto and other cities on the shores of the Great Lakes in North America, and near Kampala, Uganda near Lake Victoria, serve as examples for large lakes. Few attempts have been made so far to conserve or revitalize the still-existing littoral zones, similar to that mentioned for Lake Zürich.

Mismanagement is even more apparent in regard to the thousands of reservoirs, artificial ponds and lakes created mainly for recreational purposes. Recommendations and standards so far proposed (e.g., in Austria) to ensure desirable water quality and self purification have been largely ignored. These recommendations and standards comprise:

- Topographical requirements (e.g., construction only outside of risk zones),
- Protection of at least 30% of their area for self purification,
- A 50–100 m protected littoral zone,
- Avoidance of conflicting activities (e.g., fishing, which implies feeding of fish), and
- The proper management of submerged vegetation, and avoidance of adverse control methods (stocking with grass carp, etc.).

In Central Europe (and elsewhere), it is likely that less than 1% of recreation ponds currently meet these standards.

An even greater lack of limnological understanding can be observed in regard to ecosystem-drainage basin management (see Chapter 7). The ecosystem-drainage basin concept appears to have originated with the Hubbard Brook Watershed–Ecosystem, described in a series of publications by Bormann and Likens and others (Likens et al., 1977; Bormann and Likens, 1979), and even earlier by Mackereth (1966) as stressed by O'Sullivan (1979).

As an important outcome of this concept is the construction of models (see Chapter 5), in which the basic indivisibility of ecological and hydrological processes in any ecosystem-watershed, and the particular role of water in linking the various parts of the system, have been stressed (e.g., O'Sullivan, 1979).

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*Chapter 3*

# EVALUATING LAKE AND RESERVOIR WATER QUALITY

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

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

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

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

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

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

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

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

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

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

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

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

## 3.1 HUMAN PERCEPTIONS OF WATER QUALITY

The general public perceptions of lakes and reservoirs was discussed in Section 1.2.1. This chapter discusses perceptions of water quality in more detail. Humans perceive lake

and reservoir quality on the basis of its intended uses. The perceptions will differ, depending on whether or not an individual is swimming in a lake, boating or fishing in it, or if a lake is being used as a drinking water supply. The perceptions can arise solely on the basis of human senses or, alternatively, from specific technical determinations.

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

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

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

## 3.2 INDICATORS AND CLASSIFICATION OF WATER QUALITY

The indicated geographic differences of lakes discussed in Section 1.3.2 relate to both unpolluted and polluted lakes. However, the distinctions above do not specifically relate to pollution. This section deals with water quality indicators and the corresponding water quality classifications. The primary problem is that natural conditions differ in different regions, and accurate water quality classification must consider these differences. This is why different water quality classification systems have been developed in different countries. In some instances, they differ only in the value ranges of the same criteria used to

distinguish different classes. A review of the ecological classification of surface waters, with an emphasis on lakes, is given by Premazzi and Chiaudani (1992). Because it is not possible to enumerate all of them, this chapter will highlight representative examples.

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

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

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

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

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

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

### *3.2.1 Water Quality Consequences of Lake Morphometry*

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

Common sense dictates establishing an absolute depth limit for waterbodies to be considered shallow. From the perspective of water quality, however, the absolute depth is not

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

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

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

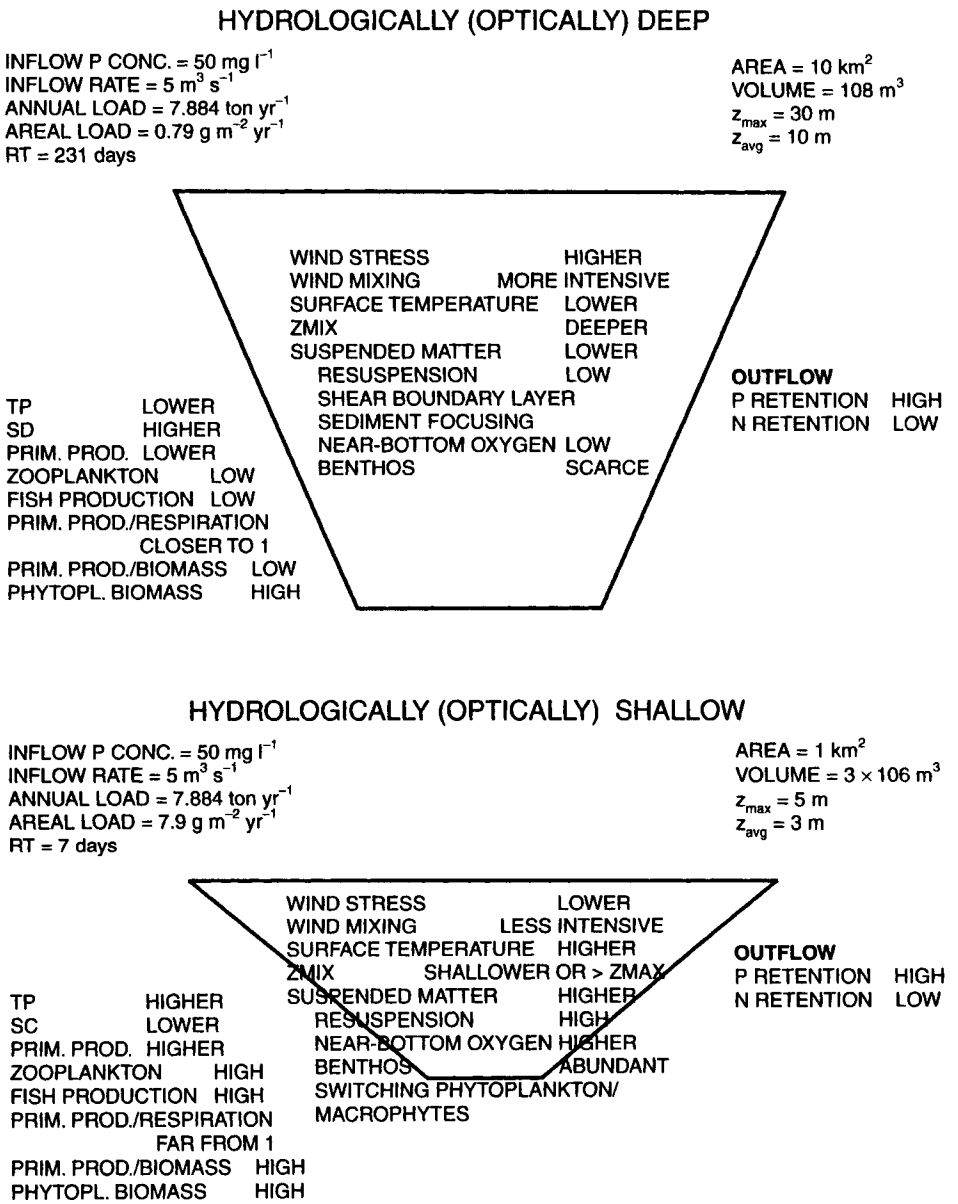

HYDROLOGICALLY (OPTICALLY) DEEP
INFLOW P CONC. = 50 mg l$^{-1}$
INFLOW RATE = 5 m$^3$ s$^{-1}$
ANNUAL LOAD = 7.884 ton yr$^{-1}$
AREAL LOAD = 0.79 g m$^{-2}$ yr$^{-1}$
RT = 231 days
AREA = 10 km$^2$
VOLUME = 108 m$^3$
$z_{max}$ = 30 m
$z_{avg}$ = 10 m
WIND STRESS HIGHER
WIND MIXING MORE INTENSIVE
SURFACE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

Fig. 3.1.

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

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

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

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

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

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

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

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

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

### *3.2.2 Chemical Water Quality Classification of Lakes*

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

*Organic matter*

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

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

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

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

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

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

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

*Acidity*

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

The natural origin of acid waters is either the direct dissolution of acidic rocks or the release of fulvic acids and related humic compounds from decomposing vegetation. Humic compounds affect a number of aquatic processes, including the binding of nutrients and the

formation of chemical complexes. They also decrease sunlight penetration into the water column, directly affecting invertebrates and fish distribution (Hessen and Travnik, 1998).

*Hardness*

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

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

### *3.2.3 Indicators of Lake Trophy and Trophic Classification*

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

More recent indicators of lake trophic state are based on work initiated by Vollenweider (e.g., Vollenweider and Kerekes, 1982), which reflects the situation outlined in Section 2.2.2—*Eutrophication*, and Figure 2.4, which remains a major threat to lake water quality and trophic condition. Because of the combined effects of organic pollution, however, the situation in developing countries with rapid population growth and industrialization is more complicated (see also Section 2.2.2—*Eutrophication*). Trophic state indices are generally based on a few variables, representing both the causes and responses of trophic conditions. The primary cause of increasing trophy is the in-lake concentration

of the critical limiting nutrient, which most often is phosphorus. Total phosphorus was selected as the typical measure of phosphorus concentrations in lakes because of its relatively easy measurement. Reactive (inorganic) phosphorus, which is more readily accessible to uptake by algae, is more difficult to measure accurately because of its rapid incorporation and release by aquatic organisms. The consequence of increased critical nutrient loads to a lake, and the resultant increased in-lake concentrations, is the increased production of organic matter within the lake, mainly in the form of algae.

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

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

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

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

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

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

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

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

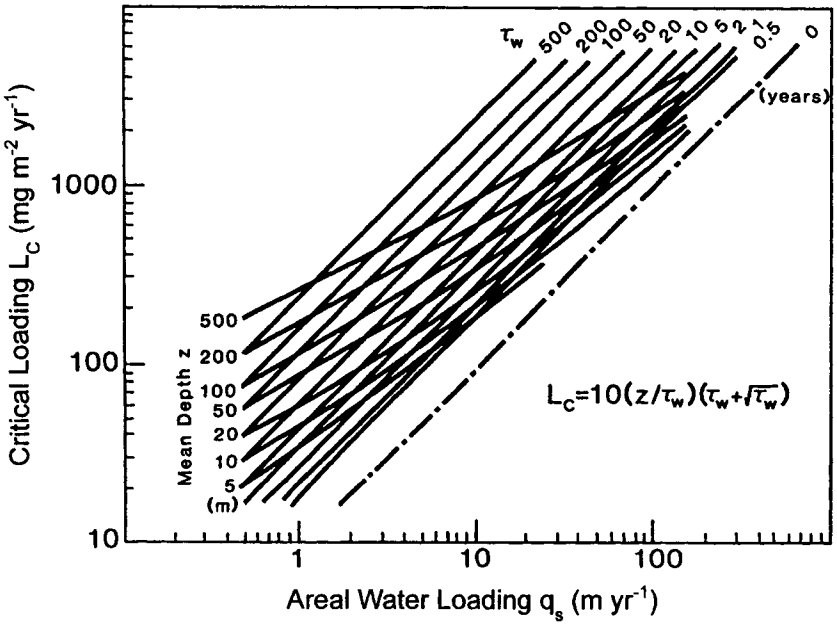



Fig. 3.2. Vollenweider diagram for critical phosphorus loading of lakes (to remain in an oligotrophic state), based on the areal water loading, mean depth and theoretical retention time ($\tau_w$, expressed in years). The formula on which the graph is based is shown (Vollenweider, 1975).

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

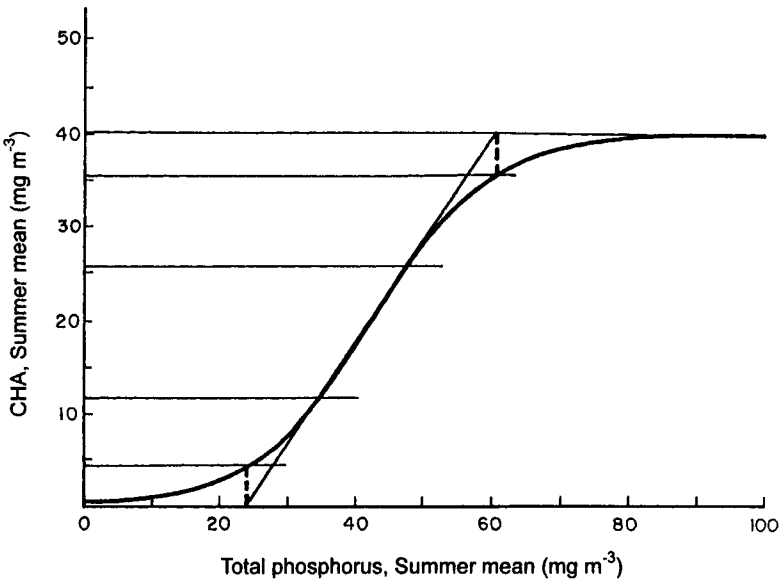



Fig. 3.3. Schematic representation of the average relationship of the summer mean chlorophyll-a (CHA) concentration to the summer mean total phosphorus concentration in phosphorus-limited, transparent, temperate, dimictic lakes. The lines indicate: 1—the critical point of chlorophyll rise; 2—the onset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

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

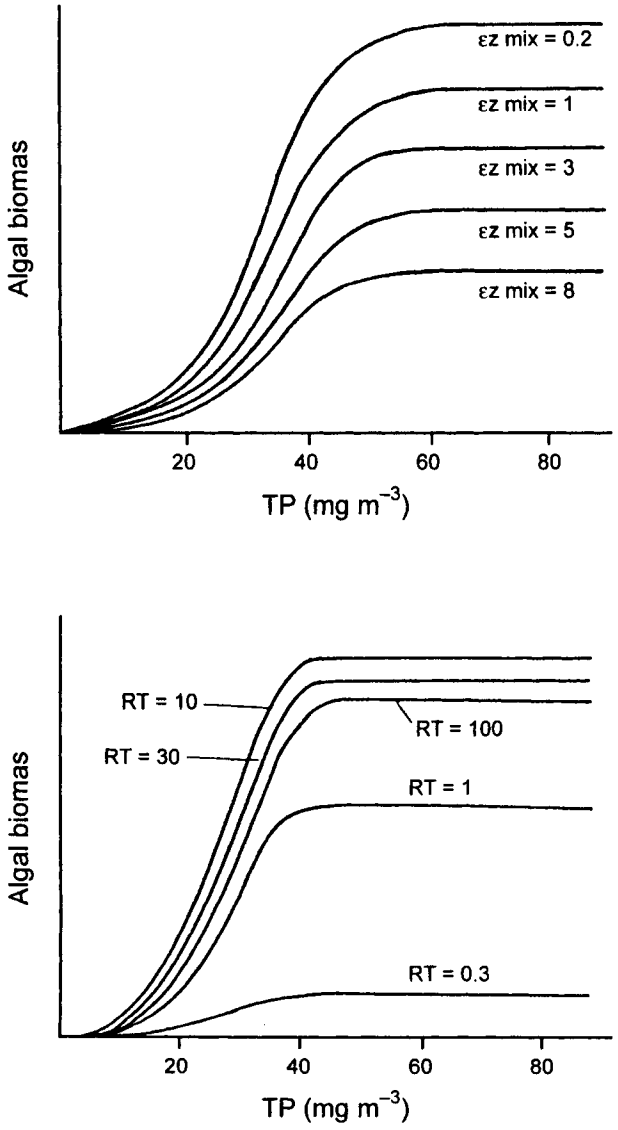



Fig. 3.4. The chlorophyll-a concentration and primary production in lakes and reservoirs depends not only on the critical nutrient (in this case phosphorus, *TP*), but simultaneously on other variables as well. Schematically, the effect of the optical depth (multiple of the extinction coefficient and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

Because algae produce biotic turbidity and the natural turbidity of lakes is generally low, a very simple measure of the trophic state of the lake is often made on the basis of its clarity,

determined by a Secchi disc. The Secchi disc depth, or transparency, is determined as the depth at which the contours of a white–black disc cease to be distinguishable. Quantitative relations are given in Section 3.4.2—*Evaluation of Water Transparency*.

*Morphoedaphic index (MEI)*

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

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

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

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

*Carlson trophic state index (TSI)*

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

three variables has some drawbacks in uniquely defining lake trophic state, the advantage of this approach is the averaging of these variables with the regression character of the relations. Averaging the values also helps reduce the effects of random data errors in each type of measurement. The relations between the TSI and these three variables are as follows:

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

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

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

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

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

An interesting evaluation using the TSI methodology was done by Havens (1994) for a shallow subtropical lake Okkechobee in Florida. He followed the seasonal changes of deviations of $TSI$(CHA—TP), the difference between $TSI$(CHA) and $TSI$(TP), and $TSI$(CHA—SD), finding that deviations among trophic state index variables indicated both

the degree of nutrient limitation and the composition of seston in the lake. The degree of nutrient limitation was positively correlated with the TSI deviation (CHA–NUTRIENT), negative values indicating limitation by the nutrient under consideration. Seston particle size is positively correlated with the TSI deviation (CHA–SD), with positive deviations indicating presence of large algal particles which attenuate less light than the same mass of small particles, and vice versa for negative deviations (Fig. 3.5).

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

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

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

*Areal hypolimnetic oxygen deficit (AHOD)*

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

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

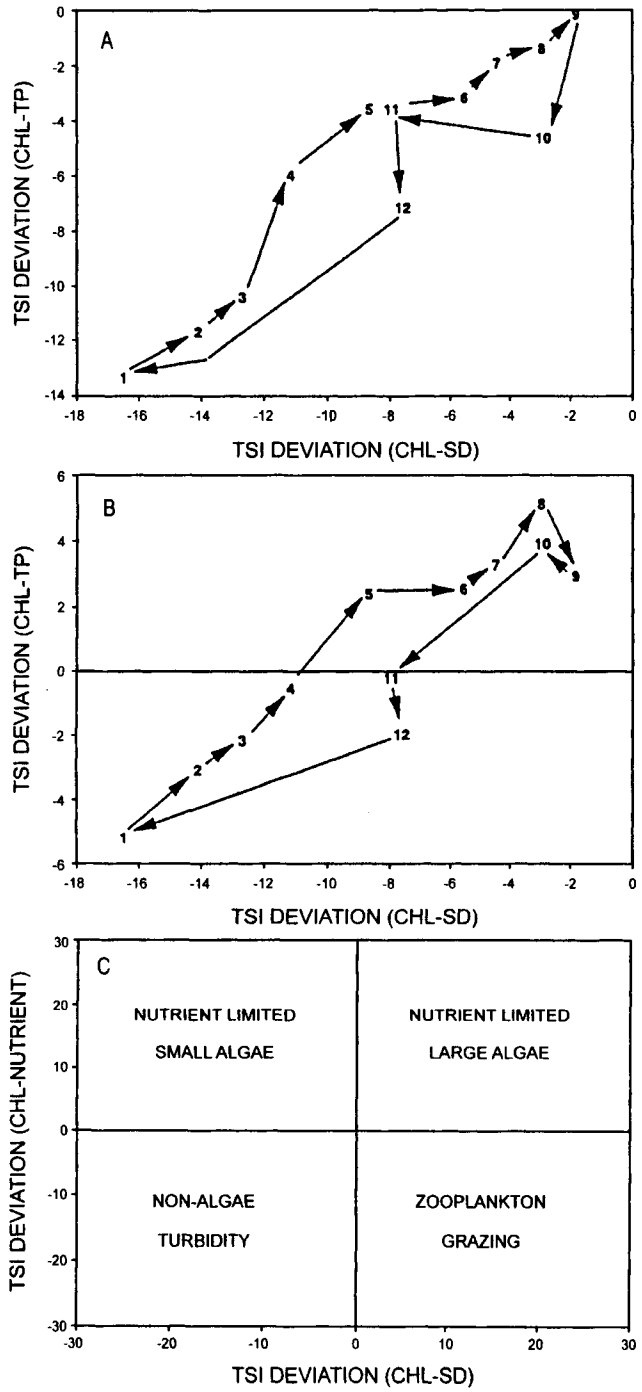

A
TSI DEVIATION (CHL-TP)
TSI DEVIATION (CHL-SD)
B
TSI DEVIATION (CHL-TP)
TSI DEVIATION (CHL-SD)
C
NUTRIENT LIMITED
SMALL ALGAE
NUTRIENT LIMITED
LARGE ALGAE
NON-ALGAE
TURBIDITY
ZOOPLANKTON
GRAZING
TSI DEVIATION (CHL-NUTRIENT)
TSI DEVIATION (CHL-SD)

Fig. 3.5.

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

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

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

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

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

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

*Comparison of trophic state indices*

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

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

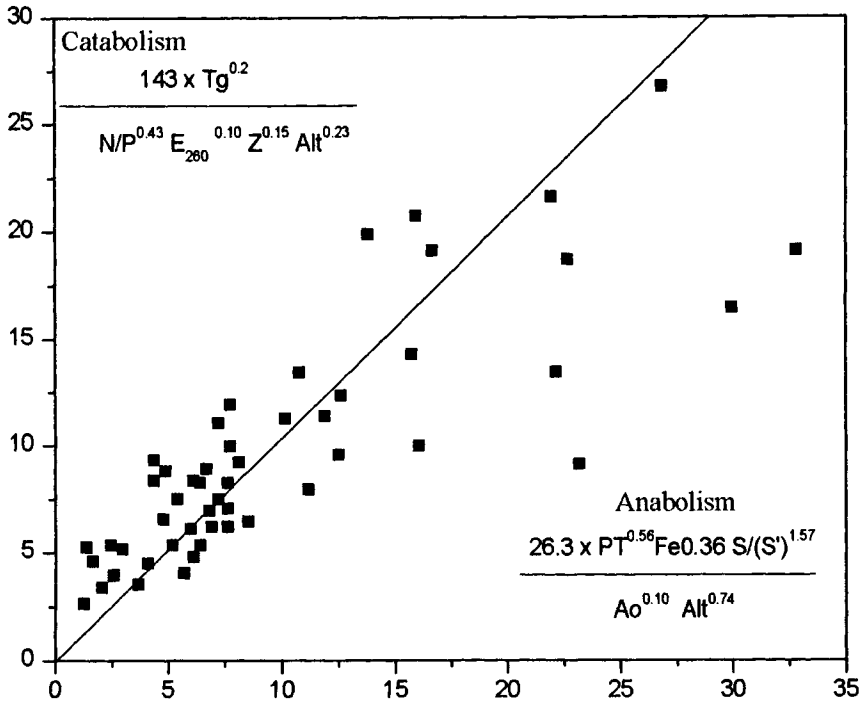



Fig. 3.6. The position of a number of German and Austrian lakes on the axes represented by anabolism and catabolism. The definitions of anabolism and catabolism are indicated on the figure (Schröder and Schröder, 1978).

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

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

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

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

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

[1] Danger of clogging of the irrigation spray jets.

[2] Impairment to turbines due to clogging by macrovegetation, to the mechanics of turbines by aggressive bottom waters.

[3] Within the concept of landscaping, the results of a natural eutrophic aging process can be desirable.

### *3.2.4 Ecological Indicators*

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

*Biomass and related units*

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

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

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

*Ascendancy*

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

*Buffer capacity*

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

### *3.2.5 Organism Species as Biological Indicators*

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

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

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

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

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

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

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

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

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

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

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

### 3.2.6 *Ecotoxicological Indicators*

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

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

for detecting the impacts of xenobiotics on aquatic life are as follows:

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

*Critical evaluation of different approaches to aquatic toxicity*

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

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

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

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

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

### 3.2.7 *Complex Water Quality Classification Systems*

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

Due to the large number of water quality variables and their high variation, a water quality index may help to describe the overall situation (Thanh and Biswas, 1990). However,

these authors also pointed out that the number of water quality indices used in different countries and in states of the United States are as high as the number of variables themselves. Several index and classification systems exist for lakes. They are very local, based on the specific conditions in a given country or region, and no generally accepted lake classification system exists.

*Lake health*

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

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

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

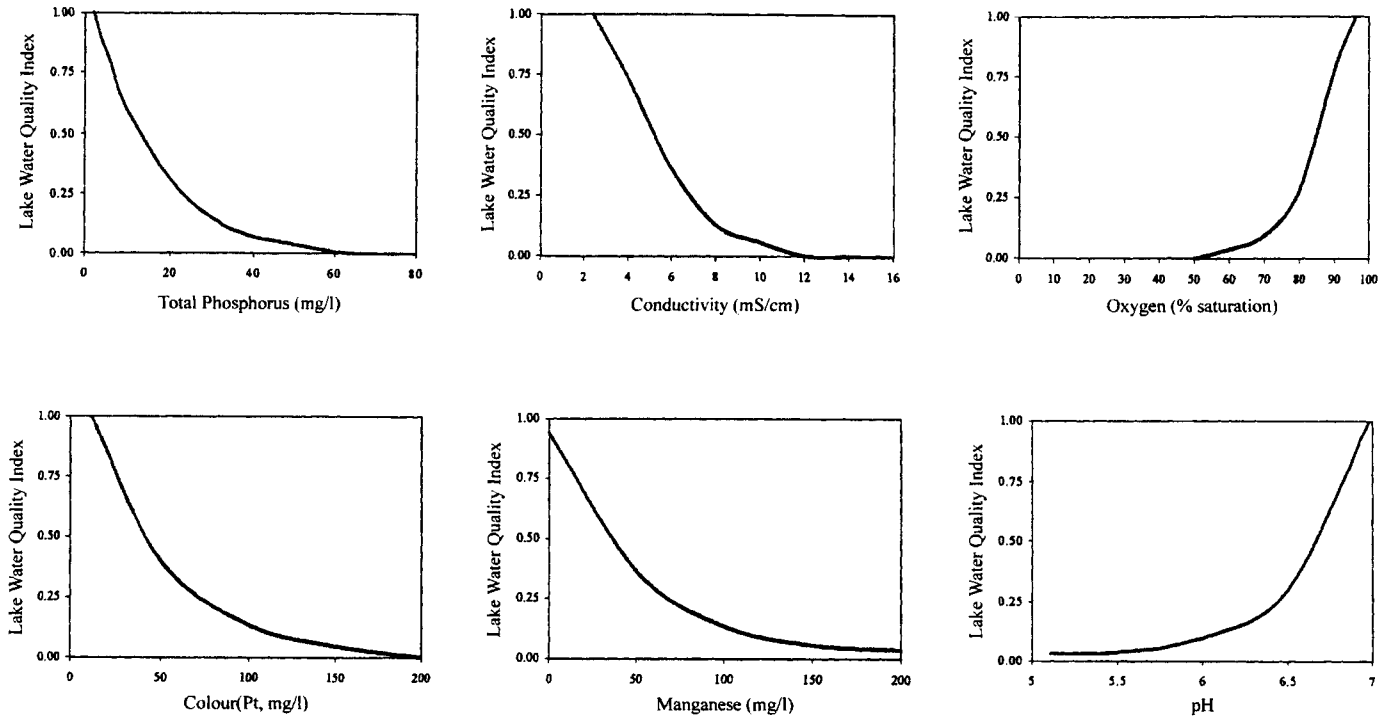



Fig. 3.7. Value functions for the General Lake Water Quality Index for Finnish deep-water lakes. The functions normalize the effects of each variable so that a value between 0 and 1 is achieved, with 1 for the best and 0 for the worst water quality. The effect is positive for some variables and negative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

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

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

## 3.3 LAKE SAMPLING GUIDELINES

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

### *3.3.1 Selection of Variables*

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

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

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

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

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

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

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

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

### 3.3.2 Selection of Sampling Stations, Depths and Timing

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

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

*Sampling spots and depths of sampling*

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

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

*Sampling frequency*

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

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

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

### *3.3.3 Automatic Monitoring*

Two types of automatic sampling can be distinguished:

- Depth recording, in which measurements are made during the lowering of a probe through different depths,
- Recording over time.

In some instances, these two approaches can be combined.

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

The sensitivity of modern electronic instruments is usually high. However, their accuracy is highly dependent on the accuracy of their calibration. Automatic instruments used without careful calibration can easily give erroneous results. Although many recently-

developed instruments have built-in procedures for their automatic calibration, the need for a periodic check of their calibration, using classical chemical or other determination methods, is still stressed. Calibration for extreme low and high water quality values results in the highest accuracy.

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

Depth profiling instruments can be classified into two types:

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

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

### *3.3.4 Remote Sensing*

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

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

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

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

## 3.4 HOW TO EVALUATE WATER QUALITY MONITORING DATA

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

### *3.4.1 Data Storage and Handling*

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

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

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

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

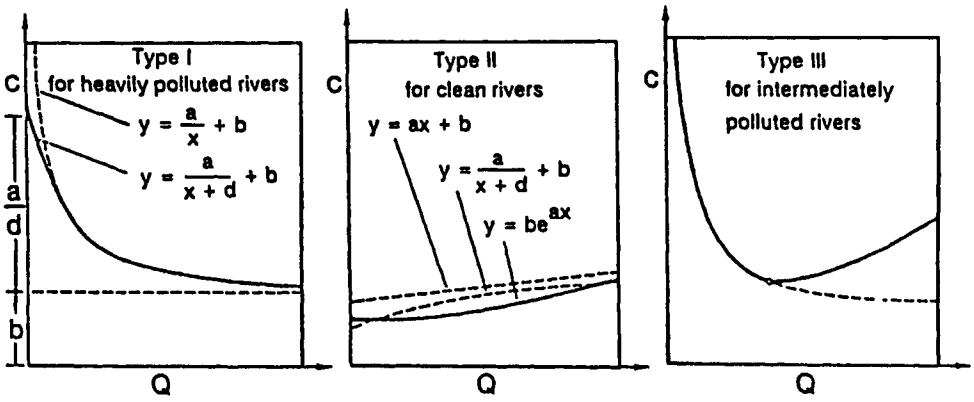



Fig. 3.8. Types of dependence of water quality variables on flow. Pollution from point sources is diluted by an increased flow, while pollution from diffuse sources increases as more material is washed by the precipitation from the soil and surroundings.

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

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

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

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

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

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

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

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

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

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

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

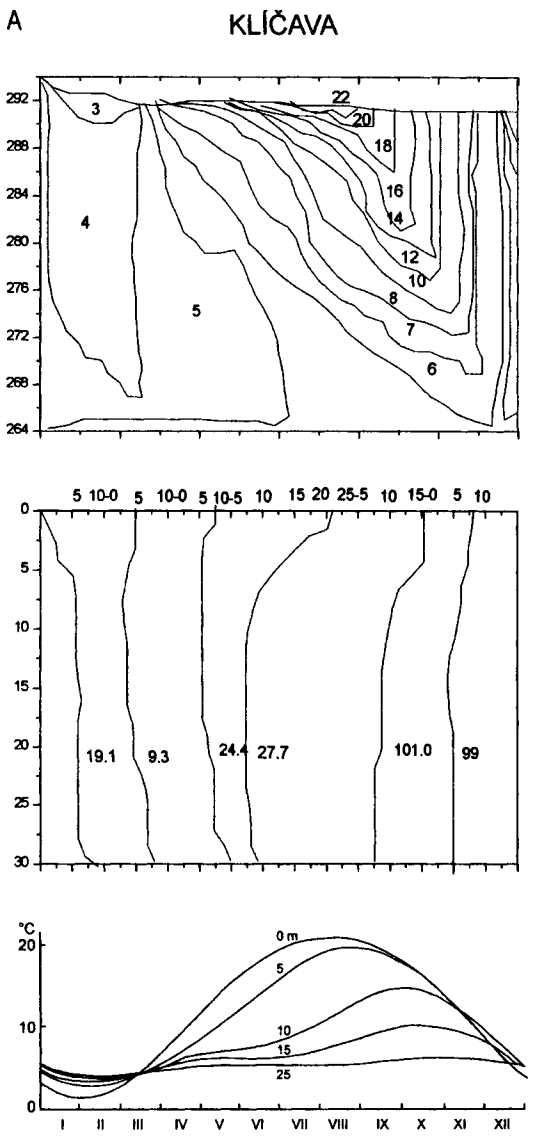



Fig. 3.9. Different ways of representing the changes of depth–time distribution or space–time distribution of daily or annual changes of temperature and other variables in lakes. A—Three ways of presenting annual changes of temperature at different depths. The above graphs represent isoline chart, the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

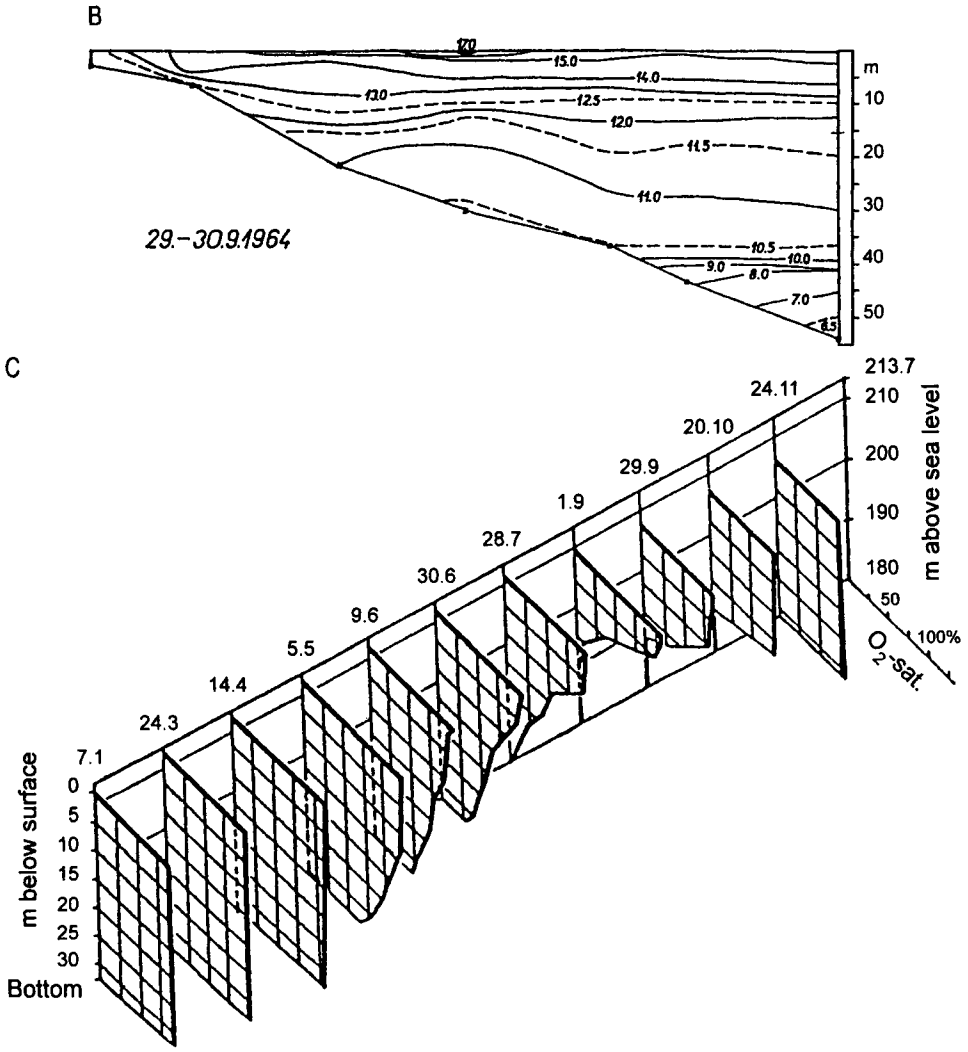



Fig. 3.9 (*continued*). B—Horizontal distribution during one sampling; C—representation of profiles in a three-dimensional form. When there is much over-crossing of the profiles, the graph becomes much less transparent.

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

*Evaluation of horizontal concentration differences*

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

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

*Evaluation of water quality trends*

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

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

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

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

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

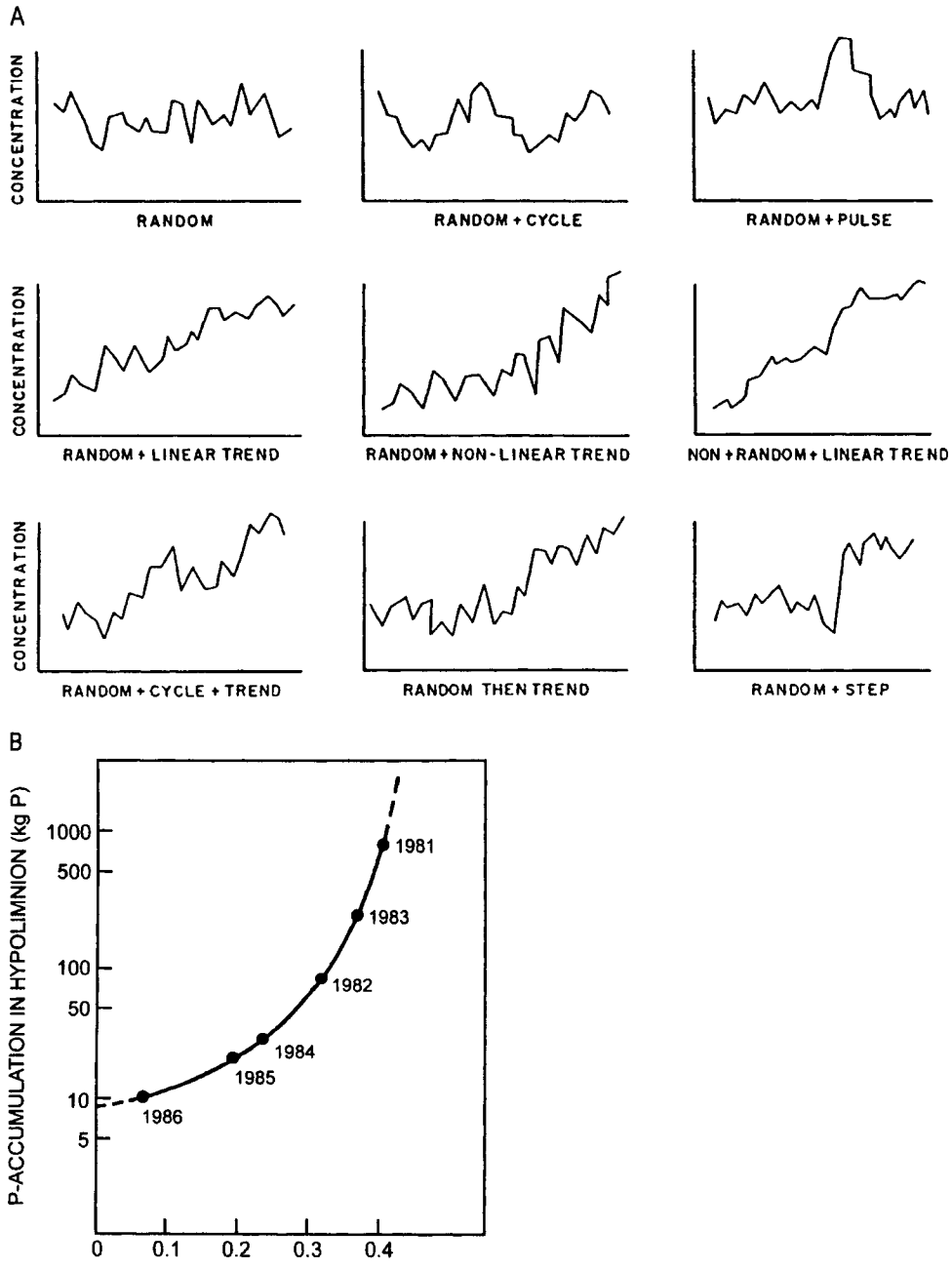



Fig. 3.10. Different types of trends of water quality variables. A—schematic representation of different possible types of trends; B—An example of a typical water quality trend.

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

### *3.4.2 Guidelines for Water Quality Evaluation*

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

*The degree of lake stratification*

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

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

The two groups of water quality variables that can be characterized by the distinctness of their stratification (*CDS*) in deep lakes/reservoirs are as follows:

(A) Markedly and consistently stratified,

(B) Nonstratified or only temporarily stratified.

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

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

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

*Evaluation of water transparency*

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

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

*Organic matter*

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

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

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

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

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

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

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

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

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

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

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

*Oxygen*

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

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

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

*Phosphorus*

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

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

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

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

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

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

*Chlorophyll-a*

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

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

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

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

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

*Nitrates*

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

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

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

*Microbial parameters*

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

*Psychrophilic bacteria*

Higher psychrophilic bacteria counts indicate a high content of easily-decomposed organic matter, which may not be of fecal origin. These compounds can make water treatment

difficult because of their odor. High psychrophilic bacteria counts are found in polluted inflowing streams and at high water flow rates. They decrease slowly in a lake, via their sedimentation and their elimination by zooplankton. In stratified waters, psychrophilic bacteria counts are usually only hundreds per milliliter of water. Bacterial counts comparable to the inflow of a reservoir (or higher) are sometimes found in water layers near the reservoir dam, and usually indicate a short-circuiting undercurrent of polluted water from the inflow directly to the dam area. In eutrophic waters, with very high levels of primary production, psychrophilic bacteria may increase to counts of thousands/milliliter and higher because of decomposition of large phytoplankton biomasses.

*Mesophyllic bacteria*

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

*Coliform bacteria*

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

*Faecal streptococci*

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

## 3.5 HOW TO MAKE CONCLUSIONS FOR LAKE MANAGEMENT

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

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

### *3.5.1 Assessing the Watershed Effects on Lake Water Quality*

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

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

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

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

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

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

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

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

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

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

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

*Recreational use* of drinking water supply waterbodies can be a serious public health problem, resulting from direct contamination of a waterbody by human pathogens and pollution by trodden mud. Thus, the public-health aspects of lakes must receive the highest priority. Nevertheless, although raw drinking water is taken from the outflow or the reservoir proper, bathing is permitted in many reservoirs. In such cases, certain minimum mea-

sures must be taken. The recreation should be restricted to zones distant from the drinking water intakes. One measure is the installation of dry latrines for bathers as far from the waterbody as possible, using concrete cofferdams insulated by 10 meter layers of sand mixed with iron swarf (1–2% of the weight of sand). Furthermore, parking lots and stalls should not be closer to the waterbody than the second zone. Restaurants and shops must not be closer than 300 meters from the shoreline, and their sewage diverted from the waterbody. Regular removal of all kinds of garbage must be strictly enforced. The hazard of bacterial contamination is particularly serious in shallow waterbodies, whose sediments are intensively disturbed by bathers. Faecal bacteria (including pathogenic species) can survive in sediments for several months, even in winter. Recreation use should be prohibited at small, shallow, water supply waterbodies of the fish-pond type.

### 3.5.2 *Conclusions Concerning the Waterbody*

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

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

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

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

(D) Deterioration of water quality, due to an *excessive production of organic matter* in the form of phytoplankton. The load of phosphorus (and other critical nutrients), the optical qualities of water (color, turbidity), pH and the carbon dioxide ($CO_2^+$) concentration of in the inflowing stream are of prime importance. The critical nutrient concentration in a waterbody depends on its internal cycling (e.g., binding or release

in sediments, incorporation by organisms), the development of phytoplankton, temperature, light and mixing, and biotic relations (i.e., consumption of phytoplankton by higher trophic level organisms in the food chain).

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

### *3.5.3 Conclusions for Water Treatment Plants*

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

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

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

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

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

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

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

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

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

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

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

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

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

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*Chapter 4*

# MEASURES FOR IMPROVING WATER QUALITY

This chapter deals with preventive and corrective methods for improving water quality, both in the watershed and in the waterbody into which it drains. Facilitating the application of preventive methods is the goal. However, corrective methods also can be important if preventive measures already have been applied to the maximum extent. Accordingly, the chapter is divided into four sections; Section 4.1 is introductory, Section 4.2 is devoted to measures applied in the watershed, Section 4.3 summarizes in-lake methods and Section 4.4 focuses on combined and innovative methods.

Section 4.2 discusses methods of water quality management within the watershed in four major categories: pollution prevention from forests and agriculture (Section 4.2.1), clean production by factories (Section 4.2.2), wastewater treatment by municipalities (Section 4.2.3), and ecotechnological methods using absorption of pollution by plant communities in natural and constructed wetlands (Section 4.2.4).

Section 4.3 distinguishes seven groups of in-lake measures, including biomanipulation or eutrophication control based on modifying food chain relations (Section 4.3.1), various methods of lake mixing and oxygenation (Section 4.3.2), hydraulic regulation by modifying the flow through lakes (Section 4.3.3), biomass removal (Section 4.3.4), methods of treating lake sediments (Section 4.3.5), nutrient inactivation methods in the waterbody (Section 4.3.6), methods that treat sediments (Section 4.3.7), and other methods (Section 4.3.8).

Section 4.4 identifies some examples of successfully used combinations of in-lake approaches. It also discusses some innovative approaches that have not yet been verified, but which show promise.

## 4.1 INTRODUCTION TO REMEDIAL MEASURES

When pollution abatement was discussed during the first green wave in the late 1960s, so-called "end-of-pipe" techniques were the primary consideration. There was a general optimistic view on the possibilities of using technology to solve all environmental problems. A typical statement from the late 1960s was: No dilution, but treatment. Today, almost 30 years later, we realize that environmental management comprises a very complex set of problems, and that only consideration of a wide spectrum of approaches can solve the problems properly. It is not surprising that solving complex environmental problems associated with very complex systems such as lakes will require complex approaches and techniques. Thus, very careful selection of proper management approaches on the basis of substantial knowledge of the system and the problem is necessary.

Thirty years ago, diffuse (nonpoint) pollution did not play any role in the green debate. Since the late 1970s and the early 1980s, however, it has become clear that nonpoint pollution sources, as well as internal sources (e.g., lake sediments), often are more significant pollution sources than the point sources, which often can be reduced significantly by an easily manageable "end-of-pipe" technique. This development has provoked a new type of pollution control approach, denoted *ecotechnology*, which is better able to cope with diffuse pollution sources, which are discussed in this chapter (also see Chapters 5 and 6).

Ecosystems are living systems, and do not react in the same predictable manner as rigid physical systems. In fact, ecosystems have often exhibited surprising reactions to changed impacts, which obviously complicates environmental management considerations. A deeper understanding of systems ecology, therefore, is a prerequisite for a more ecologically-sound environmental management approach, one of the threads unifying this entire document. The state of a lake is obviously dependent on the forcing functions that cause the observed impacts. The history of the lake, however, also is important in assessing its possible reactions to changed water quality or other type of impact. It can be shown that the same impact on a lake (e.g., the same total phosphorus concentration in lake water) may result in two different responses, depending on the structure of the system, which again is determined by the history of the system.

Many failures in lake management can be explained by the omission of the ecological consideration of the waterbody. As an example, an ecosystem resists changes, with a tendency to try to maintain its existing structure. A reduction in external forcing functions (e.g., a pronounced reduction in the phosphorus load entering a lake from wastewater discharges) is often not sufficient to restore a lake or reservoir. If the structure of the food web in a waterbody is adapted to a high phosphorus input, it will require very pronounced reductions in the phosphorus load to shift to a mesotrophic or oligotrophic condition. Thus, biomanipulation is an alternative to support the system in its efforts to shift to another food web structure. In this context it is absolutely necessary to consider the storage of nutrients and other pollutants during the implementation of a management strategy. Significant quantities of nutrients and other pollutants may be stored in the lake bottom sediments. If less-polluted water flows into a lake as a result of proper wastewater treatment, nutrients stored in the sediments may be released if appropriate chemical conditions exist in the water layers at the bottom of the lake and can significantly delay its restoration. This internal pollutant source often may be very significant, dominating over external pollutant sources for an extended period (also see Section 2.2.2—*Eutrophication*).

The importance of internal pollution sources is closely related to the proper timing of effective management strategies. Because an ecosystem has the ability to reduce the impacts of pollutant inputs by various accumulation processes, accumulation in the sediment being one of the most significant, it becomes very important to reduce the pollution sources at an early stage. Environmental management efforts in Denmark clearly illustrated how a good management strategy, applied at the wrong time, can lead to significant increases in pollution abatement costs. An ambitious environmental management strategy was launched in Denmark in 1976, with the goal of letting the ecosystem determine the best restoration

strategy. Consequently, it was necessary to make plans about the use of all aquatic ecosystems. Could discharge of wastewater be tolerated? What were the recreational values of the various aquatic ecosystems? Did some ecosystems have a particular scientific value? Which ecosystems were valuable as water resources? Many more questions became evident. To answer these crucial questions required more than 7 years of interactions between the different levels in the political hierarchy, from the communities to the government, which resulted in excellent environmental plans, but no pollution abatement. The Gordian Not was cut in 1986, when it was decided that all Danish wastewater treatments would meet certain standards, including a biochemical oxygen demand ($BOD_5$) $\leqslant 10$ mg $l^{-1}$, total phosphorus concentration $\leqslant 1.5$ mg P $l^{-1}$ and a total nitrogen concentration $\leqslant 8$ mg N $l^{-1}$. In addition, it was possible to assess even more strict requirements for particularly important or vulnerable ecosystems (e.g., lakes of important recreational value). This goal was subsequently widely applied in Danish lake management efforts. The result was that the entire environmental strategy in Denmark was quite successful—except for the timing. These measures should have been implemented 7–9 years earlier! The consequences were that 7–9 additional years of phosphorus (and nitrogen) discharges accumulated in the Danish lakes, making it considerably more difficult to find a management strategy able to return the lakes to the conditions prevailing 30–50 years earlier. A specific example of this failure is Arresø in Zealand, Denmark's largest lake. The lake is very shallow, with a maximum depth of 4 meters, which obviously accentuates the importance of nutrients accumulated in the lake sediments. If a proper environmental management had been implemented in the mid-1970s, the condition of the lake today would have been close to its condition in the 1950s. However, because of the above-noted delay of 7–9 years, so much additional phosphorus had accumulated in the sediments that it may take more than 100 years to bring the lake back to its previous conditions based solely on reduction of its phosphorus load. In fact, it may be necessary to remove the upper 0.5 meters of the lake bottom sediment to significantly reduce the internal phosphorus load, which will cost as much as US $60–80 million based on a sediment surface area of 41 $km^2$. Under all circumstances it will be necessary to apply restoration methods.

As this example illustrates, lake restoration may be very expensive. It is usually best, therefore, to develop and implement good lake environmental planning at an *early* stage, in order to be able to also take preventive measures that would eliminate the problems of phosphorus accumulation, as well as heavy metals and persistent organic chemicals, in lake sediments. A proper environmental planning also would help identify the pollution priorities that a good strategy should address.

Many developing countries lack the economic resources to undertake appropriate environmental management programs. Nevertheless, the reality is that proper environmental planning can be carried out at a fraction of the costs of major pollution abatement programs. It is strongly recommended, therefore, that development plans include environmental planning efforts at an early stage. This will facilitate their assigning a proper priority to the various steps in the environmental management planning process, and also making better use of preventative preventive measures, both of which will work to enhance the possibilities of moderate-cost solutions over the long term.

As with other ecosystems, lakes and reservoirs are "open systems". Thus, it is not sufficient to consider only the lake or reservoir water basin in a management regime. Indeed, it also is essential to consider the entire watershed of a lake or reservoir, in order to include all the pollution sources in the management considerations. It is often much more cost-effective, for example, to reduce the concentration of a pollutant at its source—a solution that can only be identified by considering a lake and its watershed as one combined system. Further, there are usually many more possible solutions to lake and reservoir problems when we attempt to optimize a large, combined lake–watershed system, rather than focusing only on the lake itself. Including the entire watershed in environmental management considerations is of particular importance for developing countries, which generally have fewer resources to devote to later pollutant abatement efforts than do the developed countries.

The use of ecological modelling has become much more important over the last 30 years, due to a need for quantification in ecology and environmental management efforts. This is probably the only significant tool available for obtaining a quantitative overview of a complex ecosystem such as a lake or reservoir, which is a prerequisite for selecting an optimum solution to the complex problems facing them. Because solutions to the entire spectrum of environmental problems require quantitative estimation methods to assist in identifying a realistic trade-off between ecological and economic concerns, it is not surprising that ecological models have been used increasingly in environmental management efforts. Chapter 5 provides an overview of models in environmental management of lakes and reservoirs.

From this introductory discussion, several important recommendations may be deduced, as summarized in the following six points:

- Expect that a proper environmental strategy will require a wide spectrum of approaches and techniques,
- Expect that proper environmental management will require the application of a combination of "end-of-pipe" technologies (environmental technology), ecotechnology, cleaner technology and environmental legislation,
- Correct timing in applying the various steps in environmental management efforts is extremely important; thus, it is recommended that a comprehensive environmental management plan be developed at a very early stage, in order to be able to use the available resources in the most optimal manner,
- It is usually very beneficial, particularly from an economic perspective, to consider prevention, rather than correction, primarily because it is often very costly to restore heavily degraded lakes,
- Because of the complexity of ecosystems and their problems, proper ecological knowledge about ecosystems is a prerequisite for ecologically sound environmental management programs; this is the only reasonable method for avoiding unexpected ecosystem responses,
- Optimum solutions to environmental management problems are best obtained if the entire lake–watershed ecosystem is taken into consideration in developing and implementing management actions.

## 4.2 WATERSHED METHODS

Ecotechnological methods used in the watershed (Table 4.1) can be classified as either preventive or corrective. Preventive methods, directed to the creation of conditions minimizing pollution, are preferable because they are cheaper for society than the implementation of corrective, remedial measures. It is very unwise, from a global perspective,

Table 4.1. Ecotechnological methods applied to reservoir watershed management and recovery (modified from Straškraba et al., 1993; Thornton et al., 1999)

| Problem to be solved | Methods |
|---|---|
| Organic pollution: | • Clean production<br>• Diversion of effluents<br>• Purification plants<br>• Wetlands |
| Excess nutrients and eutrophication: | • Diversion of wastes<br>• Tertiary treatment plants<br>• Progressive agricultural practices<br>• Meadow and riparian forest zones on the vegetated banks<br>• Natural and constructed wetlands<br>• Pre-impoundments at the inflows<br>• Wahnbach P-reduction plant |
| Eutrophication and oxygen depletion of rivers: | • River restoration<br>• Re-oxygenation |
| Reservoir siltation: | • Erosion control<br>• Rehabilitation of river banks<br>• Reforestation<br>• Groundwater recharge<br>• Pre-impoundment of inflows |
| Heavy metal contamination: | • Reduction of polluted effluents<br>• Wetlands |
| Acidification: | • Liming<br>• Organic matter additions |
| Salinization: | • Improved irrigation practices<br>• Decreased fertilizer applications<br>• Decreased road salting |
| Decreased biodiversity due to reservoir construction: | • Prohibit introduction of foreign species<br>• Reintroduction of native species<br>• Maintenance of wetlands as nursery grounds<br>• Maintenance of preserved areas for native species |

to first release pollutants in diluted form into canals and surface waters, and then to extract and treat them with costly methods. All pollution extraction procedures are cheaper when the pollutant concentrations are high. Some approaches will be the same for prevention as for correction—the only difference will be in the timing. Prevention must be done early, before pollution starts. A clear example of the importance of timing was shown above in Section 4.1. A typical corrective group of methods is represented by wastewater treatment. Unfortunately, this is frequently the most often, if not the only, method of pollution abatement considered.

One obstacle to using preventive measures is the sectoral character of infrastructure. It shifts the impacts from water to land, to sectors such as forestry, agriculture and industry. However, because water resource managers typically have the information on the state of the water resources and the costs of corrective measures, they also are responsible for documenting and advertising the societal advantages of preventive, rather than corrective, approaches, as well as the possibilities of other sectors to prevent pollution.

Many new ecotechnological methods use the capabilities of natural systems to adsorb or otherwise neutralize pollution. However, mastering these methods requires "fine-tuning" between technical and natural means, as well as the need to use more, and until now less understood, knowledge. In a way, they substitute the brute force of modern technology with specific knowledge and, in this way, are representative examples of the informational revolution currently underway. A typical example is the use of wetlands for water purification. Wetlands are considered less reliable than classical wastewater treatment for such purposes, mainly because they are affected by weather and other conditions, and because we know less about how wetlands work to produce their effects. However, such factors as their low price, their natural availability in many places around the world, and their high purification efficiency make natural wetlands ideal tools for pollution prevention, and constructed wetlands ideal for pollution abatement.

### *4.2.1 Pollution Prevention from Forests and Agriculture*

Pollution sources from forested and agricultural areas are primarily nonpoint or diffuse in character. The newest experiences with diffuse, nonpoint source pollution and the possibilities for its abatement, are summarized by Thornton et al. (1999). Fertilizers applied to fields (and sometimes to forests) and post-harvest protection chemicals (and the heavy metals they contain) are major polluting substances. The resulting pollution primarily impacts ground and surface waters. However, increasing importance also is being given to air pollution by ammonia from animal husbandry operations. There are also point pollution sources from agriculture, examples being pollution from concentrated husbandry units or agricultural product enterprises. However, the later ones can be considered industrial enterprises, the food industry being just one type.

Specific methods for treating diffuse pollution from agriculture and forestry fall into the following groups:

(i) Methods associated with crop cultivation,
(ii) Methods directed to forestry,

(iii) Methods directed to controlling the application of herbicides and pesticides,
(iv) Methods directed to decreasing the quantity of agricultural and forest pollution reaching surface waters.

*Diffuse pollution from different types of agricultural cultivation*

Agricultural components leading to water pollution may be of three kinds: crop production, pastures, and animal production. Diffuse pollution from agricultural crop activities consists mainly of nutrients, heavy metals and salts from fertilizers and organic chemicals used for crop protection. Pollution from livestock cultivation comes mainly from animal faeces, urine, and pasture erosion. The focus in this section is on nutrients, although they are often accompanied by heavy metal and mineral (salt) pollution. Salts represent a large percentage of applied fertilizers and are a significant component of the generally-increasing salt concentration in natural waters (see Section 2.2.2—*Salinization*). Although the quantity of heavy metals in applied fertilizers is low, their influence on the environment is very negative (see Section 2.2.2.—*Metals*).

Soils deprived of nutrients during harvesting are often fertilized with either manure or chemical fertilizers. Fertilizers are typically in the form of nitrogen, phosphorus and potassium (N–P–K), but only nitrogen and phosphorus are aquatic pollutants, provoking lake and reservoir eutrophication. There is a difference between the two elements in regard to their retention in soils. The retention of phosphorus is high, while nitrogen is retained to a lesser degree. There also is a difference between temperate and tropical soils in this respect; nitrogen retention in soils in tropical regions generally is higher than in temperate regions. Nitrogen also has an additional function, relative to its role as a plant nutrient; namely, it represents a possible cause of human illness (see Section 2.2.2—*Eutrophication*). Different crops have different nutrient needs, as do different varieties of the same plant, with the need being irregular in time and dependent on soil type.

The degree of cropland pastures and forest pollution depends mainly on the type of crop, soil type and soil erosion, type of fertilizer and the method of fertilizer application.

*Crop types and field cultivation methods*

Different vegetation formations have differing abilities to retain nutrients. A major method of minimizing pollution is to retain forests because they possess the highest retention capacity for pollutants, and to use meadows wherever possible. In regard to cultivated crops, those having dense growths should be used, although those whose use will produce strips of bare soils should be omitted, or else the bare strips should be filled with other vegetation. The general functions of forests, meadows, and dense vegetation are two-fold: high nutrient retention and prevention of soil erosion, thereby also preventing erosion-associated nutrient losses.

Different agricultural crops produce different quantities of nutrients exported to receiving waterbodies, depending primarily on their nutrient needs. Row crops produce larger nutrient exports than non-row crops. Low nitrogen pollution can be achieved by using the nitrogen-fixing ability of some plants, resulting from symbiotic nitrogen-fixing bacteria (particularly in the plant family Fabacae). Research in Brazil lead to the discovery that

some varieties of mass-cultivated plants grown on low nitrogen soils also possess symbiotic nitrogen-fixing bacteria, thereby having a low need for fertilizer nitrogen. Particular attention is given to sugar cane, which is used as a source of alcohol for cars. These results have great potential for improving water quality.

On the one hand, conventional tillage methods (e.g., fallow land during nongrowth season, harvest removal of crop residues) results in large nutrient exports. On the other hand, conservation tillage practices reduce nutrient export. Unfortunately, however, the latter also results in increased herbicide usage and runoff.

Pastures typically do not produce large quantities of aquatic pollutants, except when fertilized. Limiting livestock grazing time on a given parcel of land also reduces nutrient export. The greater the animal density, the greater the potential for increased pollution.

*Soil types, quantities and erosion*

Regional differences in the percentage of nitrogen released from applied fertilizers are related to the character of soils, and particularly their water permeability. Less water-permeable soils retain more nutrients (e.g., phosphorus export from sandy or gravel soil generally is small, compared to a large export from clay soils and organic soils).

Because erosion plays a major role in increasing pollutant loads to waterbodies, techniques to prevent erosion must be applied in such cases. On land with slopes of more than a 15% gradient, the field orientation must be shifted to contour strip-cropping, with proper selection of crops. Dense, permanent vegetation is the best means for protecting watersheds against erosion and the release of nutrients, particularly phosphorus. Applying purely technical means to prevent erosion can be costly.

*Fertilizer type*

The use of manure is preferred over the use of artificial fertilizers in regard to preventing water pollution. This is because the nutrients are released from manure only slowly, thereby allowing plants to keep pace with the released nutrients. Straw, which is a considerably component of manure, also can be used as a retarding element for nutrients. A general observation is that the inefficient use of nutrients simultaneously represents losses for agriculture and additional expenses for addressing water pollution. In addition to its nutrient content, the other components of fertilizer also are important in regard to water pollution concerns. For example, the availability of fertilizers for plants, and therefore the magnitude of the resulting pollution, depends on the compound in the fertilizer to which the nutrient is bound. Additional pollution problems are created by the salts contained in fertilizers. Water salinity is now generally increasing around the world, a primary reason being the increased used of fertilizers. Fertilizers also must be checked for the presence of components toxic to humans.

*Fertilizer application*

Crops are able to utilize fertilizer nutrients only up to certain doses of fertilizer application (Fig. 4.1). Thus, farmers must be shown that the application of fertilizers above a certain

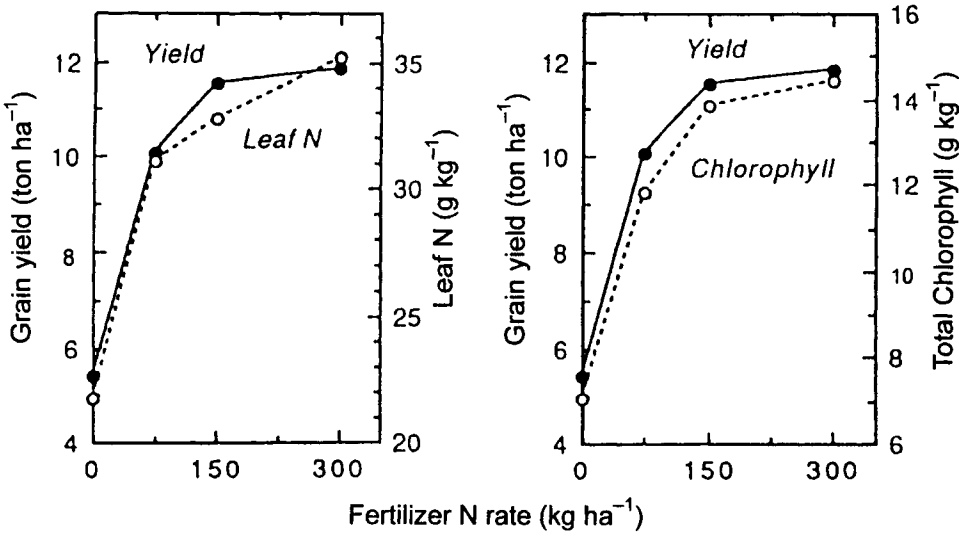



Fig. 4.1. Effect of fertilizer nitrogen rate on leaf chlorophyll and nitrogen concentrations and yield of three corn hybrids, showing the saturation effect at some 150 kg per hectare.

limit inevitably leads to fertilizer and financial losses, with the excess fertilizer also causing pollution of ground and surface water. The curvilinear dependence seen in Figure 4.1 also indicates that, as the upper limit of plant reactions to fertilization is approaching, the crop return becomes lower and, therefore, application of fertilizer up to this limit is inefficient. The respective fertilizer should be applied in a form that facilitates maximum nutrient utilization by the crops. Fertilizer in the form of pellets, particularly of certain types, is superior to powders. Airplane application during dry weather should be avoided, as it creates greatest spreading of fertilizers to waterbodies. The timing of fertilizer application should be synchronized with plant needs. Fertilizers should never be applied on bare field without vegetation. The greatest fertilizer nutrient losses are associated with winter application on snow, since much of the nutrient movement to watercourses occurs during the thaw period. Incorporating applied fertilizers into the soil also results in reduced nutrient exports. Fertilizer storage areas should not be located in the vicinity of watercourses, and should ensure protection against the dispersion of stored manure.

*Animal production*

Nutrient exports increase with increasing animal density. Nutrient exports also increase with the extent of impervious substrates in feedlots. The use of detention basins decreases nutrient export. It appears that nutrient export decreases with an increase in the roof area : feedlot area ratio (Thornton et al., 1999). Innovative ecotechnological protection methods, such as vegetative filter strips, should be used (Toombs, 1997).

*Control measures for nonpoint source pollution from agriculture*

Various control options follow from the above evaluation of different diffuse pollution sources, as summarized under the following headings (Thornton et al., 1999):

- Conservation tillage (crop residues not removed, field continuously covered),
- Contour farming/strip cropping (plugging along contours, no large bare strips),
- Integrated pest management (low pesticide use, good chemical selection, biological methods),
- Management of pasture/range lands,
- Crop rotation (with respect to differential nutrient needs; nitrogen fixing plants),
- Terraces (no steep slopes),
- Management of animal wastes (prevention of leakage to waters),
- Fertilizer management (protection of stores, application form and timing),
- Livestock exclusion.

*Preventing pollution from forest cultivation*

In addition to their positive effects on water quantity, forests also have positive effects on water quality. They not only do not release polluting substances, but are able even to retain pollution entering them from outside the forest. The only negative influence of deciduous forests on water quality is represented by the humic substances released into water, so that both streams and lakes have low pH and a brown color. Treating such water for drinking water purposes requires special care. This positive effect turns into a negative one in the case of forests damaged by acidification, when the extremely low pH causes the release of aluminum in a toxic form. Further, some forests used for very rapidly growing wood production can have negative effects on water quality. This is particularly true for the Australian *Eucalyptus*, which exists in many parts of the world.

The methods of managing forests with regard to protection of water quality, therefore, are consistent with balanced forestry practices aimed at multigoal forest functions, but not with the one-sided wood production technology: keeping a healthy mixed forest, no clear cutting, sensitive use of machinery. Forest burning caused by natural events and local pests attacking trees causes less damage than spraying forests with pesticides or the mass eradication of damaged trees using heavy machinery. Fertilizing forests should especially be avoided in watersheds used for drinking water supply.

Special attention should be given to riparian forests on the slopes of river valleys and other steep slopes. Their buffering and nature protecting function is manifold, including,

- Protection against erosion,
- Capturing silt, nutrients and other pollutants originating from fields, thereby preventing water pollution,
- Serving as migration corridors for animals,
- Protecting biodiversity by serving as refuges for different groups of plants and animals,
- Keeping landscape heterogeneity.

Recent research devoted to the quantification of forest riparian buffer zones (e.g., Lowrance et al., 1997; Snyder et al., 1998) demonstrates their efficiency for surface water protection.

Control measures to prevent water pollution from forestry can be summarized as follows (Thornton et al., 1999):

- Maintenance of ground cover,
- Management of roads and trails,
- Management of riparian zones,
- Management of biocides.

*Preventing water pollution from application of herbicides and pesticides*

Herbicides and pesticides also can be toxic to aquatic organisms and humans. As a result, only strictly-controlled chemicals and their brands can be used in developed countries. Particular care in their use is required in water supply watersheds. Water quality managers in developing countries must be careful to protect watersheds against the unfortunate habit of chemical producers providing developing countries with products not permitted in the developed world.

The rules of agrochemical application safe for water quality are the same as for fertilizers; namely, fluid forms are superior to powder forms, spreading during windy conditions and from airplanes, and in the close vicinity of streams and lakes, is to be avoided, and storage places are to be protected from leaking containers, particularly during rains.

*Preventing pollution from reaching surface waters*

Because the entry of some portion of applied fertilizers and other agrochemicals into groundwater cannot be completely avoided, management practices directed to reducing the entry of contaminated water into receiving waterbodies should be utilized. The practices consist first of all in utilizing the pollutant binding capacity of forests and meadows. Buffer zones along watercourses, particularly on the slopes of the river valleys, should consist only of forests or meadows for a minimum width of 50 meters (Haycock et al., 1997). Riparian mixed forests minimize the entry of fertilizers into watercourses, but also have other positive functions. Their hydrologic function is positive, causing water retention and retarding floods. Buffer zones are environments for organisms that consume pests, thus decreasing their spread. They also facilitate the survival and migrations of animals. The significant positive functions of marginal vegetation in decreasing nutrient concentrations is mentioned in Section 4.3.4.

If the highest nitrate concentration in receiving waters is close to 50 mg $NO_3$ $l^{-1}$ (especially in high-flow periods), the following measures should be taken in the watershed, in cooperation with appropriate agricultural enterprises:

- Strictly conserving the second zone of public-health protection,
- Using nitrogen fertilizers in quantities not exceeding 100 kg $ha^{-1}$ of farmland per year,
- Reducing the time during which fields are left without vegetation after harvest to the maximum extent (1–2 weeks at most), and to use catch crops,
- Avoiding the application of nitrogen fertilizers to frozen soil or onto fields in which no crops have been sown or planted.

Protective stretches of land surface formed by grasslands and forests prevent surface soil particles being washed down into streams and waterbodies. However, they may not reduce the leaching of dissolved pollutants (e.g., nitrates) from distant, elevated places.

Section 4.2.4 provides a discussion of the effective function of wetlands. Prevention of erosion, using methods discussed in Section 4.2.1, is necessary to stop the surface runoff of pollutants to streams.

Methods for preventing agricultural pollution can be summarized as follows:

- Agriculture is one of the greatest sources of diffuse pollution in many places around the world. Care should be taken to minimize diffuse pollution, which is only possible in cooperation with agricultural practitioners. It is to the benefit of agriculture not to allow expensive fertilizers to remain unutilized by crops.
- A general rule concerning agricultural techniques can be stated as follows: What is most efficient for plants is also most favorable for protecting water quality because it creates the least pollution. Only unutilized fertilizer causes water quality difficulties. Agricultural best management practices, therefore, also are the most useful water quality practices.
- The loss of fertile soils to agriculture is related to the greatest water pollution. Thus, it is in the interest of both agriculture and water quality to minimize erosion, using such methods as contour plowing, terracing, and not leaving soil bare without vegetation. The prevention of erosion is very important for reducing water pollution by phosphorus.
- Forests have generally positive water quality effects, and deforestation results in water quality deterioration. Rapidly growing introduced *Eucalyptus* plantations have negative effects and are exceptions to this rule. Forest fertilization also results in negative water quality impacts.
- The use of plant protection agents must be strictly controlled, particularly since some are highly toxic to aquatic life and humans.
- Buffer zone, riparian forests, stream marginal vegetation and wetlands are effective in reducing water pollution from nutrients, salts and some other chemical compounds.

The protection of water quality from agricultural activities is not just a technical issue, but also a topic of interest to farmers and farming organizations. Water quality specialists, however, must take the lead in this area. According to Fried (1991), the following actions are suggested within the European Union.

A. With respect to fertilizer use:
   - Creation or reinforcement of advisory agencies for farmers, and the establishment of codes of good agricultural practices,
   - Integrating environmental concerns in the training of farmers,
   - Reinforcement of information by all means,
   - Development of scientific research (mostly addressing the mechanical, physico-chemical and biological phenomena of leaching).

B. With respect to intensive animal breeding:
   - Improvement of storage and transport, and creation of manure banks,
   - Improvement of spreading methods,

- Analysis and differentiation of manure types to achieve better environmental adaptation,
- Improvement of waste treatment,
- Improved conjunctive use of chemical and animal fertilizers,
- Improved training and information to farmers,
- More research on alternate uses of manure.

### 4.2.2 *Cleaner Production*

One important management method usually not under the direct control of agencies that manage water is summarized as *Cleaner Production* (Misra, 1996). It consists of making changes in the processes within a production plant, such that the emitted pollution is reduced. Major advantages inherent in this approach also can benefit the producer. Aside from reducing the expenses of fees for creating pollution, considerable savings of energy, water and various materials used in the production process also can be obtained. As an example, the introduction of the clean production approach in fifty galvanizing plants in The Netherlands resulted in a reduction in pollution to 55% of the original levels during the first year and, after gaining more experience, to 37% in the second year. In two large enterprises in the Czech Republic, primary savings attained by introducing clean production techniques totaled 50 million Czech Crowns. In focusing on this technology, the pollution reduction activity shifts to the production area. Interest in this method is primarily created by the ability to save considerable amounts of money by using it. But its value in helping solve environmental problems also plays a role. Agencies that manage water must teach industry leaders to use this approach and claim initiatives for pollution abatement.

An even wider approach analyzes the whole production process; namely, the life cycle analysis of a product. The production of a product is analyzed, with the goal of minimizing wastes over the entire production cycle, starting with the mining of raw materials through its use and finally to the disposal of the final product after use. This effort usually exceeds the capabilities of a given producer, demanding instead the synchronization of efforts by several enterprises. The environment has profited greatly in recent efforts of this kind, by wise and progressive industry leaders able to increase their competitive capabilities.

Industrial enterprises face increasing costs of raw materials, energy, water and pollution fees, and face increased pressure by environmental groups to reconsider their production methods and seek to retain competitiveness, while also demonstrating good will toward environmental protection. In addition to automation, this goal can be obtained by the recirculation of materials and water inside the production plants, by energy saving, by better space organization of the process, and by the minimization of transport. All these elements are important for protecting water quality, since the processes of material extraction, transportation and energy production can cause pollution, and the quantity and degree of the pollution of the water leaving the plant is decisive in regard to the water returned as effluent back to the plant. One scientifically rooted procedure with broad positive consequences for plant efficiency and competitiveness, as well as for water quality, that is favored by large international trusts is *the life cycle evaluation of products*, or *environmental life cycle assessment* (Curran, 1996).

The evaluation process in life cycle analysis consists of following a product throughout its entire life cycle, from its creation to its disposal. The production stages covered with this analysis include mining raw materials, processing for basic chemicals or metals, transportation, production of final materials needed for the product, production of the final product, packaging and distribution, the fate and environmental consequences of the product when it is used and, finally, the disposal of the remains or unused parts (e.g., the fate of packing materials, nonfunctioning parts or nonrepairable damaged products). Each step is evaluated in regard to the economy of the product, including the needs for materials, energy and water, and the environmental consequences of decisions at each step are estimated, as a means of identifying the cheapest, most efficient and least polluting option (Fig. 4.2). Many factories that have implemented this evaluation have found considerable savings and a resulting increase in competitiveness. This is typically accompanied by a considerable savings in water and energy resources, and a corresponding reduction in pollution. Thus, it is hoped that this procedure will become widely used and will benefit water quality. Because of increased water circulation within a factory, the quantity of the effluents also is reduced. Effective, specialized pre-treatment can be used, sometimes with the regeneration of some substances, which further reduces the pollutant load to the effluent or city wastewater treatment plant.

Considerable water quality improvement can also be obtained by citizens in their daily life on the basis of this approach, including the following:

- Water quality managers must stress the usefulness of clean production and product life cycle analysis, working to facilitate such evaluations. Local water management councils can be very helpful in this direction.
- Saving energy has positive consequences for water quality, as energy generation causes environmental degradation, including water pollution.
- Saving of water in households improves water quality, since existing wastewater treatment plants function better with less entering wastes, and there is less need for upgrading, renovation and new plant construction.

### *4.2.3 Wastewater Treatment*

Wastewater discharges into lakes and reservoirs is a man-made forcing function of crucial importance for their water quality. However, it also is a controllable forcing function. It is possible in many situations to control it completely, either by water diversion or by wastewater treatment methods. Water diversion, however, results in another downstream waterbody having to cope with the pollutant load. Thus, treating the wastewater properly should be considered a generally more acceptable solution to the problem. This gives rise to two questions; namely (i) is it possible to solve all pertinent wastewater problems, and (ii) what is understood as a proper wastewater treatment?

The water pollution problems associated with municipal wastewaters includes their content of:

- Nutrients causing eutrophication,
- Biodegradable organic matter, causing oxygen depletion,

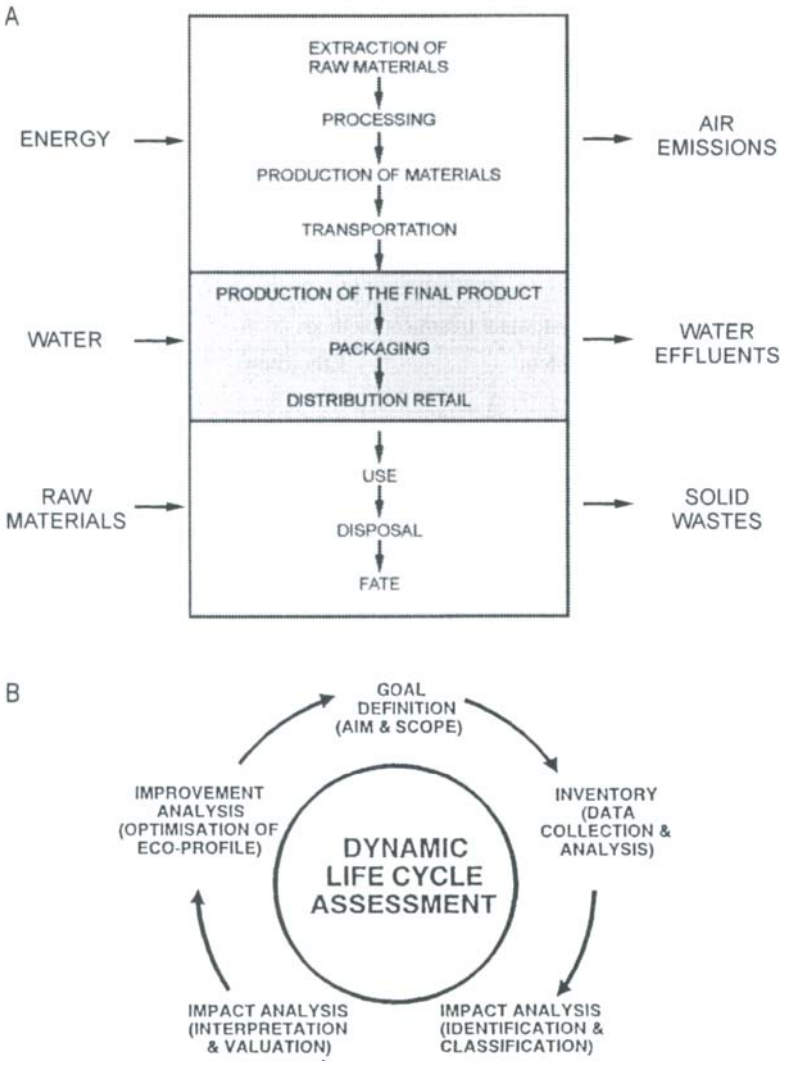



Fig. 4.2. Clean production and life cycle assessment of a product. A—The phases of the life cycle evaluation and the inputs and outputs taken into account (modified from Straškraba and Tundisi, 1999); B—Processes during the dynamic life cycle assessment (modified from Anonymous, 1994).

- Bacteria and virus affecting the sanitary quality of water, which is of particular importance when the water is used for bathing, swimming and drinking purposes,

- Heavy metals; namely, lead originating from petrol stations, zinc and cadmium from gutters, heavy metals from fungicides and other agricultural chemicals, and a wide range of other heavy metals in minor concentrations,
- Refractory organic matter, originating from industries, hospitals and the use of pesticides, and even from a wide spectrum of household articles.

Tables 4.2 and 4.3 provide an overview of a wide range of wastewater treatment methods. Clearly, there is a method available to address virtually any of the aforementioned problems.

Table 4.2. Survey of generally applied wastewater treatment methods

| Method | Pollution problem | Efficiency | Costs (US \$/100 $m^3$) |
|---|---|---|---|
| • Mechanical treatment | Suspended matter removal | 0.75–0.90 | 3–5 |
| | $BOD_5$ reduction | 0.20–0.35 | |
| • Biological treatment | $BOD_5$ reduction | 0.70–0.95 | 25–40 |
| • Flocculation | Phosphorus removal | 0.3–0.6 | 6–9 |
| | $BOD_5$ reduction | 0.4–0.6 | |
| • Chemical precipitation ($Al_2(SO_4)_3$ or $FeCl_3$) | Phosphorus removal | 0.65–0.95 | 10–15 |
| | Reduction of heavy metals concentrations | 0.40–0.80 | |
| | $BOD_5$ reduction | 0.50–0.65 | |
| • Chemical precipitation ($Ca(OH)_2$) | Phosphorus removal | 0.85–0.95 | 12–18 |
| | Reduction of heavy metals concentrations | 0.80–0.95 | |
| | $BOD_5$ reduction | 0.50–0.70 | |
| • Chemical precipitation and flocculation | Phosphorus removal | 0.9–0.98 | 12–18 |
| | $BOD_5$ reduction | 0.6–0.75 | |
| • Ammonia stripping | Ammonia removal | 0.70–0.95 | 25–40 |
| • Nitrification | Ammonium is oxidized to nitrate | 0.80–0.95 | 20–30 |
| • Active carbon adsorption | COD removal (toxic substances) | 0.40–0.95 | 60–90 |
| | $BOD_5$ reduction | 0.40–0.70 | |
| • Denitrification after nitrification | Nitrogen removal | 0.70–0.90 | 15–25 |
| • Ion exchange | $BOD_5$ reduction (e.g., proteins) | 0.20–0.40 | 40–60 |
| | Phosphorus removal | 0.80–0.95 | 70–100 |
| | Nitrogen removal | 0.80–0.95 | 45–60 |
| | Reduction of concentrations | 10–25 | |
| • Chemical oxidation (e.g., with $Cl_2$) | Oxidation of toxic compounds | 0.90–0.98 | 60–100 |
| • Extraction | Heavy metals and other toxic compounds | 0.50–0.95 | 80–120 |
| • Reverse osmosis | Removes pollutants with high efficiency, but is expensive | | 100–200 |
| • Disinfection methods | Reduction of microorganisms | High, can hardly be indicated | 6–10 |

Table 4.3. Efficiency matrix relating pollution parameters and wastewater treatment methods

| Method | Suspended matter | $BOD_5$ | COD | Total | | Ammonium nitrogen | Heavy metals | *E. coli* | Color | Turbidity |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | | Phosphorus | Nitrogen | | | | | |
| • Mechanical treatment | 0.75–0.90 | 0.20–0.35 | 0.20–0.35 | 0.05–0.10 | ~0 | 0.10–0.25 | 0.20–0.40 | — | 0.80–0.98 | |
| • Biological treatment* | 0.75–0.95 | 0.65–0.90 | 0.10–0.20 | 0.05–0.10 | ~0 | 0.10–0.25 | 0.30–0.65 | Fair | ~0 | — |
| • Chemical precipitation | 0.80–0.95 | 0.50–0.75 | 0.50–0.75 | 0.80–0.95 | ~0 | 0.10–0.60 | 0.80–0.98 | Good | 0.30–0.70 | 0.80–0.98 |
| • Ammonia stripping | ~0 | ~0 | ~0 | ~0 | 0.70–0.96 | 0.60–0.90 | ~0 | ~0 | ~0 | ~0 |
| • Nitrification | ~0 | ~0 | ~0 | ~0 | 0.80–0.95 | 0.80–0.95 | ~0 | Fair | ~0 | ~0 |
| • Active carbon adsorption* | — | 0.40–0.70 | 0.40–0.95 | ~0.1 | High** | High** | 0.10–0.70 | Good | 0.70–0.90 | 0.60–0.90 |
| • Denitrification after nitrification | ~0 | — | — | ~0 | — | 0.70–0.90 | ~0 | Good | ~0 | — |
| • Ion exchange | −0.40 | 0.20–0.50 | 0.20–0.95 | 0.80–0.95 | 0.80–0.95 | 0.80–0.95 | 0.80–0.95 | Very good | 0.60–0.90 | 0.70–0.90 |
| • Chemical oxidation | — | Corresponding to oxidation | ~0 | ~0 | ~0 | ~0 | ~0 | ~0 | 0.60–0.90 | 0.50–0.80 |
| • Extraction | — | Corresponding to extraction of toxic compounds | ~0 | ~0 | ~0 | ~0 | 0.50–0.95 | ~0 | ~0 | ~0 |
| • Reverse osmosis*—see Table 4.2 | | | | | | | | | | |
| • Disinfection methods—essentially application of chlorine, ozone, etc. | | | | | | Very high | 0.50–0.90 | 0.30–0.60 | | |

*Depends on the composition.

**As chloramines.

Industrial wastewater can cause the same water pollution problems as municipal wastewater. In addition, they also can contain toxic organic and/or inorganic compounds (particularly heavy metals and persistent organic pollutants). However, it is necessary to solve at the source the problems associated with industrial wastewater that cannot be solved with municipal wastewater treatment methods. The major portion of toxic substances must be removed by the industries, since they will be removed only partially, if at all, at municipal wastewater treatment plants, and/or will contaminate the sludge produced at municipal wastewater treatment plants, thereby eliminating the possibility of their use as a soil conditioner.

The removal of high concentrations of biodegradable organic matter at the source also is recommended, since it usually is much more cost-effective to remove these components, at least partially, when they are present in high concentrations.

The listed methods often are used in combinations of two or more steps to obtain the overall removal efficiency required by the most cost-moderate solution. Because wastewater treatment often is costly, it is recommended to examine all possible combinations of treatment options in the planning phase, in order to identify the most feasible and appropriate option.

Many existing municipal wastewater treatment plants were constructed years or decades ago, and may not meet today's higher standards. Nevertheless, upgrading existing wastewater treatment plants is possible, and may be more effective in some cases than building new ones (Novotny and Somlyódy, 1995; van Loosdrecht, 1998). Because the funding allocated to pollution abatement often is limited, the overall effect of upgrading wastewater treatment plants that can be upgraded with sufficient efficiency will be to the benefit of the environment. An attractive solution is often to introduce *tertiary treatment* by chemical precipitation and flocculation in an existing mechanical–biological treatment plant, with the addition of chemicals and flocculants before the primary sedimentation phase. The installation costs for this solution are minor, and the additional running costs are limited to the costs of chemicals. The result is a 85–95% removal of phosphorus at low cost. Similarly, nitrification and denitrification, ensuring a 80–85% removal of nitrogen, can be realized with the installation of additional capacity for biological treatment (the overall water retention time in the plant is increased by 4–12 hours, depending on the standards and composition of the wastewater), which is considerably less costly than construction of a completely new treatment plant. For details, see Hahn and Muller (1995).

The second question refers to the selection of the right standards for the treated wastewater. Any removal efficiency of any pertinent parameters ($BOD_5$, nutrients, bacteria, viruses, toxic organic chemicals, color, taste, heavy metals) is possible, using a suitable combination of the available treatment methods. However, what removal efficiencies are needed in the focal case? Because wastewater treatment is costly, the maximum allowable concentrations should not be set significantly lower than the lake or reservoir receiving the effluents can satisfactorily tolerate. The ban of phosphate detergents to decrease phosphorus concentrations in municipal wastewater treatment plant effluents is a point to consider. On the other hand, it might be even more expensive to install an insufficient treatment plant. Thus, the potential effects of a wide range of possible pollutant inputs on water quality and on

the entire lake or reservoir should be assessed, as the basis for selecting an acceptable option. This will require a quantification of the impacts of various possible pollutant inputs, considering a wide range of solutions. All processes and components affected significantly by the impacts should be included in the quantification. It usually is very helpful to develop a water quality/ecosystem model, and use it properly to assist in the selection of specific environmental treatment methods (see Section 5.1). It is important to emphasize that a model has an uncertainty in all its predictions that must be considered in making a final decision. Thus, it is essential to use safety factors to the benefit of the environment, in order to ensure that the selected treatment methods will have the anticipated effects. If the uncertainty is not taken into account for the sake of economy, as is unfortunately often done, the investment may be essentially wasted because the foreseen restoration of the lake or reservoir will not be realized, as discussed in more detail in Chapter 5.

Application of the methods identified in Tables 4.2 and 4.3 gives only approximate results, which should be used with caution. More details always will be required in reality. However, first estimates, such as those shown in the tables, are useful for evaluating alternative solutions to wastewater pollution problems. The biological treatment may either be an activated sludge plant or a trickling filter.

The cost of treating 100 $m^3$ of wastewater is also based on approximate indications, because they can vary from place to place, and are highly dependent on the size of the wastewater treatment plant. The costs are calculated as the running costs (electricity, labor, chemicals and maintenance), plus 10% of the investment to cover interest and annual appreciation. The annual water consumption of one person in an industrialized country corresponds to approximately 100 $m^3$.

A problem in many developing countries is the relatively high cost of wastewater treatment. Although this cost might justify diversion of the wastewater, the application of "soft technology"—"ecotechnology"—also should be considered. Some corresponding methods will be touched on in the following sections, but proper planning at an early phase, and consideration of all predictable problems, offers the widest range of cost-effective possibilities, and may allow prevention of the pollution problems before they can occur.

Corrections at a later stage, when pollution has already degraded the water quality and associated ecosystems, are possible, but will always be more expensive than the costs of proper wastewater treatment at an early stage. This is due in part to the fact that the accumulation of pollutants in a lake over time will always cause additional problems and, therefore, result in additional costs. Thus, pollution prevention at an early stage is better than curing pollution at a later stage. Removal of phosphorus from wastewater at an early stage, for example, is always beneficial, since the surplus phosphorus will accumulate in the lake sediments to a large extent, allow its remobilization back into the water column under certain chemical conditions in the waterbody.

Model studies are able to reveal how long it may require to restore a lake, or how much higher phosphorus removal efficiency will be required to compensate for each year that implementation of an appropriate phosphorus removal technology is postponed. However, it is not unusual that implementation of a phosphorus removal technology a few years later than when it was first feasible may delay the restoration of a lake by one or more

decades, due to the fact that the additional phosphorus accumulated in the sediments may significantly exceed the quantity of phosphorus in the water column.

Important pollution sources may be reduced if the liquid waste and sludge land disposal is avoided or minimized. For municipal areas, two options are used to decrease the hydraulic load of wastewater treatment plants, including (i) decreasing water use, thereby saving water and producing smaller volumes of polluted water, and (ii) separating storm water from municipal domestic waste, with a similar result. One result of these options is that the capacity of wastewater treatment plants can be kept smaller, achieving significant cost savings.

New approaches have emerged in regard to sustainable development. For instance, serious consideration is being given to separation toilets in some locations, which collect urine separately from faeces, thereby allowing utilization of the septic urine as fertilizer.

The selection of proper wastewater treatment methods for point sources of pollution is summarized in the following points:

- Develop models for the impacts of the wastewater on a lake or reservoir, considering the impacts on the water quality and the entire lake ecosystem.
- Apply the model to identify the maximum allowable pollutant concentration in the treated wastewater. Any uncertainty associated with the model predictions should be reflected in identifying the lower maximum allowable concentrations.
- Select the combination of available treatment methods able to meet the standards at the lowest costs without impacting the proper operation of the plant.
- If the investment needed for a proper solution to a problem cannot be provided, the application of cost-moderate technology that will reduce the accumulation of pollutants in the lake should be considered. Any measures taken at an early stage will likely reduce the costs at a later stage.

### *4.2.4 Natural and Constructed Wetlands*

*Natural wetlands*

The ecotones between lakes and terrestrial ecosystems are crucial for protecting lakes against anthropogenic impacts (Haycock et al., 1997). These transition areas have the same function for a lake as a membrane does for a cell; namely, they prevent the input of undesirable components into a lake. Thus, it is crucial to preserve the shore ecotones around a lake and wetlands in the catchment area, independent of the management strategy being implemented. Further, man-made constructions should be prohibited in a zone 50–100 meters from the lake shoreline to keep the ecotone area intact.

In addition to preserving lakeshore ecotones, there are many other approaches for enhancing vegetation growth along and in rivers and their channels (Klapper, 1991), in order to increase the transformation and absorption of pollution (Fig. 4.3A). Efficient pollution removal can be obtained by maintaining natural vegetation in rivers entering reservoirs, particularly in tropical regions. An example of the significant reduction of nitrate nitrogen to almost complete exhaustion by vegetation along the stream banks within a distance of 1.5 km is shown in Figure 4.3B. Both rooted and submerged macrophytes, including

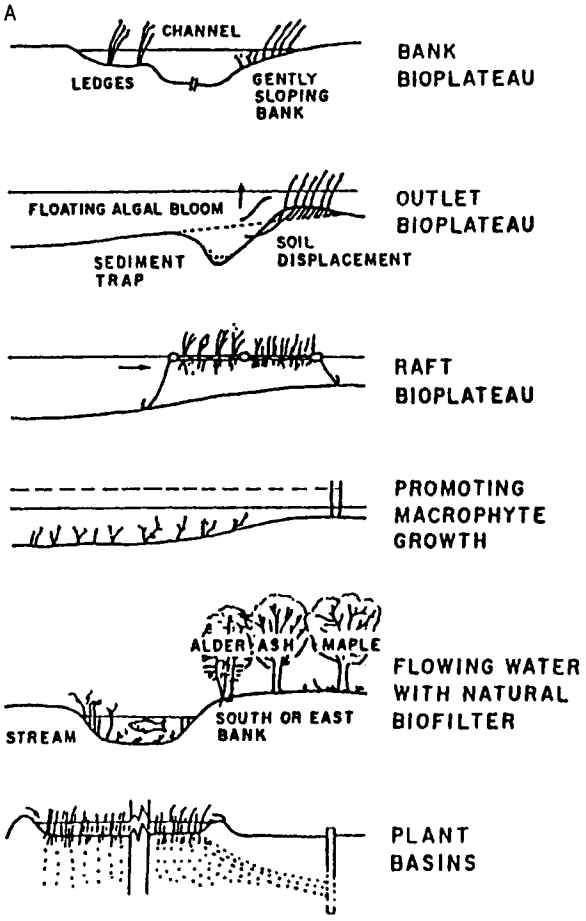



Fig. 4.3. Different forms of the application of vegetation for water quality protection and their efficiency. A—Approaches used (modified from Klapper, 1991).

the intensively-spreading tropical *Eichhornia crassipes* and *Pistia stratiotes*, are efficient removers. Artificial floating mats can be created when conditions for macrophytes are otherwise not adequate. These mats function as biofilters for pollution adsorption.

Although nonpoint or diffuse pollutants will inevitably flow toward a lake or reservoir, the land–lake transition zone can transform and/or adsorb the pollutants entirely or partially. Thus, it will significantly reduce the overall irreversible effects on the lake ecosystem. The most important processes can be summarized as follows:

- Nitrate is denitrified in the anaerobic conditions in the wetlands. Organic matter accumulated in the wetland converts nitrate to free nitrogen.
- Clay mineral is able to adsorb ammonium and metal ions.

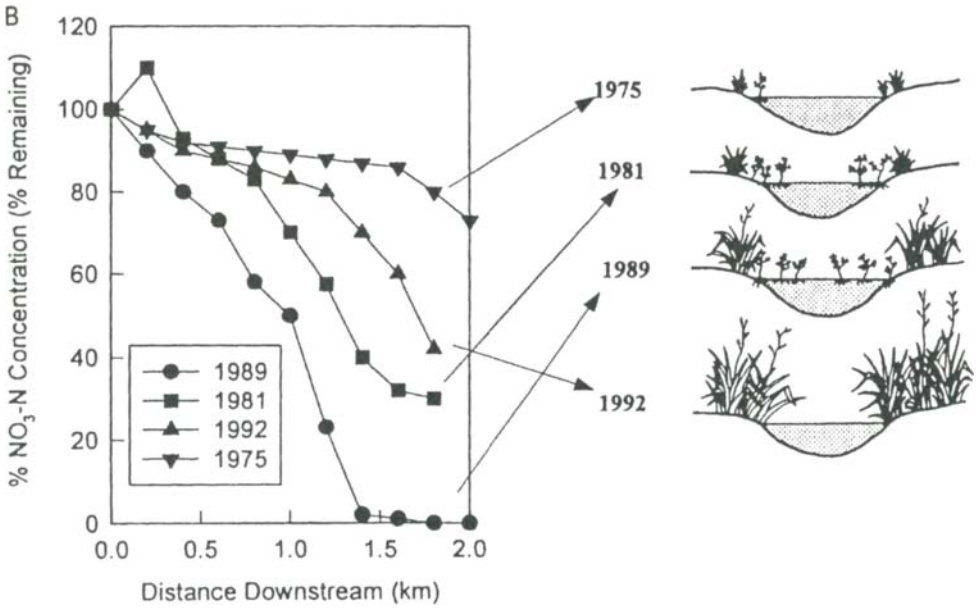



Fig. 4.3 (*continued*). B—Downstream changes in nitrate concentrations in the Whangamata stream shown with diagrammatic representation of changes in the streambank vegetation (from Downes et al., 1997).

- Organic matter is able to adsorb metal ions, pesticides and phosphorous compounds. Metal ions form complexes with humic acids and other polymer organics, which significantly reduce their toxicity.
- Biodegradable organic matter is decomposed aerobically or anaerobically by the microorganisms in the transition zone.
- Pathogens are out-competed by the natural microorganisms in the transition zone.
- Macrophytes are able to uptake heavy metals with a high efficiency and high specificity. Different plant species may take up different heavy metals, and this property can be utilized for removing heavy metals from contaminated land. Other toxic substances also may be removed by macrophytes, although it is difficult to provide any general rule for the removal efficiency.
- Toxic organic compounds will be decomposed by anaerobic processes in wetlands to a certain extent, depending on the biodegradability of the compounds and the water retention time in the wetland.

The denitrification potential of wetlands is often surprisingly high. As much as 2000–3000 kg of nitrate nitrogen can be denitrified annually per hectare of wetland area, depending on the hydraulic conditions (Jørgensen, 1994; Jørgensen et al., 1995). This is important for the protection of lakes, given that significant quantities of nitrate are released from agricultural areas. As much as 100 kg nitrate-N/ha may be found in drainage

water from intensive agricultural activities. Because denitrification is accompanied by a stoichiometric oxidation of organic matter, significant quantities of organic matter also are removed in this process. However, the phosphorus bound as organic matter, or adsorbed to the organic matter, may be released by these processes. Thus, these processes should be examined carefully and quantitatively in each case, including consideration of whether the released phosphorus will flow toward a lake or groundwater aquifer, and these possibilities should be considered in developing lake management strategies.

Figure 4.4 illustrates the mass balances of nutrients in a natural wetland adjacent to a lake, applied for nitrogen removal. The experimental wetland was a 0.5 hectare freshwater marsh located in front of a shallow lake, and receiving a significant load nitrogen from nonpoint agricultural sources. As can be seen, the denitrification capacity is high, but the process is accompanied by a release of phosphorus. Thus, it is important to ensure the released phosphorus is not significantly reducing the effects of the nitrate removal. In this specific case, it was advantageous that the phosphorus was not flowing directly to the lake, but rather toward ground water.

The adsorption capacity of the transition zone offers significant protection against pollution by toxic substances, including heavy metals and toxic organic chemicals (primarily pesticides originating from agricultural activities). The ratio concentration of heavy metals or pesticides in organic matter to its concentration in water at equilibrium is strongly dependent on the composition of the organic matter and the presence of complex forming ligands. It is usually between 50 and 5000, indicating that the transition zone has an enormous binding capacity for these pollutants.

Ecotones serve as a buffer zone not only for pollutants, but also for biological species present in adjacent ecosystems. Thus, preservation of wetlands at the lakeshore may be crucial for maintenance of the biodiversity in lake ecosystems—a function that a manager should not overlook in developing an appropriate lake management strategy.

Table 4.4 provides an overview of the different types of wetlands found adjacent to lakes, including wet meadows, forested wetlands, marshes, bogs and shore wetlands. The table also identifies the characteristics of these seven types of wetlands, and their differing ability to cope with nonpoint pollution problems.

The importance of wetlands located adjacent to lake ecosystems have provoked such attention that drainage of wetlands has ceased in many countries, and previously drained wetlands are restored in some locations (Salánki and Herodek, 1989; Herodek, 1990; Salánki and Biró, 1994). In accordance with U.S. legislation, wetland areas cannot be drained for other uses, unless another wetland of the same size is developed elsewhere.

*Constructed wetlands*

Artificial wetlands also can be constructed to cope with diffuse pollution originating from agriculture, septic tanks and other sources (examples are given in Mitsch, 1994).

As an example of wetland pollutant removal efficiency, Jørgensen and Jørgensen (1988) have compared the efficiency of the methods used to abate eutrophication for a lake in Denmark, using a eutrophication model. The result of their case study has shown that the use of wetlands is a very effective pollution control method. While most of the other control

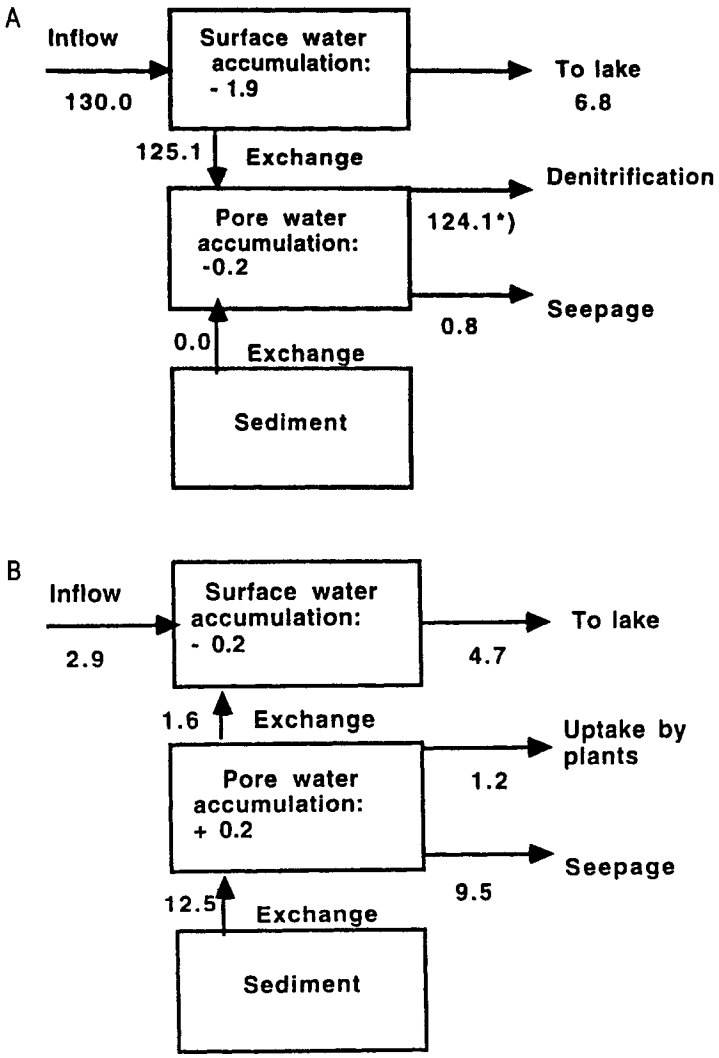



Fig. 4.4. Nitrogen and phosphorus budgets for a natural 5000 $m^2$ wetland adjacent to a Danish lake. A—Nitrogen budget in kg/day. Nitrite and nitrate nitrogen for period August 23–March 14. For denitrification, it is assumed that no nitrification takes place. If nitrification is considered, the value will correspond to 200 kg; B—Phosphorus budget in kg/day.

methods being considered did not reduce the phytoplankton concentrations and primary production more than 10%, constructing a wetland for nitrogen removal and partial phosphorus removal from the inflowing stream reduced the primary production and maximum

Table 4.4. Characteristics of wetlands adjacent to lakes (Patten, 1990)

| Wetland type | Characteristics | Ability to retain nonpoint pollutants |
|---|---|---|
| Wet meadows | Grassland with water-logged soil; standing water for a part of the year | Denitrification only in the presence of standing water; removal of N and P possible by harvest |
| Freshwater marshes | Reed-grass dominated, often with peat accumulation | High potential for denitrification, which is limited by the hydraulic conductivity |
| Forested wetlands | Dominated by trees and shrubs; standing water not always for the entire year | High potential for denitrification and accumulation of pollutants, provided that standing water is present |
| Salt water marshes | Herbaceous vegetation usually with mineral soil substrate | Medium potential for denitrification Harvest possible |
| Bogs | A peat-accumulating wetland with minor flows | High potential for denitrification but limited by the small hydraulic conductivity |
| Shore wetlands | Littoral vegetation, often of great importance for the lake | High potential for denitrification and accumulation of pollutants, but area coverage variable |

phytoplankton concentration by about 33%, thereby also increasing the water transparency from 0.6 m to 1.0 m.

Constructed wetlands may have either surface flow or sub-surface flows, with the latter sometimes called root zone plants.

Construction of artificial wetlands is an attractive, cost-moderate solution for nonpoint source pollution (Hammer, 1989; Cooper and Findlater, 1990; Moshiri, 1993; Vymazal et al., 1997). Although artificial wetlands are generally able to cope with nitrogen and heavy metal pollution from these sources, it is essential to properly plan where to locate them, since their efficiency is dependent on the hydrology and landscape patterns. They should remain covered by water most of the year, and have a sufficient water retention time to allow treatment of the pollution problems, thereby also protecting the most vulnerable ecosystems, which often are lakes and reservoirs. It also is important to ensure that the wetlands are not releasing other components (e.g., phosphorus; see the results from the experimental wetland, mentioned above).

It is noted that, in most cases, a wetland will reduce the water budget due to evapotranspiration. However, wetlands also reduce the wind speed at the water surface, thereby also perhaps reducing the evaporation rate. It is important to consider these factors during the planning phase for artificial wetlands. Finally, it should not be forgotten that full development of an artificial wetland may require 2–4 years, in order to obtain sufficient plant coverage and biodiversity to be fully operational. It is clear from past experiences that the application of models for wetlands encompassing all the processes reviewed above, as well

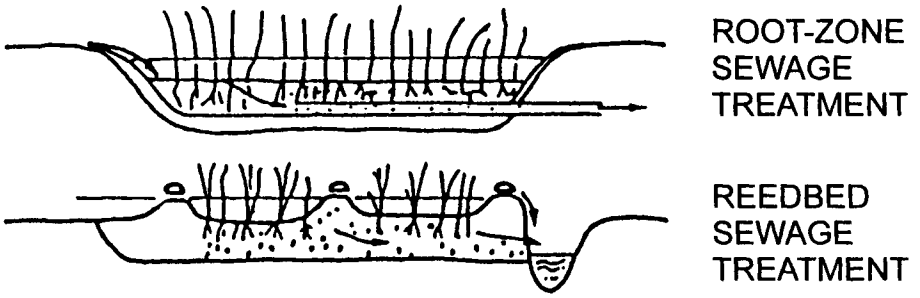



Fig. 4.5. Constructed wetlands can be distinguished into root-zone sewage treatment plants (sub-surface flow) and reedbed sewage treatment plants (surface flow) (modified from Klapper, 1991).

as for the lake, is compulsory if positive results are anticipated. Artificial wetlands are basically of two types; namely, root-zone plants (sub-surface flow) and reedbed plants (surface flow) (Fig. 4.5).

Wetlands encompassing the so-called root-zone plants also may be utilized as wastewater treatment facilities. This application of "soft technology" seems particularly advantageous for developing countries because of its moderate cost.

The self-purification ability of wetlands has found wide application as a wastewater treatment method in several developing countries (China, Philippines, Burma, India, Tanzania, Thailand).

Much attention also has been devoted to stocking fish cultivated in biological sewage stabilization ponds. These studies show that the nitrogen and phosphorus are retrieved from the sewage by fish, through the intermediate activities of bacteria, algae and other types of plankton. In order to mobilize the self-purification capability of a waterbody for sewage treatment, the main principle is to utilize the coordination between four types of primary components: producers, aquatic vegetation, consumers (including fish), decomposers (mainly bacteria) and abiotic factors (solar radiation, water exchange, etc.).

Bacteria decompose organic materials, including organophosphates, and purify water in a preliminary phase. Carbon dioxide, ammonia, phosphates are produced and utilized by algae and other aquatic vegetation. The oxygen released by photosynthesis is available to meet the needs of oxidizing organic matter by bacteria. A large quantity of algae and other types of plankton are utilized by rearing fries. The energy fixed by algae finally appears in the form of fish harvesting, with the water being further purified.

Waste stabilization ponds without fish cultures also are widely used, particularly in many African countries. This solution is cost moderate, being about US \$2–3/100 $m^3$ of water treated, including the operation costs, interest and appreciation. The typical system consists of three types of ponds working in series: anaerobic ponds, facultative ponds, and aerobic ponds. Unfortunately, many stabilization ponds are not operating properly due to pollutant overloading, inadequate maintenance, or use of incorrect design criteria. The systems often are designed by European or North American consulting engineering firms,

which do not take the local socioeconomic and climatic conditions into account. The experience of DANIDA in East Africa has shown that application of ecological engineering approaches, in combination with an ecological model of the stabilization ponds, may yield a better design with a higher certainty of proper operation, provided that a proper maintenance scheme is applied. The produced plants may be used as fodder for domestic animals.

Wastewater treatment using water hyacinths (*Eichhornia crassipes*) has been applied as an alternative to waste stabilization ponds. The inorganic nitrogen and phosphorus from the sewage and the decomposition of organic pollutants by microorganisms are absorbed by the water hyacinths as a nutrient source. Experiments in China have shown that an average water hyacinth yield of about 10 kg $m^{-2}$ may be obtained during the growing period (May–November). This level of production can absorb 1500 kg of nitrogen, 350 kg of phosphorus and 200 kg of sulfur per hectare. The water hyacinths, with the microorganisms and organic pollutants attached or coagulated on the root surfaces, are harvested as food for fish culture ponds, duck farms and pig farms. From May to November, the concentrations of chemical oxygen demand (COD), total nitrogen, ammonium, total phosphorus and orthophosphate were less than half the concentrations in the inlet. In contrast, when the water hyacinths were absent (December–April), the differences in COD, nitrogen and phosphorus concentrations between inlet and outlet were very small (Ma and Yan, 1989).

Many heavy metals are concentrated and accumulated in water hyacinths from very low concentrations in water. However, the heavy metal enrichment in this plant varies with the aquatic habitat. Water hyacinths with high residual amounts of heavy metals obviously cannot be used as fodder, limiting this ecological engineering approach for treatment of water polluted by organic pollutants (mainly municipal wastewater).

The root zone plant has found its application in treatment of small volumes of municipal wastewater in industrialized countries, particularly where construction of a sewage system to an adjacent wastewater treatment plant would be prohibitively expensive. The decomposition of organic matter and denitrification usually does not cause any problems, provided the plant has 5–10 $m^2$/person equivalent, dependent on the climatic conditions. Phosphorus removal is only about 10–20% by a root zone plant, although the addition of iron chloride may increase the efficiency to 80% or more, due to precipitation of iron phosphate. The application of this method seems attractive for recreational areas, where the population density is low, but the wastewater loading has a significant impact.

The recommended management of natural or artificial wetlands and of lakeshore ecotones may be summarized as follows:

- Maintenance and preservation of the transition zone between the terrestrial ecosystems and the lake ecosystem should be compulsory. The ecotone between the two types of ecosystems functions as a buffer zone for preservation of species diversity in both ecosystems.
- The different types of wetlands forming the transition zone have a high adsorption capacity for many pollutants, such as heavy metals and toxic organic compounds.
- The different types of wetlands forming the transition zone are able to denitrify up to several tons of nitrate nitrogen per hectare annually.

- Recovery and construction of artificial wetlands are crucial abatement methods for treating diffuse pollutants originating from agricultural activities.
- Application of artificial wetlands, including waste stabilization ponds and root zone plants, may be attractive wastewater treatment methods for developing countries, for recreational areas adjacent to a lake ecosystems, and generally for areas with low population densities.

The application of a quantitatively-based management scheme of all the identified wetland types, including development of models for these ecosystems, is recommended.

## 4.3 IN-LAKE METHODS

The approaches that can be used to manage water quality within a lake include mechanical, chemical and biological methods. The useful methods are those that are comparatively inexpensive, and that produce the smaller impacts on the total lake environment. These include mainly methods based on ecological knowledge, such as regulating fish populations by using biomanipulation, etc. Further, using a combination of methods is probably the best overall approach, utilizing the partial capabilities of the individual methods for the greatest overall success.

### *4.3.1 Biomanipulation*

Biomanipulation is a term coined by Shapiro et al. (1975) for various lake water quality management methods based on biological interventions. This term is usually restricted to one particular method, based on manipulating the fish populations in a lake so as to change algal biomass. The theoretical basis of this method was established during the 1950s and 1960s by Hrbácek et al. (1961, 1986), Hrbácek (1962), Brooks and Dodson (1965) and Hall et al. (1970). Hrbácek demonstrated that fish stock levels in ponds influenced the level of primary production and nutrient cycling via the herbivorous grazing by zooplankton. He demonstrated that when overpopulated planktivorous fish are eradicated, the waterbody changed from having high algal concentrations and low transparency to a condition with less algae and increased transparency. Brooks and Dodson (1965) demonstrated a change in the zoo- and phytoplankton composition, and an increase of phytoplankton density, when a herring-type zooplankton feeding fish was introduced into a lake in the United States (Fig. 4.6).

Shapiro et al. (1975) and Shapiro (1978, 1995) elaborated the idea by manipulating piscivorous fish in food webs to produce a desirable pelagic community structure within the ecosystem, favoring low algal levels. Comparative studies on several reservoirs in the Czech Republic (Hrbácek et al., 1986) and in England (Duncan, 1990; Duncan et al., 1991; also see Section 9.6) have shown that, for the same nutrient level, the algal crops can be reduced by up to by 50% with this method. Reviews of all aspects of biomanipulation are provided by Hosper (1989), Gulati et al. (1990), Reynolds (1994), DeBernardi and

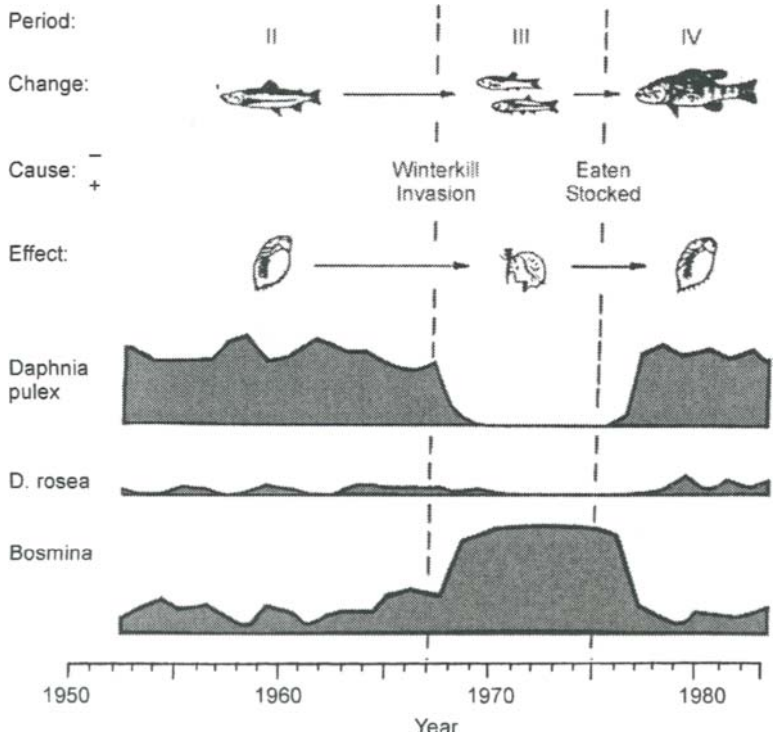



Fig. 4.6. Results of biomanipulation experiment in a natural lake. After winter kill of predators and invasion of the zooplankton feeding *Alosa*, the zooplankton population changed from larger *Daphnia* species to small *Bosmina*; after introduction of invasive predator, the large species capable of controlling phytoplankton were rapidly reestablished (modified from Brooks and Dodson, 1965).

Giussani (1995) and Shapiro (1995). The use of biomanipulation in shallow lakes is discussed by Perrow et al. (1997). The reviews generally show that if the method is to be successful, specific conditions have to be fulfilled.

Biomanipulation is based on what is called in contemporary limnology *top down control* (Fig. 4.7). The primary control is the natural, internal control within the ecosystem, based on mutual interactions between the abiotic variables and different organisms within an aquatic ecosystem. The type of control discussed in classical limnology was a *bottom-up control*, based on the idea that abiotic variables such as light, temperature and nutrients affect the most elemental steps in the aquatic production process; namely, the algae and aquatic macrophytes. The control then moves from the lower to increasingly higher levels of trophic organization, from the phytoplankton to the organisms that feed on it; namely, the zooplankton. The zooplankton is the decisive element for the growth of zooplankton-feeding fish, which are in turn preyed upon by predatory fish. The importance of the top

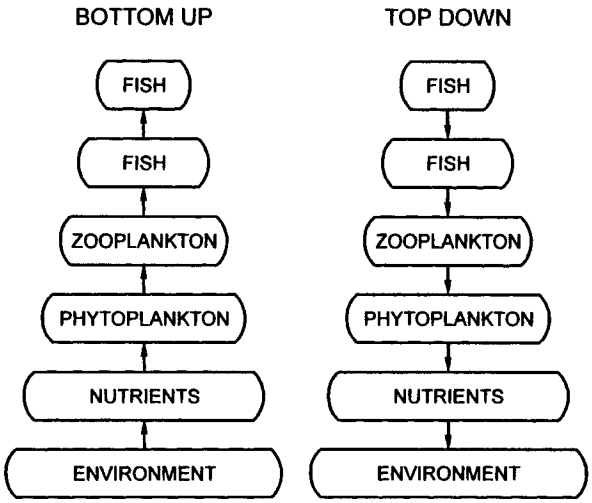



Fig. 4.7. Schema of bottom-up and top-down control in aquatic ecosystems. Both controls are realized simultaneously, with the effect of each one prevailing under certain conditions. (Orig.)

down control (i.e., the effects that predators have on the development of zooplankton and the zooplankton control of phytoplankton development) was only recognized more recently. It is now known that both types of control are simultaneous, and that there are feedback relations between the abiotic environment and its various biotic components.

The top down component of the natural, internal control of an aquatic ecosystem is so strong that some typically physical variables, including temperature, have been determined to be controlled by the biota. The effect in this case is due to adsorption of sunlight radiation by microscopic algae in the upper water layers. When dense algal populations exist in a waterbody, heat is adsorbed in its surface layers, warming the water. In contrast, less heat penetrates to the deeper water layers, making them colder than if the waterbody had less concentrated algal populations.

The principle of biomanipulation rests in food chain manipulation, by reducing the feeding pressure of fish on zooplankton. As a result, large species of zooplankton predominate, which are capable of keeping the phytoplankton levels down. The two typical states existing before and after biomanipulation are shown in Figure 4.8.

The feeding pressure of fish on zooplankton is high when the number of zooplankton-feeding fish is high. This situation is common in lakes and reservoirs with stunted fish populations (i.e., overcrowded fish populations which only grow slowly). Because of their numbers and small size (small fish have relatively greater food requirements), however, they decimate larger zooplankton species, and the smaller zooplankton are unable to control algae populations. The distribution of energy and dominant flow pathways in these two situations is very different.

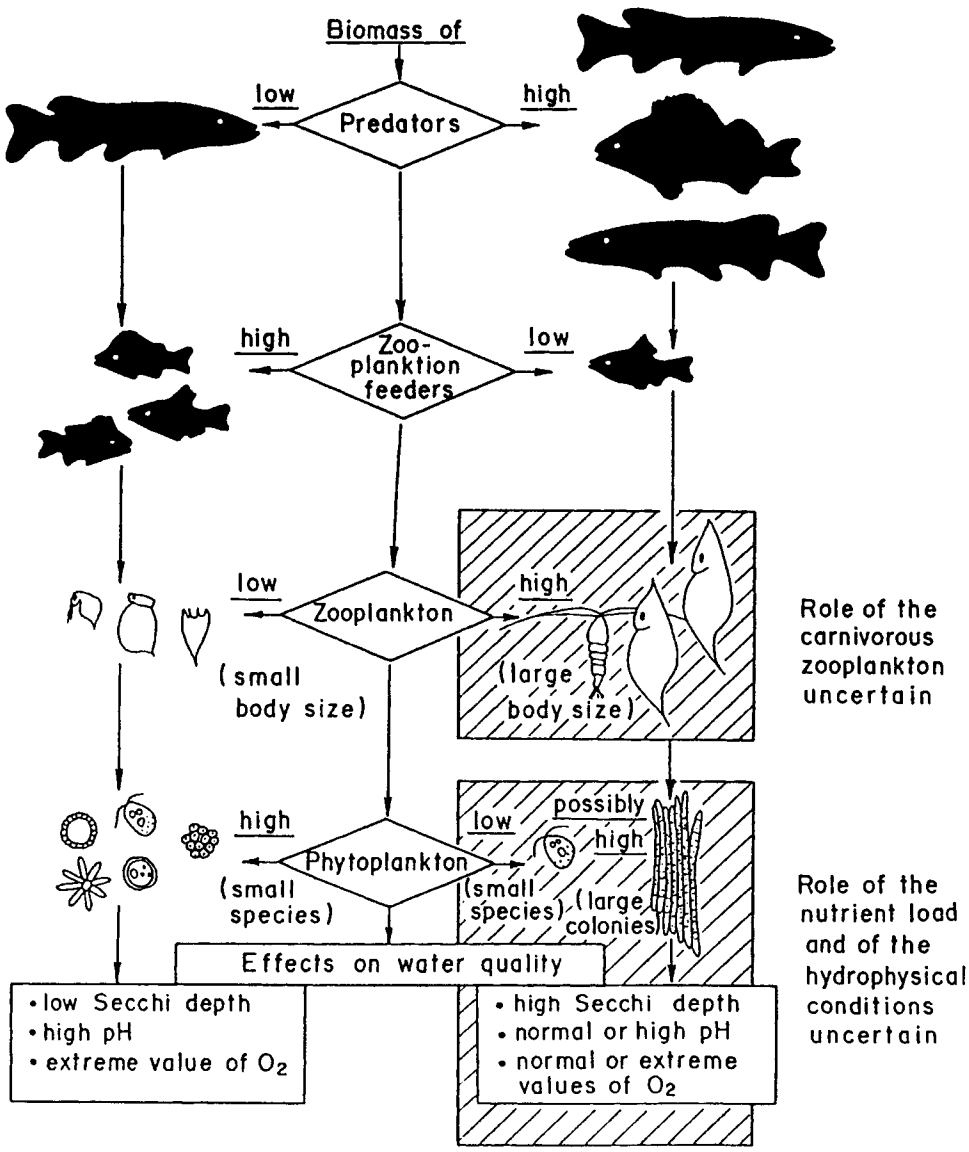



Fig. 4.8. Schematic representation of two situations in European lakes/reservoirs, with (i) domination of planktivorous fish when small zooplankton are unable to control phytoplankton development and with (ii) domination of predatory fish which reduce zooplanktivorous fish and, therefore, enhance large zooplankton capable of significantly reducing phytoplankton population. At high phosphorus concentrations, however, a switch to large phytoplankton inedible by zooplankton or to macrophytes is possible (modified from Benndorf et al., 1994).

*Procedures used in temperate Northern Hemisphere lakes*

*Deep lakes.* A low feeding pressure of fish on zooplankton is obtained when a "healthy" composition of fish stock is present. A healthy fish stock means well-developed predator populations and reduced numbers of well growing noncarnivorous fish.

The composition of fish populations that result in the control of phytoplankton by zooplankton can be achieved in two ways:

- Temporarily, by eradication of stunted fish populations with rotenone poisoning and predator stocking (rotenone is not toxic to invertebrates and phytoplankton; Stenson, 1972).
- Continuously, by introducing predatory fish and net harvesting of nonpredatory fish. A collaboration with the local sport fishery and the use of commercial fishery methods is needed. Drawdown of the reservoir water level during the reproduction period of undesirable fish species laying eggs on shore vegetation is a possibility to reduce the abundance of zooplanctivorous fish.

Winter fish kills in some freezing reservoirs can also be a good starting point for modifying species composition. The methods used for fish removal in England for purposes of biomanipulation are identified in Table 4.5.

Most effective examples of biomanipulation apply to relatively small waterbodies (Leventer, 1979) because of the greater difficulty of continuously manipulating fish populations in large ones. There are not many published examples for larger waterbodies such as reservoirs (Benndorf, 1987, 1988; Benndorf et al., 1988; Hrbácek et al., 1986; Seda et al., 1989; Duncan, 1990; Zalewski et al., 1990; Zalewski, 1994), and the results are usually not as convincing as those from smaller, but deep waterbodies. However, the results from the

Table 4.5. Removal techniques used in fish removal in shallow lakes in England (modified from Moss et al., 1996)

| |
|---|
| *Systematic electrofishing*—This has been done from a rowboat, with a rower and operator |
| *Electrofishing for isolated large fish or small shoals of such fish*—The fish are chased in calm, clear water with a powered boat, with the cathode tied across the bottom of the boat and the anode applied to the bow |
| *Seine netting*—Nets from 50 to 120 m have been used for encircling concentrations of fish. A floating mobile pontoon is used |
| *Isolating nets*—Light weight 60 m monofilament nets were used, paid out from the back of a rowboat to surround shoals and concentrate fish for removal by electrofishing |
| *Passive fish traps (fyke nets)*—The nets used conventionally by eel fisherman are set for periods of 12–24 hours to catch large fish which have evaded other methods |
| *Scare lines and fish traps*—Ropes onto which brightly colored scraps of cloth are towed through the water by boats. This herds fish to traps made from stop nets, from which the fish are removed by seining |
| *Prevention of successful spawning*—Bream, roach and tench, and perhaps other fish, will readily spawn onto netting if this is placed in traditional spawning areas. The netting can then be removed to dry |

highly eutrophic London reservoirs (Duncan, 1990; Duncan et al., 1991) are very convincing, with the fish biomass being very low (5–37 kg/hectare) for about 20 years or more, with grazer control of algal populations being dominant throughout the summer, clearing the water and reducing subsequent treatment costs.

*Shallow lakes.* Willemsen (1980) observed two states in shallow waterbodies in The Netherlands: one with clear water and macrophytes, and the fish fauna dominated by pike, and one with turbid water, no macrophytes present and fish fauna dominated by bream. A mathematical model constructed to clarify the conditions needed to switch between the two states is discussed in Chapter 5. The model shows the importance of the critical nutrient level for successful biomanipulation, measured as a substantial decrease of phytoplankton and increased transparency.

The principal objective of biomanipulation in shallow lakes is to generate a sufficiently-long period of clear water to allow macrophytes to become established. To achieve this goal, and for other technical reasons, biomanipulation is best undertaken in the winter and early spring, in order to generate clear water as early as possible in the season. The biomanipulation activity may have to be repeated if macrophytes do not colonize effectively within the first season. Understanding the nature of the factors and mechanisms responsible for turbid water preventing macrophytes growth is critical if biomanipulation of shallow lake is to be successful. If sufficient information is available, particular components of the fish community may be targeted and precise figures for exceeding critical threshold values may be set. In the absence of this information, a safe strategy is to remove at least 75% of the fish. Stocking with piscivores is a useful additional measure of fish removal (Perrow et al., 1997). A practical decision tree for biomanipulation of shallow flood-plain lakes in England was elaborated by Moss et al. (1996).

*Advantages and drawbacks.* One of the most extensive data set demonstrating the positive effect of balanced fish populations on low algal biomass is from Quiros (1995), who investigated the effect of fish assemblage composition on lake water quality in 110 lakes in Argentina (between 25–55.8°S latitude). He demonstrated that, at comparable total phosphorus concentrations, lakes with planktivores, but without piscivores, had the highest phytoplankton biomass and lowest water clarity. The average chlorophyll-a concentration was 40.6 mg $m^{-3}$ for these lakes, while the chlorophyll-a concentration of lakes with both planktivores and planktivores was only 26.3 mg $m^{-3}$, a 65% reduction. A thorough evaluation of the biomanipulation experiments published through 1992 by Kasprzak (1995) and Jørgensen and de Bernardi (1998), comprising 22 whole lake experiments and 10 enclosure experiments are summarized below. With one exception, all were performed in temperate regions, mostly in stratified lakes:

- A success, measured as noted above, was achieved in two-thirds of the 25 observations; in 28%, the result can be described as ambiguous, while a disappointing result was obtained in only two cases.
- The disappointing results are due to too-high initial total phosphorus concentrations in the waterbody. It can be shown that the two possible structures (domination by phytoplankton and planktivorous fish, and domination by zooplankton and top-carnivorous

fish) are both stable at a total phosphorus concentration between about 60–125 $\mu g\ l^{-1}$. Thus, any improvement with biomanipulation at total phosphorus concentration above 120 $\mu g\ l^{-1}$ will only be temporary. The domination by phytoplankton and planktivorous fish will eventually occur at total phosphorus concentrations above 125 $\mu g\ l^{-1}$ (also see structurally-dynamic modelling discussion in Chapter 5).

- Poor results were observed with very high phosphorus concentrations. In very eutrophic waterbodies, therefore, it is necessary to combine biomanipulation with nutrient reduction by other methods.
- Biomanipulation success is generally more pronounced in smaller, shallow waterbodies, particularly because the control of fish populations is easier under such conditions.
- A shallow waterbody, or shallower areas of deeper ones, may switch to a macrophyte dominated state, thereby dictating the consequences for the respective use of the waterbody.
- The creation and stabilization of a strong population of piscivorous, predatory fish populations is difficult and time consuming, unless achieved by commercial net fishery. The use of rotenone poisoning is often the only shock sufficiently strong to trigger the conversion.
- The extent to which planktivorous fish must be removed depends largely on the species and size composition of the fish community.
- Due to very different turnover times of the organisms, a new, stable equilibrium may require several years to develop. However, this is also true for other eutrophication processes, such as nutrient input reduction when the high internal loading is capable of extending the duration of the eutrophic state for a number of years.
- The biomanipulation procedure cannot be considered a routine method, since it depends on a number of circumstances and can be performed only with the participation of skilled limnologists.
- Local and waterbody-type specific criteria for successful application of biomanipulation must be used.

As stressed above, the present experience is based on temperate conditions. However, it focuses primarily on the northern part of the temperate region, with relatively uniform species composition of fish and zooplankton fauna.

*Application of biomanipulation in subtropical and tropical regions.* Lazzaro et al. (1992) provide a discussion of the application of biomanipulation in the southern part of the United States. The fish at the top of the food chain in this situation is the omnivorous gizzard shad (*Dorosoma vittatus*), which is able to switch its food source from zooplankton to bottom detritus, thereby not exerting efficient control of zooplankton. Further, high fecundity and development in early years allows it to successfully out-compete other predatory and zooplankton-feeding species.

Some studies on the application of biomanipulation in subtropical and tropical regions do exist. Straškraba et al. (1969) reported on the results of a rotenone poisoning experiment in a small Cuban lake. Although the lake was only observed for 4 months, the phyto-

plankton decreased and the transparency increased. Arcifa et al. (1986) and Starling et al. (1998) made short term, enclosure experiments in the Paranoa Reservoir in Brazil, a very eutrophic reservoir located at a latitude of 15°S in Brasilia, the capital of Brazil. The total phosphorus concentration was about 190 mg $m^{-3}$ at the time of the experiment. The addition of different numbers of planktivores (*Hypophthalmichthys molitrix*, silver carp) to the enclosures resulted in changes to the zooplankton and phytoplankton composition. Although the experiments in the Broa Reservoir (22°S, elevation 800 meters) by Roche et al. (1993) show conditions more complicated than those seen in the North, the trends were generally similar. Magadza (1995) and McQueen (1997) discuss other applications of this technique under subtropical and tropical conditions.

Successful applications and experiments, using the silver carp (*Hypophthalmichthys molitrix*) have been reported for lakes in Israel and Brazil (Leventer, 1979; Leventer and Teltsch, 1990). The food web effect is different from that seen in Figures 4.7 and 4.8, due to the feeding of grass carp directly on large colonies of phytoplankton and macrophytes. However, as in all other cases of fish introductions, caution must be used in introducing foreign species into a new waterbody. In fact, in Israel, the introduction of silver carp also affected the populations of other species (Gophen, 1995).

Applications in subtropical and tropical regions are more complicated because of high fish diversity, changing species composition from place to place, the presence of omnivorous fish and more complicated food chains (Arcifa and Northcote, 1997; Lazarro et al., 1992; Lazarro, 1997). More knowledge of the food webs in such systems is needed. The more diversified fish fauna and its trophic relations must be known for each lake situation. The higher reproduction rates of fish, zooplankton and phytoplankton will need greater control efforts to keep the structure of these populations optimal from the perspective of phytoplankton reduction. More knowledge on the zooplankton feeding of tropical zooplankton also is needed. On the other hand, a positive feature might be the greater interest in fish as a food source, and the existing skill of fisherman to use the net fishery. In temperate climates, the sport fishery concentrates on selective elimination of piscivores. However, from the perspective of biomanipulation, the presence of piscivores should be facilitated.

The *advantage* of biomanipulation is that it is completely natural in its components, with no chemicals or machinery needed. It only requires manpower to implement. Further, it also can combine both the need to protect water quality and maintain fisheries. Education of fisherman is often needed to convince them not to remove predators, and not to stock a lake with nonpredatory species. The *drawback* is the need of continuous control of fish populations, which tend to return to the stunted population state not because of natural processes, but rather because of recreation sport fisheries concerned with predatory, rather than zooplankton-feeding fish. Biomanipulation is one method with no significant potential *negative impacts* if performed without rotenone poisoning. Rotenone poisoning is not desirable in drinking water reservoirs and may result in the killing of rare and endangered species. The *cost* depends on the way in which the method is performed. It is relatively low, particularly if done in combination with organized fishery, but will increase in cost when these elements are not combined. Further, the cost of rotenone is quite high.

### *4.3.2 Mixing and Oxygenation*

Methods discussed in this section are some of the most commonly used for lakes and reservoirs. Some belong to the category of preventive means, while others are used in a curative way to get rid of the water quality problems that have reached a critical stage.

Artificial mixing is used either to oxygenate a deoxygenated hypolimnion, or to inhibit phytoplankton growth. As shown below, seven mixing methods exist, with the four most common ones presented schematically in Figure 4.9:

- Destratification by total mixing of the water column,
- Mixing and re-aeration of the hypolimnion,

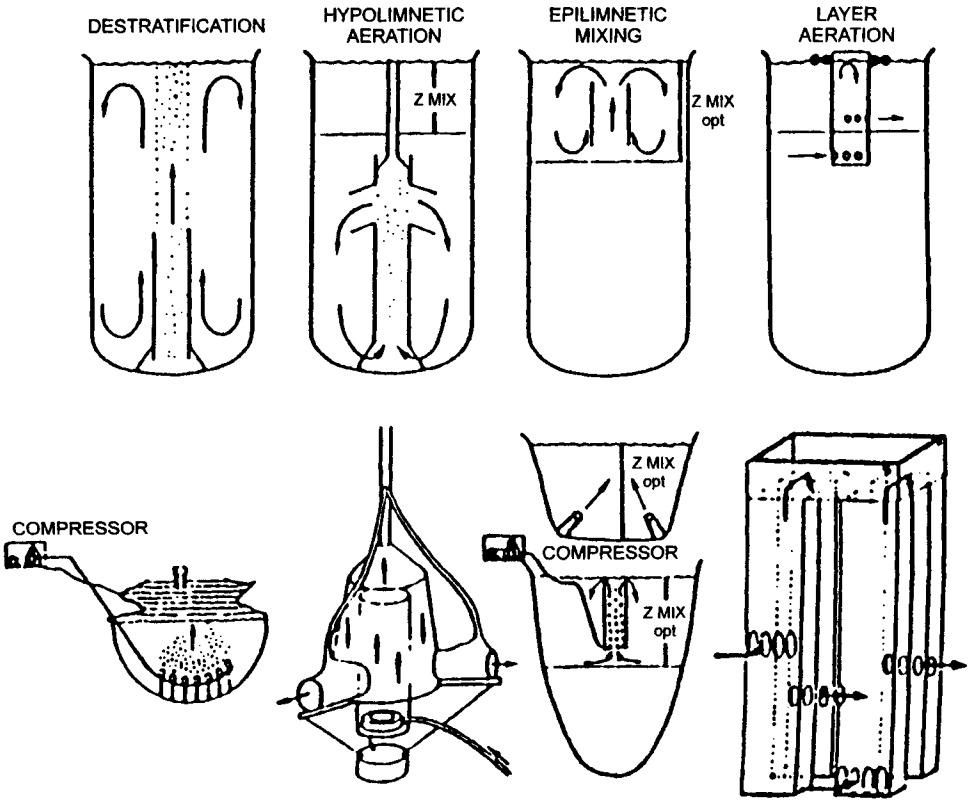



Fig. 4.9. Schematic representation of the four most common mixing types. The top row shows the mixing, the bottom row the actual arrangement. From left to right: destratification with compressed air with an aerator or with tubing laid on the bottom, hypolimnetic aeration (the partial air-lift hypolimnetic aerator LIMNO of the firm Aqua Techniques), epilimnetic mixing in two versions—in an embankment reservoir and a deep lake or valley reservoir, and the layer aerator of Kortmann (modified from Straškraba and Tundisi, 1999).

- Epilimnetic mixing,
- Layer aeration,
- Hypolimnetic oxygen aeration,
- Metalimnetic mixing,
- Propeller mixing.

The use of these different approaches depends on the type of problem encountered and on the treatment possibilities. Destratification is the simplest method, being used with varying success. As will be shown, unsuccessful use of this method may have two reasons, including inadequate mixing and its use under conditions that do not consider the limits of the application of this method. Hypolimnetic mixing and re-aeration is the most commonly used approach, avoiding some of the negative effects and limitations of destratification. In contrast to the previous methods, epilimnetic mixing is curative in nature. Curative methods are advantageous from the ecotechnology perspective and, therefore, highly recommended. Metalimnetic mixing is the newest method, designed for conditions when a lake has developed specific layers of low water quality. The Speece cone used for hypolimnetic oxygenation is a technically elaborate and expensive method. Only limited information exists regarding the newest approach of using surface propellers.

*Destratification–artificial circulation–aeration*

The name of this procedure highlights its purpose. Lake stratification is destroyed and lake circulation is restored. The theory is explained in Kortmann et al. (1994). In shallower, smaller lakes, it is done by injection of compressed air from a diffuser to the water at the lake bottom (Fig. 4.10). When aeration is the main goal, this simple bubble plume diffuser was found to be the simplest, most efficient arrangement, even for a deep lake (McGinnis et al., 1998). In larger, deeper waterbodies, the stream of bubbles must be directed by a broad tube of 2–3 meters in diameter to create in-lake circulation. The lake bottom water is enriched by oxygen, algae are mixed to greater depths with a subsequent decrease in their net production, the pH of bottom water is increased, and that of the surface layers decreased. Three goals are simultaneously being sought:

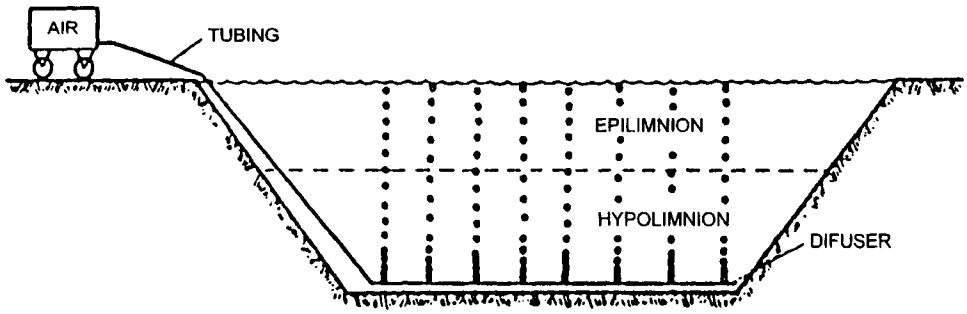



Fig. 4.10. Typical arrangements for lake mixing in smaller lakes (modified from Straškraba and Tundisi, 1999).

- The goal of destratification is to prevent algae from remaining in the light-illuminated water layer, with a resultant decrease in phytoplankton biomass formation.
- The goal of circulation is to decrease the pH, and cause a shift from blue-green algae to less noxious green algae.
- The goal of aeration is to oxidize the hypolimnion, with the consequence of preventing the release of phosphorus, iron and manganese.

There may be an environmental price to pay, however, because the process of lifting the hypolimnetic water to the lake surface may also result in higher phosphorus concentrations and increased algal growth. Lorenzen and Mitchel (1973) determined the circumstances under which the desirable effects of destratification were to be expected. Earlier examples of the practical use of this technique, made without regard to the Lorenzen and Mitchel rules, are summarized by Pastorok et al. (1980). Based on 40 attempts, they claim a significant change in algal biomass in 65% and a decrease in 70%, and an increase accompanied by changes in species composition in 30%. Raman (1988) provides practical examples for twelve lakes in Illinois. A good example of the successful use of destratification in tropical Malaysia is provided by Kassim et al. (1997).

Model studies by Schladow (1992), Wüest et al. (1992), Schladow and Fisher (1995) and McGinnis et al. (1998) showed the realistic shape of the rising plumes of compressed air, allowing it to calculate the necessary forces to efficiently circulate the whole water column. Schladow's results indicated that some of the earlier attempts were insufficiently designed to adequately mix the entire water column. Thus, it is necessary to stick to the rules of Lorenzen and Mitchel (1973) and Schladow (1993) for successful application of this technique. Model studies by Imtaez and Aseada (2000) indicated that water quality improvement can be obtained by using the appropriate gas flow rate and number of bubble ports, and applying it at an appropriate period of the year.

In addition to the review by Pastorok et al. (1980), Burns and Powling (1981) and Burns (1990) also reported examples from Australia with *negative impacts*. It is not possible to estimate how much of the negative effects were due either to insufficient mixing, or to the increasing surface concentrations of phosphorus resulting from the mixing (i.e., lifting the phosphorus from deeper water layers to the surface). In a case described by Fast and Hulquist (1982), compressed air used to destratify a lake caused supersaturation of the water with dissolved nitrogen and causing downstream fish kills. In smaller remote lakes in Australia with no electricity supply, Burns (1998) has used solar batteries to run the air compressor. The *cost* of this procedure is low, equaling the price of the compressor and the installation of the pipe and diffuser. If applied following the conditions given by Lorenzen and Mitchel (1973), and the criteria by Schladow (1993), the method has the following *advantages*:

- Hypolimnetic oxygen increases,
- No increased phosphorus release from the sediments,
- The quantities of iron and manganese remain low or absent,
- A decreasing quantity of algae.

New types of aeration systems for efficient mixing and aeration of shallower waters are produced by various firms; namely, a system with hollow fiber membranes (Weiss et al.,

1996), the NOPOL aeration system (Anonymous, 1997). Aeration may be also done by propellers with air injections, as shown below in the discussion on propeller mixing.

A method based on particular knowledge of limnology and algae was theoretically developed by Reynolds et al. (1984) and applied practically in a German lake by Steinberg and Zimmermann (1988). The water column is intermittently destratified in a way so as to minimize the growth and maximize the losses of different algae. Cyanobacteria and algae were highly reduced with the application of this method, with the instantaneous development of different species in the lake.

*Hypolimnetic aeration*

There are aerators which can oxidize the hypolimnion without breaking down the thermocline (e.g., Bernhardt and Hotter, 1967; Fast et al., 1975; Anonymous, 1985; McQueen and Lean, 1986; Bernhardt, 1987; Prien and Bernhardt, 1989; Anonymous, 1989; Little, 1995; McGinnis and Little, 1999; McGinnis et al., 1998). The study by Burris and Little (1998), using a mathematical model, showed that for one of the devices, it is possible to double the oxygen transfer to water in hypolimnetic aerators when the initial bubble diameter is reduced from the more standard 5 mm to just 2.5 mm. More generalized results to predict efficient bubble diameters for different devices and conditions are underway (Little and Del Vecchio, 1999).

The principle is evident in Figure 4.11, which shows one of the aerator types. The air bubbles rising from the diffusers at the lake bottom are allowed to reach the surface in a narrow aeration tube, while water is pumped to be released at a predestined depth. This corrective technique is used in instances of high hypolimnetic oxygen deficits, taste and odor problems, and increased concentrations of manganese and iron (e.g., Prepas and Burke, 1997). It is not applicable in shallow lakes without well-developed hypolimnia. It is difficult to mix a narrow water stratum near the bottom water layer. For successful application of this technique, the total oxygen demand of the lake must be correctly estimated, checked for possible metalimnetic oxygen minima, and the phosphorus-binding capacity of the bottom sediments considered. When sediments are rich in sulphate, a decoupling of the iron and phosphorus cycles takes place, and no positive effect is achieved (Hupfer and Zippel, 1998). The procedure has the *advantage* that the high concentrations of elements in the hypolimnion are not transferred to the epilimnion, and do not enhance algae growth. The improved oxygen conditions in the hypolimnion improve water quality by decreasing the iron, manganese, tastes and odor problems for drinking water supply, decreasing the damage to turbines and other structures by corrosion, and improving the downstream water quality. Cultivation of sensitive fish also is facilitated. Possible *negative environmental impacts* may occur in lakes and reservoirs in pristine areas by the transportation and installation of the fairly large equipment, and the installation and use of electricity. The investment *cost* is much higher than for destratification, due to the needed equipment. The operating costs depend on the hypolimnion area, the rate of oxygen consumption in the lake and the degree of thermal stratification, and can be calculated with a procedure given by Cooke et al. (1986). A combination of hypolimnetic mixing, with the injection

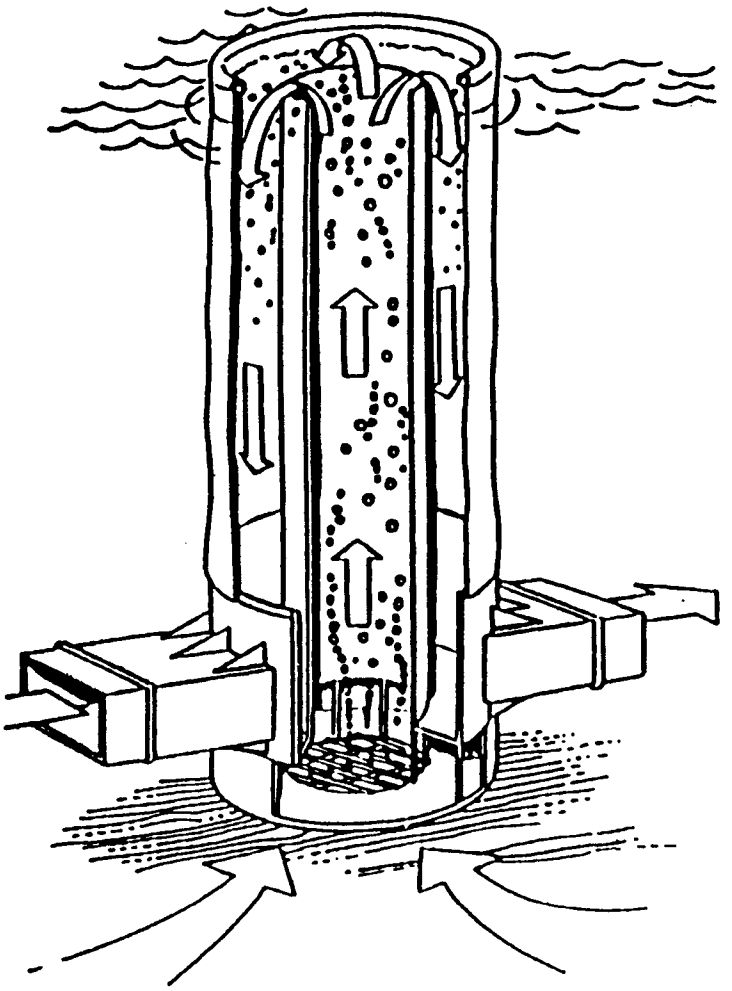


Fig. 4.11. A hypolimnetic aerator in operation (modified from Anonymous, 1989).

of hypolimnetic calcium hydroxide [$Ca(OH)_2$] was used in Germany (Anonymous, 1994) (Fig. 4.12).

*Hypolimnetic oxygenation*

Pure oxygen generated at the lakeshore is used to increase the oxygen concentrations at the lake bottom without extensive mixing (Fast et al., 1975; Gemza, 1997). The oxygen is injected into the device, with the movement of water from the hypolimnion and back created with a propeller. Oxygen-enriched water was injected at a rate of 0.2 $m^3\ s^{-1}$, with concentrations between 8–11 $mg\ l^{-1}$. The fine bubbles released from the device that do not

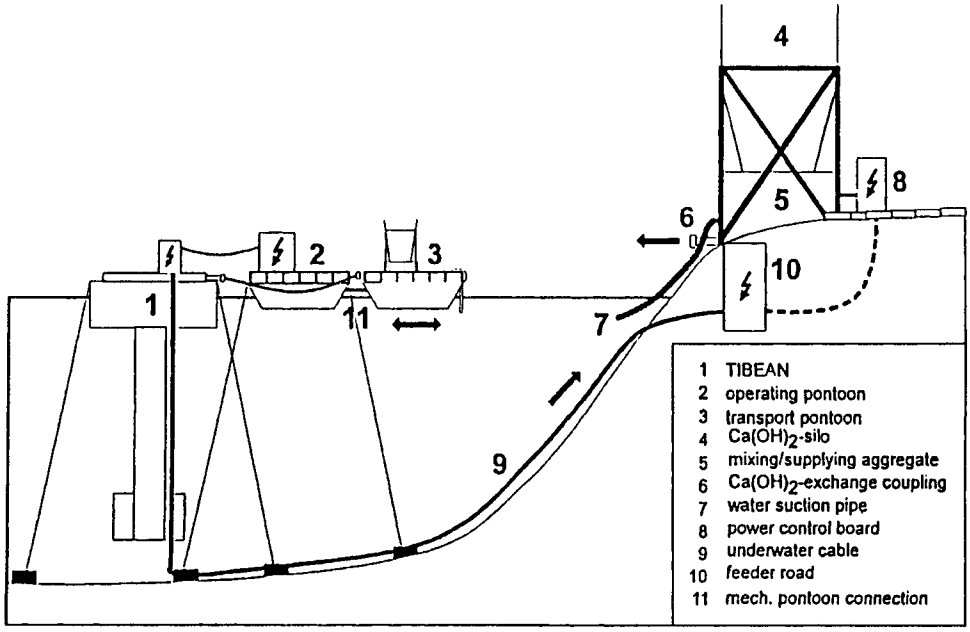
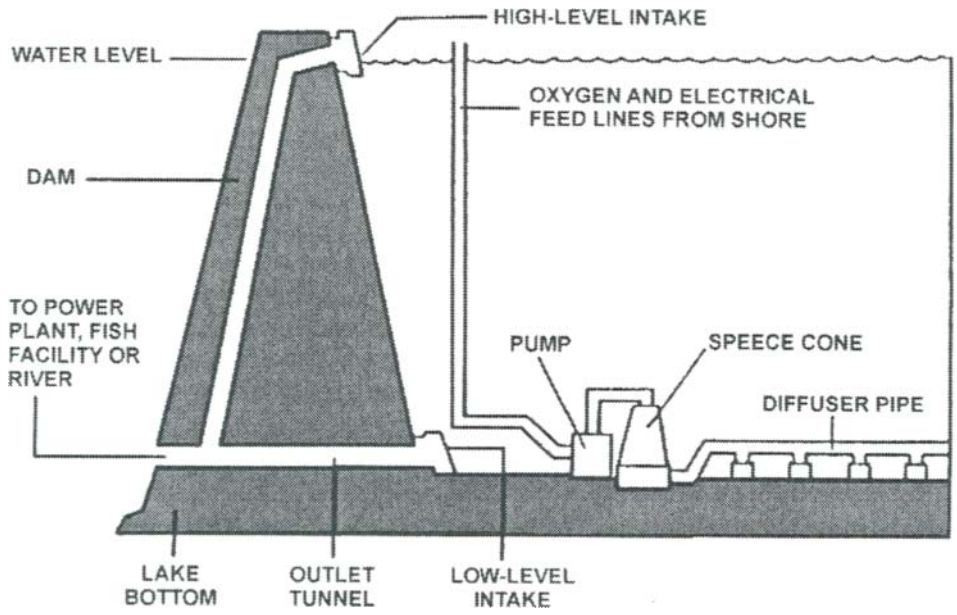



Fig. 4.12. Schematics of the pilot project on combination of hypolimnetic aeration and hypolimnetic $Ca(OH)_2$ addition, with the use of the hypolimnetic aerator type TIBEAN, as used in Germany. From Koschel et al. (1998).

dissolve in the water are trapped by a bubble collection hood, so that little mixing takes part while the water is being enriched with oxygen. The number of operations using this method has been low to the present time. Thus, it is difficult to highlight its *advantages* relative to hypolimnetic aeration. In present applications, however, the improvement in water quality was significant. Because the device used is simpler than the one used for aeration, the *costs* can be less.

Speece et al. (1982) and Speece (1994) used a special device (i.e., *Speece cone*) to supersaturate a certain amount of water up to 50–150 mg $O_2$ $l^{-1}$ in a special underwater conical chamber. The water is then horizontally diffused into the hypolimnion at a low speed, without breaking the thermal stratification. Detailed studies of the bubble dynamics and oxygen transfer in the Speece cone were made using a mathematical model by McGinnis and Little (1998). Practical experience with the water quality improvement resulting from the hypolimnetic oxygenation, in comparison to the widely-used hypolimnetic aeration, shows higher hypolimnetic dissolved oxygen levels, lower levels of induced oxygen demand, and maintenance of more stable thermal stratification (Beutel and Horne, 1999). The procedure is illustrated in Figure 4.13. Kennedy et al. (1995) reported the investment cost for a large reservoir (a volume of 1.29 billion $m^3$) to correspond to US $5 million, with the operating costs being about US $800,000 per year.




*Epilimnetic mixing*

Straškraba (1986) named this technique, suggesting its use in deep waters. However, it has generally only been used in situations in which the whole waterbody, not just the epilimnion, is mixed. Two possible arrangements are evident in Figure 4.9. The method is highly recommended, as it is the only mixing method aimed at prevention, rather than correction. The main goal is to reduce phytoplankton growth, irrespective of high phosphorus concentrations. This is achieved by decreasing the light available to the phytoplankton population, by mixing it to deep, dark strata. The surface water layers are mixed to a predestined depth conducive for minimal net phytoplankton production. This is achieved when the mixing depth ($z_{mix}opt$) is equivalent to the depth at which net photosynthesis is reduced to zero (i.e., water column respiration equals water column photosynthesis). More detailed explanation of this approach is given in Section 4.3.8—*Manipulation of the Underwater Light Regime*. Such mixing systems were originally developed for stratified impoundments in England. A mixing system destroys or prevents thermal stratification. Some systems use compressed units which discharge air bubbles through bottom diffusers (Symons et al., 1965, 1967), while others physically transport water with pumps (Ridley et al., 1966).

Cooley and Harris (1954) designed the mixing system of $\Phi$-angled inlet jets installed in the London drinking supply reservoirs. Summer thermal stratification was prevented, or an established thermocline in these reservoirs, about 17 meters deep, was destroyed. Wind

energy applied to the longest axis of the reservoirs helped mixing. In practice, the degree of achieved horizontal and vertical mixing resulted in isotherms throughout the summer, and the mixing of algae to dark strata. Ridley and Steel (1975) proposed changing the optical properties of the reservoir water, by using the $\Phi$-angled jet to re-suspend bottom sediment material. It appeared that this option could increase water treatment costs.

Several *advantages* of epilimnetic mixing of London reservoirs are reported:

- Deoxygenation and the release of nutrients from the sediments is prevented by the absence of a mid-summer hypolimnion,
- Mixing operations reduced algal crops in these highly eutrophic conditions,
- Transformation of a stratified waterbody with a warm epilimnion and deoxygenated hypolimnion into a cooler, well-mixed and well-oxygenated one improved trout fishery.

A result of treating raw, eutrophic riverine water was that the overall *costs* of producing a satisfactory drinking water were much reduced. The mixing system originally designed to double the supply of potable water by oxidizing the deoxygenated hypolimnion (Cooley and Harris, 1954) proved to be a major cost-effective tool for the limnological management of eutrophic impoundments that previously had large blue-green blooms that did not respond to algaecide treatment (Steel, 1972, 1978a, 1978b; Duncan, 1990).

In deep waterbodies it is suggested the principle of epilimnetic mixing by the standard diffuser installed on the bottom of a tube of 2–3 meters in diameter be used, locating the lower end to the depth $z_{mix}opt$. In this way, only the upper layers of the waterbody are mixed, and the hypolimnetic conditions are not directly tackled. Anoxia does not occur, however, since the primary algal production is minimized.

*Metalimnetic aeration*

Metalimnetic aeration oxygenates the water in the metalimnion without destructing either the hypolimnion or the epilimnion (Stefan et al., 1987). The purpose is to produce an oxygenated refuge in the metalimnion for *Daphnia*. A fully submerged bag-type device is used.

*Layer aeration*

Kortmann et al. (1982, 1988) and Kortmann (1994) used a detailed knowledge of stratification and heat conditions in lakes and reservoirs for the redistribution of available heat and oxygen in a stratified lake or reservoir. By means of specific devices (Fig. 4.14), discrete layers of lakes or reservoirs are created and aerated, establishing several thermocline-type barriers to vertical diffusion, and reducing the volume and areal extent of the hypolimnion. Methods for calculating the size of the system are given by Kortmann et al. (1994). *Advantages* of the method include that manipulation of the thermal structure creates desirable physical/chemical conditions, as well as an oxygen supply. Because it is possible to act selectively, the negative side effects of mixing which often accompany destratification (e.g., nitrogen supersaturation, increased diffusion transport to the epilimnion, metalimnetic anoxia) are avoided. Oxygenation is very efficient; in one case, only 3.2 $m^3\ d^{-1}km^{-2}$ were needed, compared to 41 $m^3\ d^{-1}km^{-2}$ for hypolimnetic aeration (average of 15 applications, Cooke et al., 1986). The *cost* is about one-fifth that of hypolimnetic aeration.

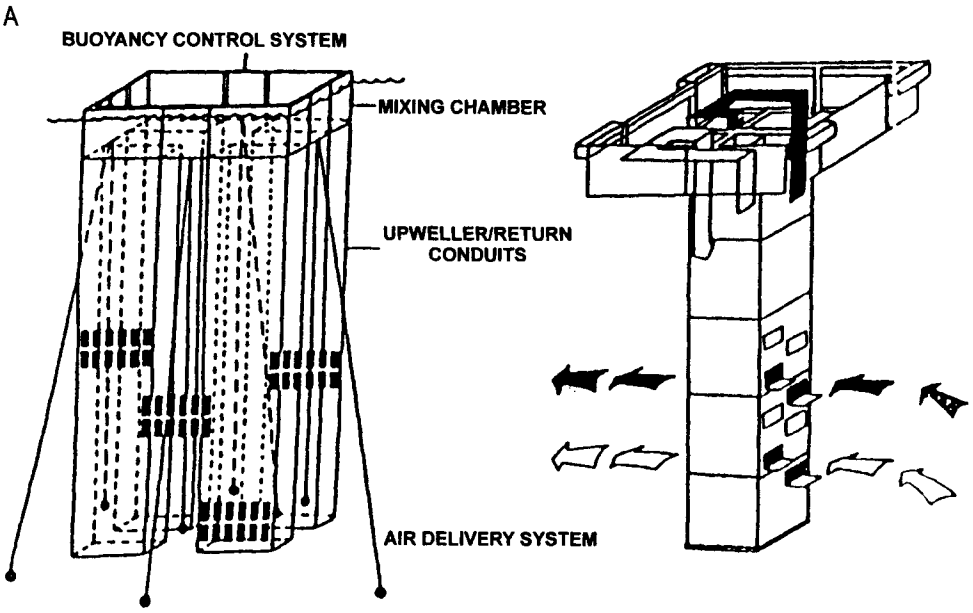



Fig. 4.14. Principle of layer aeration and devices used for layer aeration. Modified from Straškraba and Tundisi (1999). A—Schematic representation of difference between layer aeration, hypolimnetic aeration and artificial circulation (from Kortmann, 1994).

*Propeller mixing and oxygenation*

Propeller mixing and oxygenation is a system differing from all previously-discussed methods in the manner in which mixing is accomplished. The others are based on mixing accomplished by rising air bubbles from lower layers in a waterbody. This method entails mixing from above by the use of a propeller. The propeller is attached to the bottom of a pontoon, and can be used in many different applications. It is combined with a compressor that jets bubbles into the propeller region so that the surface layer becomes oxygenated. Technical specifications are given in Fay (1994). A device called the Stratification Breaking Propeller, with the propeller located at the bottom of a brad pipe similar to standard aerators for deep waters, reverts the operation of bubbler-based mixing devices by sucking water from the waterbody surface and releasing it near the bottom. A flexible "accordion-like" pipe for this propeller is described by Zoran and Milstein (1999).

*Comparison of different oxygen transfer devices*

The efficiency and cost of three different oxygen transfer devices (hypolimnetic aerator, bubble plume and Speece cone) were compared by McGinnis et al. (1998). A combination of direct measurements performed on the reservoir in different years, and a mathematical model representing a modification of the model by Wüest et al. (1992), was used. The results have shown that, in a 70 meter deep drinking water supply reservoir, the bubble

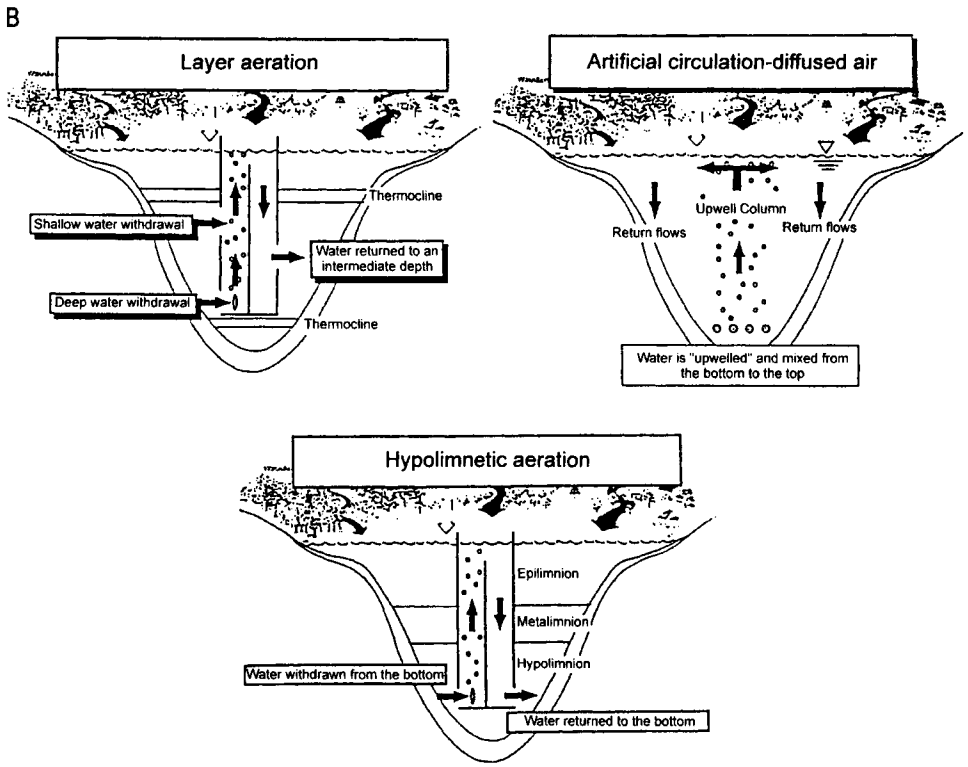



Fig. 4.14 (*continued*). B—Two devices for layer aeration (modified from Straškraba and Tundisi, 1999).

plume oxygenator (with a length of 2500 meters and initial bubble diameter of 2.5 mm) were very efficient in oxygenating the hypolimnion. The initial bubble diameter proved to be a very sensitive parameter. However, the consequences for other components of the system, and the water quality in general, were not investigated.

### *4.3.3 Hydraulic Regulation*

Hydraulics and hydrodynamics play an important role in defining the water quality of standing waters. The possibilities for modifying the natural flow conditions in lakes are limited. The situation is different for reservoirs, however, for which the location of the outlet, intensity of inflow and outflow can be manipulated to a different degree. Proper location of the reservoir and the positions of the outlet elevations (multiple outlets) play an important role in water quality optimization. From a water quality perspective, one important variable is the retention time of water in the reservoir, both the theoretical retention time

and the actual currents in the reservoir. Determination of the theoretical water retention time during the planning stage represents an inexpensive and feasible management strategy for water quality improvement.

The following options are distinguished:

- Hydraulic regulation (also called selective withdrawal),
- Diversion of inflow water,
- Extraction of hypolimnetic water; hypolimnetic siphoning.

*Dilution*

Welch and Patmont (1980) applied the method of dilution in Moses Lake in Washington, obtaining a significant reduction in algal blooms by decreasing the water retention time of the lake from 10 to 5 days. The limitations of the approach are great. It is applicable only when sufficient water of low phosphorus and algae content is available. Because of the high demands on water quantity and quality, it is rarely possible to dilute an entire lake or reservoir.

*Lake level manipulation*

During decreased lake levels, the shallow sediments can desiccate, getting 20–50% compacted and consolidated. During sufficient aeration of the sediments, the organic matter is oxidized and the content of organic matter decreased.

Macrophytes are reduced with decreased lake levels. Three studies have shown a reduction of 50–90%, due to freezing and drying of roots. The decrease in the winter time was more effective than that during summer. The decrease was drastic for *Elodea densa* (by 84% in one case), but the stand regenerated rapidly the following year.

*Diversion of inflow waters*

The best known example of the successful diversion of polluted waters from a lake is that of Lake Washington (Edmondson, 1972, 1991). Lake Washington is an urban lake, being surrounded by a population of 500,000 inhabitants, and used for drinking water supply of Seattle, as well as for the recreation of its drainage basin inhabitants. This successful story was the result of concentrated scientific and political activities of W.T. Edmondson, who convinced the local politicians and managers about the usefulness and need to divert the pollution from the lake in order to save its water quality. A collector of domestic sewage was constructed along the lake to divert the treated wastes to Puget sound.

In this way, the wastes were diluted with a much higher water amount, producing little apparent harm to Puget sound, and leading to successful restoration of water quality in Lake Washington. The use of this approach is possible, however, only when a much larger volume water body is available for diluting the pollution, which is relatively rare.

The *advantages* of the method include the costs, which are less than if all the effluent sources have to be treated separately, including tertiary treatment. The *limitation* consists of the need for a sufficiently large waterbody capable of assimilating the effluents.

*Hypolimnetic withdrawal by the Olszewski method*

This method consists of sucking hypolimnetic waters of low oxygen and high iron, manganese and phosphate contents, from a lake. It is a method proposed by Ruttner (1931) and used very early in the history of lake restoration (Olszewski, 1961, 1967). The principle is shown in Figure 4.15. It is based on the presumption that a place can be found around a lake which is considerably below the lake's surface level, so that with some sucking a siphoning effect can be created to produce a free flow of water through a tube. Some minimal harm utilization of the water sucked out of the lake hypolimnion also must be guaranteed, either to fields as a source of fertilization, or to a larger river where the pollution will be sufficiently diluted.

Nurnberg (1987) provided a summary of the procedure used in 17 eutrophic or hypertrophic lakes with phosphorus inputs drastically reduced, before its application in Europe and the United States. Limited beneficial effects resulting from the reduction was observed, because of substantial phosphorus releases from the anoxic sediments. Both the depth of anoxic water and the total phosphorus concentrations decreased after hypolimnetic withdrawal was begun. However, no decrease of the duration of the anoxic period was observed in larger lakes. Significant decreases of epilimnetic total phosphorus concentrations were observed after 2–3 years of operation (or earlier in smaller lakes), and appeared more distinct as the water withdrawal has continued. The average annual decrease was 11%, with highly eutrophic lakes experiencing a slightly larger percentage decrease than lakes with lower initial total phosphorus concentrations. The decrease of the epilimnetic total phosphorus concentration was significantly correlated with total phosphorus export via hypolimnetic withdrawal.

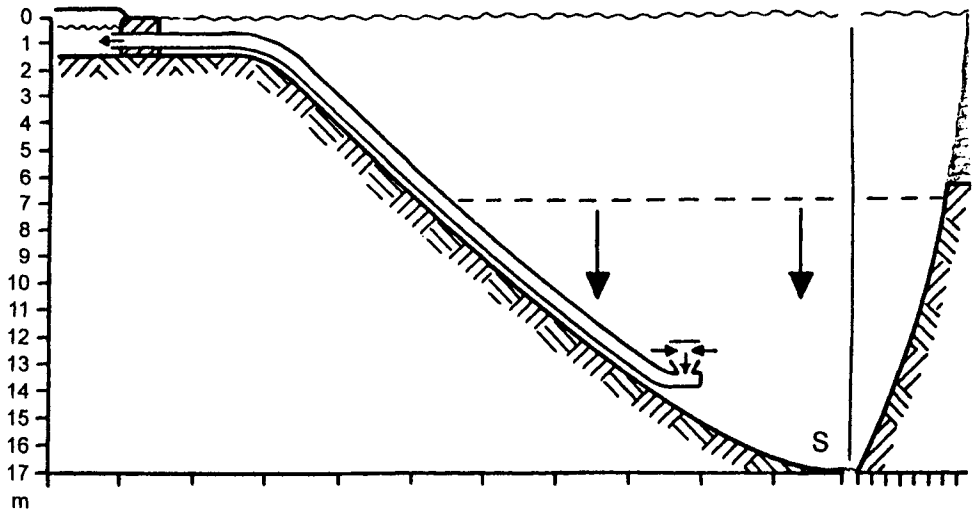



Fig. 4.15. The principle of the hypolimnion siphoning method of Olszewski (1961, 1967). (Orig.)

*Advantages* of the method consist of its very low labor needs and its continuous effects. Once installed, it can run unattended. A *limitation* is that the condition decreases the phosphorus load before the application starts. The method is directed to shortening the period of internal phosphorus loading effects. The needed location of the tube outlet below the level of the tube inlet is not feasible in a flat territory. The best results are achieved when the tube inlet is placed into the depth of maximum phosphorus concentrations, but does not touch the sediments. The *cost* is very low for both installation and operation.

### *4.3.4 Biomass Removal*

The removal of biomass from a lake focuses primarily on removing aquatic plants. It implies that a corresponding quantity of the elements (phosphorus and nitrogen being of most interest from the context of eutrophication) that constitute biomass also will be removed. The composition of the littoral and benthic vegetation reflects the composition of the environment, (i.e., the sediment and the water). Macrophytes, for example, can take up not only nutrients, but also pesticides and heavy metals.

The methods of biomass removal fall into three categories:

- Biomass harvesting,
- Use of plant-eating aquatic animals,
- Other methods.

*Mechanical plant harvesting*

Biomass may be removed from a lake or reservoir by harvesting the macrophytes (e.g., Moss, 1995). In addition to eradicating obstructions to transport, clogging of structures, and organic decay problems, this method may result in a nutrient and toxic substance reduction of major significance for an aquatic ecosystem and should be considered as a supplement to other management possibilities.

There are many different types of harvesters, depending on the type of vegetation to be harvested (submergent, emergent, floating), the size of the waterbody, the quantity of plants to be harvested, and the depths to be reached. They range from relatively simple arrangements on small paddle boats, to strong barges of several meters length with elaborate mechanics for harvesting. Many firms in several lake-rich industrial countries sell diversified arsenals of such boats.

Macrophyte harvest is applied to many Chinese lakes to provide pig, duck and cattle feed. Mass balance calculations show that the nutrient removal by this process may be important for the entire lake's nutrient budget. Depending of the magnitude of the external nutrient load, the potential removal of phosphorus through weed harvest has been estimated to range from 20% (Wile, 1975), 37% (Carpenter and Adams, 1977) and 60% (Welch et al., 1979). A hypothesis to explain the relations between macrophytes and internal phosphorus loading was presented by Welch and Kelly (1990).

The time of harvest is often important. *Phragmites* have their peak nutrient concentrations in western Europe in the late summer (August/September), and harvesting them

at that time will often double or triple the quantity of nutrients removed, compared with harvesting them 1–2 months later.

As indicated above, removal of heavy metals and pesticides is possible with the use of macrophytes. The use of plants for removal of toxic substances has been practiced for purifying contaminated soil, with good results (see Jørgensen, 1993). The plants are able to attain concentrations of heavy metals 50–1000 times the concentration in soil water. The concentrations are slightly smaller for pesticides (in the order of 10–200 times). If these results are considered in a lake management context, it should be possible to remove a significant amount heavy metals over a period of 3–4 years by harvesting 2000–3000 kg $ha^{-1}$ each year.

If rooted vegetation is harvested, however, the diffusion from the lake water to the interstitial water in the sediment may cause a slower removal rate. In fact, large-scale experiments of the application of vegetation for removal of heavy metals and pesticides from lake and reservoir water have still not been carried out.

A theoretical problem is associated with the application of weed harvesting to lower the phosphorus concentration in a lake and, hence, the algal concentrations. If rooted macrophytes obtain most of their nutrients from sediments, then weed harvesting would actually be reducing an internal nutrient source, rather than removing a portion of the annual phosphorus inflow. If the internal phosphorus supply is relatively large, a reduction of lake phosphorus could result from harvesting and interrupting the transport of phosphorus from sediment to water via rooted plants (Welch and Kelly, 1990; Welch et al., 1994). There may be other problems, however, because increased excretion of phosphorus, and thereby of algal blooms, have been observed following weed harvesting in some cases (Nichols, 1974). In addition, the harvestable biomass may be depleted with each successive year of harvest. If the external phosphorus load is relatively large, the phosphorus content of the lake generally cannot be reduced, even though large masses of weeds can be removed.

Thus, each individual case must be examined carefully before this management tool is applied in practice. The examination should always include in-lake experiments, and the use of quantification assessments either by mass balance calculations or models (see Chapter 5). The role of submerged vegetation for fish spawning, however, should always be considered before implementing a biomass harvest plan.

*Use of aquatic animals for plant removal*

Because aquatic plant harvesting methods are relatively expensive, and chemical treatment methods have negative consequences, there is a recent trend toward the use of biological organisms, such as vertebrates (manatees), and insects (beetles, sandhoppers). Different groups of aquatic animals can be helpful in macrophytes removal. The order of their extent of use is as follows:

- Fish,
- Aquatic vertebrates,
- Invertebrates.

*Fish*

Table 4.6 gives the characteristics of the fish species generally used for biomass removal. Grass carp are particularly attractive for this purpose, as they use aquatic vegetation as their only food source. It is possible to remove plant biomass equivalent to 1–5 times the weight of the grass carp on an annual basis (wet-weight basis). Heavily-eutrophic ponds typically contain in the order of 10,000 kilograms or more of plant biomass (wet-weight basis) per hectare, and a complete removal of this biomass would imply the removal of about 50 kilograms of nitrogen and 10 kilograms of phosphorus per hectare, a significant quantity of nutrient removal. If a depth of 5 meters is considered, it will correspond to the reduction of the nitrogen and phosphorus concentrations, respectively, of 1 and 0.2 g $m^{-3}$. This removal, consistent with the above-noted removal capacity of 1 kg/carp, requires a standing stock of about 3000–10,000 kilograms of grass carp. Experience shows that such a high density is very difficult to maintain in a waterbody unless artificial oxygenation is used. A more realistic, but still relatively high, density would be in the order of 1000 kilogram of grass carp per hectare, which would yield a removal capacity of 1000–3000 kilograms of biomass, corresponding to a removal of 5–15 kilograms of nitrogen and 1–3 kilograms of phosphorus. For a pond with a depth of 5 meters, this would correspond to a reduction of the nitrogen concentration by 0.1–0.3 g $m^{-3}$, and the phosphorus concentration by 0.02–0.06 g $m^{-3}$. The need for a continuing harvest of a sufficient fraction of the grass carp also should be emphasized. In many cases, the effect of fish release is hardly measurable without a significant harvest of this fish, which obviously will mean a reduction of the phosphorus and nitrogen corresponding to the content of these elements in the harvested biomass.

It can be seen from these calculations that a significant biomass removal by the use of grass carp is only feasible for relatively small lakes (probably up to about 10 hectares in area) and ponds. However, this does not imply that fish release should not be used for the removal of macrophytes for larger lakes and reservoirs. It is necessary in each case to establish management goals, and compare different strategies to achieve these goals, in order to make a proper selection of the environmental management plan for a lake or reservoir. If macrophyte removal is desirable for a larger lake, one of the test scenarios should be to use fish release for removal of macrophytes. The evaluation of the text scenarios, however, should encompasses all quantitative aspects of the problem, to ensure that the expected results will be achieved.

This method is not recommended for a lake or the reservoir contaminated by toxic substances, since the fish will biomagnify the toxic substance, rendering the fish useless for human consumption.

A new, but already widely-spread nuisance is the invasion of aquatic macrophytes into areas in which they were previously not established. Two management approaches are applicable in this case; namely, a protectionist and an interventionist strategy. The former group attempts to prevent invasions, usually on the basis of legislation. However, this often does not work, since the invasions are caused only indirectly by human activities, and are not intended. The interventionist strategies remove or suppress the invaders by these discussed methods.

Table 4.6. Characteristics of fish species used for biomass removal

| Characteristics | Silver carp | Grass carp | Wuchang fish |
|---|---|---|---|
| • Largest body weight | 20 kg | 35 kg | 3 kg |
| • Typical mature weight | 5 kg | 5 kg | 0.5 kg |
| • Feeding habits | Filter | Herbivorous | Herbivorous |
| • Main food (adult) | Phytoplankton algae, detritus | Aquatic macrophytes | Zooplankton with aquatic vegetation |
| • Water quality | Eutrophic | Clear water | Clear water |
| • Highest tolerated $BOD_5$ (mg $l^{-1}$) | 30 | 15 | 30 |
| • Spawning temperature | 18–30°C | 22–28°C | 20–28°C |
| • Dissolved oxygen at high feeding intensity | > 4–5 mg $l^{-1}$ | > 4–5 mg $l^{-1}$ | > 4–5 mg $l^{-1}$ |
| • Dissolved oxygen at appetite loss | 1 mg $l^{-1}$ | 1 mg $l^{-1}$ | 1 mg $l^{-1}$ |

*Aquatic vertebrates*

Recent support of the proliferation of the water cow, manatee (*Halicore dugong*), in Florida is used to increase consumption of aquatic macrophytes and thus decrease the blockage of waterways. Grazing by domestic cattle also can be an efficient means of controlling macrophytes in the lake shallows. Although other aquatic vertebrates, such as water buffalo and hippopotamus also can feed intensively on aquatic vegetation, their numbers in a given location are usually not sufficient to promote and maintain significant plant destruction.

*Invertebrate*

Among the beetles, the most intensively studied are weevils, belonging to the family of Curculionidae. In the tropics, *Neochetina eichhorniae* and *N. brucei* are used to control floating vegetation, limiting reproduction of these plants. The weevil *Eurhynchiopsis lecontei* is native to North America and is used to control the densities of an invasive species to this region, the European milfoil, *Myriophyllum spicatum* (Creed, 1998). The application of weevils to control aquatic vegetation is reaching a commercial level in the United States. The alligator flea beetle (*Agasicles hydrophila*) also is considered a potential agent. Some naturally-reproduced Brazilian grasshoppers have been observed to destroy floating vegetation in newly-flooded reservoirs.

*Other methods*

Methods for chemical eradication of macrophytes and blue-green algae (see Section 4.3.8) are still fairly widely used, although their negative consequences are widely known. Many of the chemicals accumulate in the waterbodies, and are incorporated into different biotic and abiotic components of the environment (particularly predatory fish and higher vertebrates). They are potentially dangerous from the human health perspective, particularly when the water is used for drinking water supply. Some of the newer products have not yet

been sufficiently tested to guarantee the proclaimed specificity in regard to their potential impacts of the health of humans and other higher organisms.

One new method for local protection of selected shoreslines, like lake beaches, against macrophyte growth is the use of new generations of highly-resistant, stable screens.

Of course, aquatic vegetation does not only have negative consequences and, therefore, its control must be considered. In small impoundments in the United States, for example, fishermen have observed a decline of predatory fish harvest due to previous overeradication of aquatic vegetation. In such cases, efforts have been made to re-introduce vegetation into such waters (Fischer et al., 1999). Motor-boating is also a cause of macrophyte vegetation disturbance, with no-wake zones being discussed as a means of vegetation protection (Asplund and Cook, 1999). Conflicts between different users groups, therefore, may be created in such situations.

### *4.3.5 Sediment Treatment Methods*

Lake bottom sediments accumulate phosphorus over long time periods and, therefore, the phosphorus concentrations in the upper few millimeters of the sediments can be much higher than the phosphorus content of the overlying water column. This large phosphorus source is, to some degree, exchanged with the upperlying water, the net direction of the exchange depending on such things as the differences in concentrations at the water–sediment boundary. When the phosphorus concentration in the water column decreases (e.g., by decreasing the phosphorus load to a reservoir), the direction of phosphorus movement is from the sediment to the water. Because of this chemical reality, an individual waterbody may exhibit eutrophic conditions for several years after the external phosphorus load was reduced. Further, this exchange is enhanced as much as a factor of 10 when the bottom of the lake exhibits anoxic conditions in the water column. Thus, various procedures are used to decrease this exchange, including increasing the near-bottom oxygen concentration and/or by removing the upper layers of sediment and using mechanical barriers for the exchange.

*Sediment removal*

This method consists of removing the upper layers of the lake bottom sediments that are rich in phosphorus. Peterson (1982) reviewed different methods of sediment removal and their cost-effectiveness. Several types of dredging equipment are typically used for such purposes. The sediment has to be transported as a slurry, with an 80–90% water content, to a disposal area for dewatering. After drying and some chemical treatment, the sediment can be used as fertilizer (if it contains little or no heavy metals) or for other purposes (Fig. 4.16).

The *advantages* of the method include its relatively long-lasting effect. In Lake Trummen in Sweden, for example, the phosphorus concentration in the water dropped from a peak level as high as 900 $\mu g\, l^{-1}$ to less than 10 $\mu g\, l^{-1}$ and remained at that low level for the entire observation period extending over 9 years.[1] The *negative impacts* of the procedure

[1] In this example, the stormwater overflow from impermeable surfaces was not managed, and the problem began to reappear after a longer period of time.

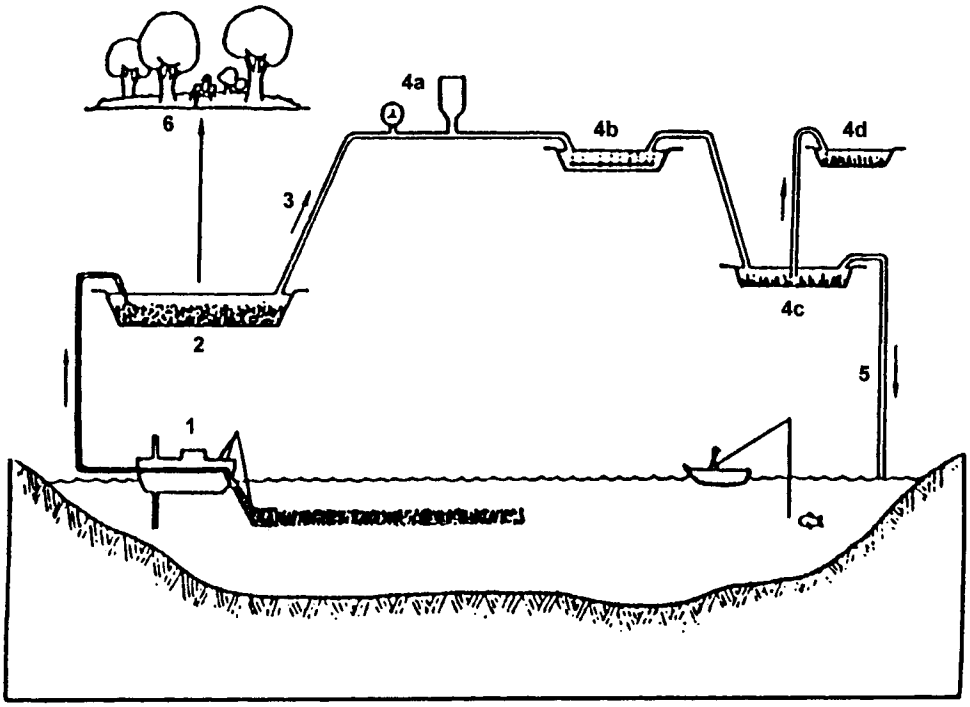



Fig. 4.16. Dredging a shallow lake for sediment removal. 1—The bottom mud suction dredger. 2—Settling pond for sediment drying. The dried material can be used for construction purposes. 3—Runoff water taken to the aluminum sulfate automatic dosing instrument 4a and its aeration basin 4b. The overlying water is returned to the lake through tube 5 (redrawn from Eiseltová, 1994).

are significant, as an extensive area is needed to store the dredged slurry before it dries out and can be used as field fertilizer. The *cost* of dredging is high, the figure given by Peterson (1981) for dredging alone amounting to between US \$0.23–15.00/$m^2$, and does not include disposal and transport costs. Using a mathematical model, Stefan et al. (1980) provided a means of predicting the dredging depth that will minimize internal nutrient recycling in shallow lakes.

*Sediment aeration and oxidation*

The *RIPLOX method* of sediment aeration and oxidation has been widely used in Scandinavia and Germany (Ripl, 1994) and focuses on decreasing phosphorus release from sediments. Ferric chloride is applied to the sediments low in iron to decrease phosphorus release. A very low pH is created in the sediment (pH = 3). Lime is then added in a quantity so as to create a pH optimal for denitrification (usually between 7.0–7.5). Calcium nitrate is subsequently injected into the top 30 cm of sediments to oxidize and break down organic matter and denitrify the sediments (Ripl, 1976, 1980, 1983) (Fig. 4.17). The

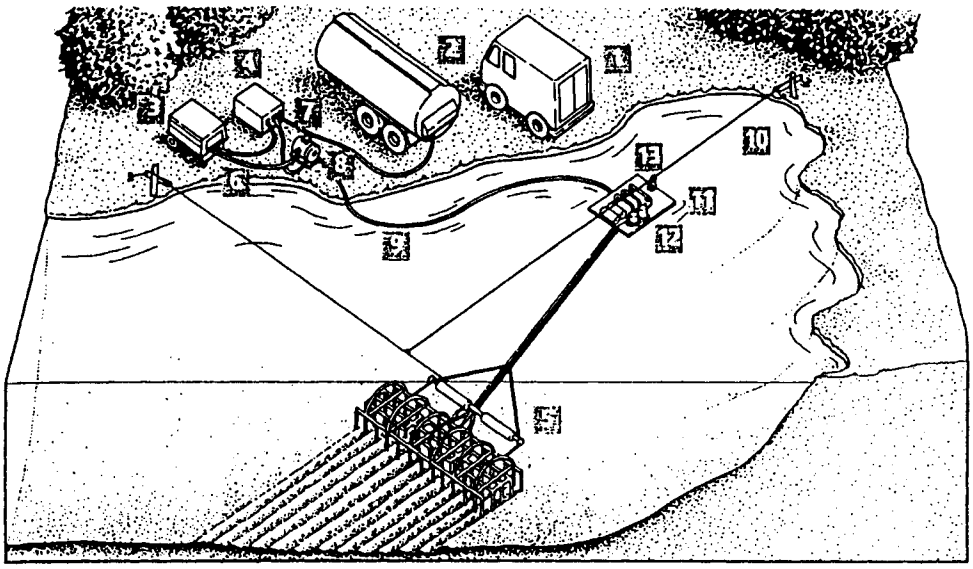


Fig. 4.17. The Ripl's RIPLOX procedure for sediment dosing with iron chloride to reduce the rate of phosphorus release. 1—Field laboratory. 2—Chemical supply tank. 3—Portable compressor for bubbling air into the chemical mixing tank. 4—Chemical mixing tank for the chemicals (calcium nitrate, ferric chloride). 5—Harrow-like device to loosen sediment with compressed air and inject chemicals. 6 and 7—Air feed lines to mixing tank and to drive pumps. 8—Air-driven pump. 9—supply line for chemicals. 10—Guide line for pulling the harrow across lake bottom. 11—Air driven dilution pump for mixing chemical with water and injecting mixture into sediment. 12—Dilution water intake. 13—Pneumatic winch (redrawn from Eiseltová, 1994).

first application of this method in Sweden was made in Lake Lillesjö, which was used for many years as a sewage recipient. As a consequence, fish kills often were observed, the lake being covered by *Lemna minor* in early spring. The oxygen concentration was close to zero for most of the year. When the sewage was diverted from the lake, the lake was treated with the RIPLOX method and the macrophytes also were harvested. After the treatment, the oxygen levels were high, the Secchi disk transparency increased, and the phosphorus concentrations decreased drastically. Recent application in an urban river arm in Vienna resulted in a significant reduction in nutrient and chlorophyll levels, a shift from Cyanobacteria to diatoms and green algae, and an increase in Secchi depth. Simultaneously, macrophyte growth became apparent (Donabaum et al., 1999). The *advantage* of this method is that the treatment is very intensive and has an immediate effect. Thus, it can be used for very degraded, shallow urban lakes of relatively small size. No large space is required, as with the previous method, and the same piece of equipment can be used in different lakes. The major *limitation* is the fact that the injection requires special equipment, which can only be used on flat and shallow bottoms. The *cost* of the first application in a small, shallow lake (e.g., area = 42,000 $m^2$, of which only 12,000 $m^2$ of bottom were

treated, mean depth ~ 2 m, and a maximum depth of 4.2 m) was US $112,000 (1995 dollars), spent mostly for the development of the equipment. Not linked to any specific mixing device, the equipment can be used in most applications, thus considerably decreasing the cost. The chemicals used with this method totaled about 13,000 kg of iron-III-chloride ($FeCl_3$) (= 10,800 kg $ha^{-1}$ treated and 3100 kg $ha^{-1}$ of the whole lake) applied over 6 days, 4200 kg $ha^{-1}$ of $Ca(OH)_2$ which was treated applied to the lake for 1 week, and 10,000 kg $ha^{-1}$ of $Ca(NO_3)_2$, with a total cost of ~US $3500/ha (1995 dollars).

*The Froth Tailing System* is a means of extracting organic matter from sediments. It is still in the experimental phase, giving promising results, but has not yet applied under natural conditions. The technology uses micro air bubbles and lime for washing out organic matter.

*Sediment capping*

An alternative and cheaper technique for the goal of reducing nutrient remobilization is to cover the lake bottom sediments with foil, clay, sand or crushed bricks, or other inert materials. A review of the properties, costs and effectiveness of alternative materials is given by Cooke and Kennedy (1988). Palermo (1998) presents design considerations for capping contaminated sediments. Capping the sediment with calcite was performed in Lake Arendsee in Germany (Rönicke et al., 1998). *Cyanobacteria* blooms disappeared from the previously heavily-infected lake, periods with phosphorus concentrations below 10 mg $m^{-3}$ were prolonged, and the mean Secchi disc transparency doubled.

### 4.3.6 *In situ Nutrient Inactivation*

*Inactivation of phosphorus*

This procedure consists of spreading alum (aluminum sulfate, $AlSO_4$, at present no longer recommended), sodium aluminate ($Na_2Al_2O_4$), or ferrous chloride into a lake. This forms flocculates that bind phosphorus and coagulate algal cells, which then settle to the lake bottom sediments, also sealing them from further release of phosphorus. The dense population of a mixture of dominantly green algae sedimented significantly during the first two days after the addition of alum to the water. Over a 30-day period, a sealing layer of the flocculate on the bottom was completed (Piedrahita, 1998). Experience in some countries has shown that the chemical coagulation of phosphorus in lakes is highly effective for prolonged periods up to 20 years (Cooke et al., 1993a, 1993b; Welch and Cooke, 1998). Based on comparisons of 21 applications in both polymictic and dimictic lakes by Welch and Cooke (1998), the average period of reduction ranged up to 10 years. Interference with this process occurs in shallow lakes overgrown by macrophytes, and when the external loading exceeds the phosphorus binding capacity of the flocculate. Evaluating the effects of the phosphorus reduction on overall water quality, Holz and Hoagland (1998) concluded that alum was extremely effective in controlling sediment phosphorus release rates, improving water clarity, reducing phytoplankton biomass, shifting phytoplankton species composition from Cyanobacteria dominance toward bacillariophytes and chlorophytes, increasing daphnid biomass, and increasing usable fish habitat. Keeping the water pH between a range

of 6–8 is necessary to enable the formation of sedimentable flocculate and to prevent an increase in the formation of potentially toxic dissolved aluminum. Thus, the dosage and effects will depend on the initial alkalinity and pH, and this techniques does not function well in eutrophic waters with very high pH values (Francko and Heath, 1981). Although no special equipment was used in most instances (Cooke et al., 1986; Welch et al., 1988; Conover, 1988; Welch and Cooke, 1995), Quaak et al. (1993) developed a technology using heavy machinery. Recent use of this technique involves applying alum and sodium aluminate separately. The aluminium dose is maximized to create a seal preventing the release of phosphorus from the sediment, and to control aluminum toxicity by maintaining a pH range between 6–8. The total lake area can be treated or, alternatively, just the areas exhibiting hypolimnetic oxygen depletion. Kortmann et al. (1994) have withdrawn water from the hypolimnion and, after adding chemicals, returned it to the metalimnetic–hypolimnetic interface. In large reservoirs, the application may be restricted to places with the greatest phosphorus release (Barko et al., 1990; Smeltzer, 1990). For small reservoirs, the addition of the chemicals to the inflowing water proportional to flow rates proved a feasible method (Bannink and van der Vlugt, 1978).

Lake treatment with iron (Fe) or calcium (Ca), which can be considered substitutes for aluminium, did not exhibit such long-term positive effects, although shorter-term positive effect were achieved. Walker et al. (1989) increased the efficiency of iron precipitation by releasing it by means of an hypolimnetic aerator. Deppe and Benndorf (1998) reported on successful remediation of a Cyanobacteria bloom in a reservoir, using a combination of bivalent iron application for phosphate precipitation, with the simultaneous transport of hypolimnetic water rich in free carbon dioxide into the upper layers. Randall et al. (1999) experimentally observed some effects of ferric sulphate on zooplankton feeding, due to its presence as particles interfering with *Daphnia* filtration, as well as indication of some toxic effects causing a reduction of reproduction and increased mortality rates. The safe level of iron (Fe) was estimated in this case as 1.7 mg $l^{-1}$.

Positive results also were achieved with calcium in hard-water lakes (see next subsection on calcite precipitation).

Calculation models by Kennedy and Cooke (1982), Kennedy et al. (1987) and Rydin and Welch (1998) can be used to determine the dose of alum necessary for a given lake's pH and alkalinity values.

*Advantages* of this technique are becoming increasingly appreciated, particularly since the addition of chemicals, particularly iron, does not cause any side effects. No special equipment is needed. Alum treatment is long lasting, observations showing at least three years, but typically longer, positive effects (average expectation of 10 years). A corresponding decrease in chlorophyll concentrations is not always observed after one chemical application (explicable on the basis of the nonlinear relationship between phosphorus and chlorophyll; see Section 2.2.2—*Eutrophication*). No similar evaluations of the length of effects are known for iron treatment, although the effect is definitely not as long lasting as for alum. *Drawbacks* of alum application include intensive labor needs, a decrease in the water pH due to alum, and the possible appearance of toxic dissolved aluminum from an overdose and decreased pH. Aluminum in drinking water is considered a health risk

(Reiber et al., 1995), and if the application conditions are not carefully followed, negative consequences may occur.

*Limitations* of this procedure include its inefficiency for lakes and reservoirs with retention times under one year, and its ineffectiveness in shallow lakes overgrown by plants. For example, the effects of the application of iron chloride to a shallow Dutch lake with a water retention time of 35 days lasted only for three months (Boers et al., 1992). The method also works better in deeper lakes, due to the absence of the re-suspension and dispersion that takes place in shallower lakes. However, one lake with an average depth of 2 meters was also successfully treated. Shallow lakes overgrown with submerged vegetation do not enable the sedimentation of flocculates and the sealing of the lake bottom. The magnitude of the sulfate loading during an alum treatment may adversely interact with the iron cycle through anaerobic respiration and, therefore, decrease the phosphorus binding capacity of the sediment. Kortmann et al. (1994) suggested that the use of aluminum nitrate may retain the beneficial properties of alum, while avoiding the negative impacts of sulfate loading. The *negative impacts* of this method are minimal, although there is some uncertainty about the possible long-term accumulation of aluminium in the sediments, and its possible release in toxic form at low pH (e.g., as a consequence of acidification). For these reasons, the addition of ferrous chloride is preferred in Europe, while alum application is widespread in the United States. The *costs* are relatively low, corresponding in Sweden to about US $3000–10,000 per hectare (1994 dollars).

Schulze-Rettmer (1991) has used a method potentially applicable in waterbodies for the simultaneous chemical precipitation of ammonium and phosphate in the form of magnesium–ammonium–phosphate.

*Calcite precipitation*

Co-precipitation of phosphorus with calcium carbonate is known to significantly reduce the productivity of hard-water lakes (Wetzel, 1975; Rossknecht, 1980; Avnimelech, 1983; Koschel et al., 1983, 1987; Murphy and Prepas, 1990). This observation led to attempts to enhance this process by adding calcium carbonate and calcium hydroxide to waterbodies (Murphy et al., 1983, 1990; Koschel, 1990, 1997; Babin et al., 1994; Dittrich et al., 1997).

Long-term investigations of calcite precipitation in four German and Austrian hard-water lakes indicated that the trophic state affects the dynamics, quantity, deposition, co-reactions and structures of calcite in the pelagic zone of the lakes. The highest average calcite precipitation in stratified hard-water lakes were found in eutrophic lakes. Calcite precipitation increased during the eutrophication process in lakes with trophic states ranging from oligotrophic to slightly eutrophic, decreasing again in the more polluted lakes. The highest efficiency of self-purification (rise of turbidity, co-precipitation of phosphate, rise in sedimentation and deposition) was reached with an increase of calcite precipitation and a change of calcite forms (Koschel, 1997). Applications in Germany in Lake Schmaler Luzin were directed to dosing $Ca(OH)_2$ into the hypolimnion, with simultaneous hypolimnetic aeration (Koschel et al., 1998). The result was a continuous decrease of the hypolimnetic phosphorus concentration, a more than doubling of phosphorus sedimentation, and a three-quarters decrease in the mobilization of phosphorus from the sediments.

In hard-water Frisken Lake, British Columbia, which receives high phosphorus loads from the natural weathering of apatite, the addition of 0.8 tons of $Ca(OH)_2$ per hectare resulted in the removal of 80% of the chlorophyll-a and 97% of the soluble reactive phosphorus from the lake. The addition of 0.8 tons/hectare of $Ca(OH)_2$ and 0.24 tons/hectare of $CaCO_3$ to hypereutrophic hard-water Halfmoon Lake in Alberta resulted in a decrease of total phosphorus and chlorophyll-a in two successive years of 53% and 63% of the pre-treatment values, respectively (Babin et al., 1994). The sediment release in the summer during both years was 50% lower. Because of trends of increasing loading, as observed in increased total phosphorus and chlorophyll-a concentrations in nearby lakes, the authors estimated that, after correcting for this trend, the chlorophyll-a concentration decreased to 24%, and the total phosphorus concentration decreased to 54%, of the pre-treatment values. Higher winter oxygen concentrations also resulted from the treatment.

*In situ nitrate elimination*

Preliminary attempts to eliminate nitrate were made in reservoirs in former East Germany. This was attempted by means of (i) a lattice of straw, and (ii) the addition of natrium thiosulfate to the lake hypolimnion. Both methods were used for Zeulenroda Reservoir. The methods resulted in bacterial decomposition of nitrate to elemental nitrogen (denitrification). The maximum obtained efficiency was a reduction of the hypolimnetic nitrate concentration of 90%. Economically, the straw lattice method was advantageous, as the cost corresponded to about 40% of the cost of the natrium thiosulfate method. However, the application of natrium thiosulfate to a lake was easier. The danger of the increased liberation of phosphorus and heavy metals was noted. The use of straw also was investigated in other places (Soares and Abeliowich, 1998).

### *4.3.7 Methods for Correcting Acidified Lakes*

Only two options are considered to be useful for correcting acidified lakes, namely liming and organic matter addition.

*Liming of acidified lakes*

In northern Europe, extensive liming was used to decrease the symptoms of acidification of streams and lakes. The symptoms include not only the disappearance of fish, but also toxicity to humans due to dissolution of aluminium. More details about the magnitude of this problem and methods for its solution can be obtained from Section 6.3. Liming is recommended only as an emergency method, and will not fundamentally solve the problem. Its ultimate solution lies in preventing acidification by reducing or eliminating $NO_2$ and $NO_x$ atmospheric emissions at their sources.

*Adding organic matter to acidified lakes*

It was recently recognized that some addition of organic wastes to an acidified lake improves its condition by raising its pH, and promoting some growth of algae which represents an additional buffer (Davison et al., 1991). The degree of wastes and their desirable composition has to be estimated for each case on the basis of a simple balance calculation.

### 4.3.8 Other Methods

This section discusses methods that do not belong to any of the previously-defined groups. The first technique focuses on the effects of light on phytoplankton photosynthesis, and was previously mentioned briefly in connection with epilimnetic mixing. The second technique is poisoning of algae by chemicals, discussed here to provide a complete picture of alternative methods, but recommended to be avoided whenever possible.

*Manipulation of the underwater light regime*

The most effective preventive technique is to create conditions that do not allow algal biomass to grow. The net algal growth in a lake or reservoir is a result of two simultaneous processes, although occurring with variable intensity during the day and night. On the one hand, algal photosynthesis takes place during the day, during which organic matter is produced. On the other hand, respiration degrades the organic matter back to its mineral components. While photosynthesis depends on light, therefore occurring only in the uppermost sunlight-illuminated water layers during the day, respiration takes place in all depths and over the full 24 hours of a day. The illuminated zone discussed here is defined approximately as the depth in the waterbody to which at least 1% of the surface light penetrates, and is labeled $z_{eu}$ (Fig. 4.18). No photosynthesis of algae is assumed to take place below $z_{eu}$. If the algae are mixed to depths greater than $z_{eu}$, they obtain less light on average because they are only within the illuminated zone for a part of the day. During the other part, they are located deeper in the waterbody, and do not photosynthesize in the dark, but only respire. The mixing depth within which the algae are intensively mixed can be calculated, for which the photosynthesis in the whole mixed water column equals respiration and there is no net growth of algal biomass. This depth is termed $z_{mix}opt$.

The depth of light penetration depends on the extinction coefficient of the water, which defines which proportion of light is adsorbed in one meter of the water column. The extinction coefficient depends on the mineral and biotic particles, and the content of refractory colored organic matter, in the water. Thus, it is a unique characteristic of each waterbody (and possibly each period of time). In determining the extinction coefficient for the purposes of this discussion, the biotic particles (mainly algae, producing the so-called self-shading effect, termed $\varepsilon_q$) are not considered. In addition to light, algal photosynthesis also depends on temperature and the concentration and supply of nutrients. When nutrients are in abundant supply, algae will photosynthesize vigorously. It is fortunate that the minimal net algal growth described above can also be achieved in waterbodies with high nutrients. The temperature effect can be taken into account with the detailed calculation of $z_{mix}opt$, by means of a mathematical model of the integral photosynthesis (integrated over mixing depth and over 24 hours). As a rough preliminary approximation for temperate summer conditions, $z_{mix}opt$ can be considered to be equal to the condition of $z_{eu} < z_{mix}$ which is given by $z_{eu}\varepsilon_q = 3.7$ (Steel, 1972). An additional manipulation might be possible by increasing the background light attenuation coefficient ($\varepsilon_q$), as has been suggested by Ridley and Steel (1975).

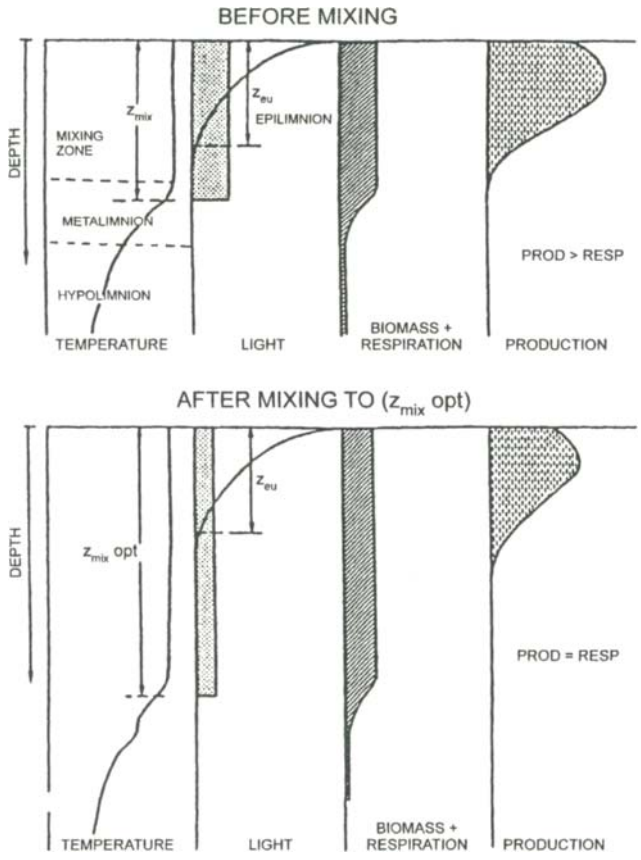



Fig. 4.18. Explanation of the functioning of epilimnetic mixing. (Orig.)

*Algicid use, particularly copper poisoning*

The use of algaecides is to be discouraged. The addition of algaecides (e.g., simazine, copper sulfate) has long been used as an emergency measure to control excessive algal growths, usually when they are already well advanced. The copper sulfate ($CuSO_4$) dosage varies as a function of the depth of the algal layer, with a concentration of 1–2 mg $l^{-1}$ being desirable. The primary *advantage* of the method is that it works rapidly. The *limitations* are its high toxicity of the accumulating copper, and the short duration of its effects. In waters with an alkalinity above 150 mg $l^{-1}$ calcium carbonate ($CaCO_3$), or with high contents of organic matter, a chelated copper form must be used or copper will be rapidly lost from solution. McGuire et al. (1984) improved this method by focusing the application early in the bloom cycle, using granular copper sulfate. Kortmann et al. (1988) increased the effectiveness and achieved cost reductions by depth-discrete application.

Laboratory investigations of six methods of chemical control of Cyanobacteria blooms producing hepatotoxic microcystin showed lime and alum treatment did not produce cell lysis and subsequent release of toxins into water, but caused cell coagulation and sedimentation without any (lime) or little (alum) increase in the toxin concentrations in the water (Lam et al., 1995). Lime, and to a lesser degree alum, appear more suitable than either algaecides or chlorine for controlling microcystin-containing Cyanobacteria blooms in drinking water.

This method is not advisable because of its *negative impacts* on the environment. Even at doses below regulatory limits, copper is toxic to fish and zooplankton. In some instances, zooplankton kills caused by algaecides result in peaks of phytoplankton after detoxification, because the algae recover faster than zooplankton. Algae in regularly treated lakes also can get adapted to copper, necessitating a continuous increase in the copper dosage. This is particularly the case in tropical conditions, where regrowth is rapid and frequent reapplication is necessary. Demayo et al. (1982) observed that some algae develop resistance to copper. Thus, its application can favor some nuisance species. The decay of dead algal or weed biomass can cause oxygen loss and nutrient regeneration, resulting in fish kills and algal blooms following chemical treatment. Its use also leads to long-term accumulation of copper in sediments, or the addition of a toxic chemical to drinking water. The side effects of 58 years of copper sulfate treatment of Fairmont Lake (Minnesota) are described by Hanson and Stefan (1984). The associated *costs* depend on the needed dosage.

## 4.4 INNOVATIVE AND COMBINED METHODS

This section identifies some combinations of methods that have been considered and/or used for solving water quality problems for particular lakes and reservoirs, as well as some suggested innovative experimental approaches. The general principles of the integrated water management are discussed in Chapter 7.

### *4.4.1 Innovative Methods*

There is a number of innovative methods, particular from densely-populated countries (e.g., China, Japan), usually related to complex ecological considerations of water quality treatment.

As an example, Pu et al. (1998) have experimented with a hypertrophic, macrophyte-covered shallow bay of Taihu Lake in China, using what they call a physico-ecological engineering method. It is based on the in-lake creation of stable artificial ecosystems, in this case specifically developed water-isolated materials and filters constructed from bamboo stakes. The filters represent a physical–biological membrane, improving the water quality passing into the system from the surrounding lake to replace the water taken for drinking water supply. Filtration through two such membranes also is an option. In spite of decreased nutrient concentrations and turbidity inside the areas, certain kinds of snails appear to be efficient in further decreasing the turbidity. The low turbidity enables dense

growths of submerged vegetation, which also were harvested for use. Difficulties related to plant overgrowth by filamentous algae were solved by presenting artificial substrates for growth, which were then periodically taken out. There is no indication whether or not fish also were used to control water quality inside the system.

Niwa et al. (1995) reported on four complexes of innovative measures from Japan, including fallen leaf control system, current control system, pumice filtration system, and artificial suspended reef. The current control system is intended for riverine reservoirs, and represents an elaboration of the selective discharge facility as described in Chapter 6 with artificial mixing. Pumice with an automated cleaning method is used to remove phytoplankton. The suspended "reef" is constructed to provide suitable habitat for zooplankton, fish and snails to multiply and prey on Cyanobacteria and algae before they become abundant.

A mobile plant to withdraw the high nutrient content from the hypolimnion of lakes, called PELICON (from Phosphorus ELImination CONtainer), was developed by Keil and Meyer-Jenin (1995). Hypolimnetic water is pumped into the plant, consisting of one or more floating phosphate separators supported by a containerized shore base, supplying the coagulant to form sludge that is pumped into a sludge collector.

### *4.4.2 Combined Methods*

Because the efficiency of the management methods given in this chapter varies in different situations, there is good reason to combine different approaches. A typical, and necessary, combination is always to use both watershed and in-lake methods. The probability of getting water of good quality from very polluted lakes is extremely low, and the primary task should always be to decrease the pollution within the watershed. A combination of watershed and in-lake methods gives the best results. As an example, Wehrli and Wüest reported very successful reduction of the total phosphorus in Lakes Baldeggersee and Sempachersee in Switzerland, by combining the reduction of phosphorus loads from municipal wastewater and artificial in-lake mixing. In Baldeggersee, the reduction was from the peak annual averages of over 500 $\mu g\ l^{-1}$ in 1975 to below 100 $\mu g\ l^{-1}$ in 1994, and in Sempachersee from a maximum annual average of about 170 $\mu g\ l^{-1}$ in 1983–1985 to only 60 $\mu g\ l^{-1}$ in 1995.

Nevertheless, the success of the different in-lake methods given in Section 4.3 also depends on a number of circumstances, and good knowledge of the specific water quality situation is always necessary. Major differences exist particularly between the possibilities for shallow versus deep lakes, as outlined in Table 4.7. Thus, successful attempts also exist to combine different in-lake methods.

Stanley Lake, a drinking water reservoir near Denver, Colorado, had water quality problems arising from nutrients in local runoff, in discharges from upstream treatment plants, and within the reservoir itself. McGinnis et al. (1998) evaluated five general control strategies, including oxygenation, alum treatment, shoreline stabilization, an inflow sedimentation pond, and inflow wetland treatment. A technical and economic analysis showed that oxygenation was the most effective control strategy.

Table 4.7. Different possibilities for in-lake techniques in deep and shallow lakes (Orig.)

| Management technique | Deep lakes | Shallow lakes |
|---|---|---|
| • Epilimnetic mixing | Useful–prevention | Conditional use |
| • Destratification | Conditionally useful | Use only if periodic stratification |
| • Hypolimnetic mixing | Useful | No use |
| • Layer aeration | Possible | No use |
| • Biomanipulation | Difficult | Successful |
| • Phosphorus precipitation | Expensive | Efficient |
| • Sediment capping | Difficult | Feasible |
| • Sediment dredging | Difficult | Feasible |
| • Selective off-takes + curtains | Useful | |
| • Hypolimnion siphoning | Feasible | |

Lake Alte Donau, a shallow lake in Vienna, was restored by a combination of internal and external measures (Donabaum et al., 1999). Watershed measures concentrated on minimizing the nutrient load from contaminated groundwater, from stormwater, and from a large number of waterfowl. The in-lake methods used included enhanced water exchange, chemical flocculation, and nitrate oxidation of the sediments with macrophyte re-colonization and stocking with fish predators for biomanipulation. Zalewski (1999) restored shallow Polish lakes using rehabilitation of buffer zones, naturalization of river channel morphology, regulation of the flow regime of the incoming river to decrease the nutrient supply, creating wetland systems and applying a variety of biomanipulation techniques.

In addition to the selection of appropriate methods, consideration also should be given to the possibilities of zoning their application within a waterbody. As an example, recreation should be located far away from water abstraction sites, in-lake methods should be concentrated on the smaller volume of water taken for drinking water treatment, etc. One such "zoning" brought to an extreme, as developed in China, was given above in the section on innovative approaches.

The conclusion to be emphasized is that these examples should not just be followed, but should also lead to an analysis of the particular situation, taking into considerations a multitude of methods. Considerations of sustainability give clear preference to pollution prevention methods, rather than correction approaches. Mastering methods and considering the range of their possible applications has typically led to the selection of the most suitable combination of approaches, in contrast to continuing to use one or a few "well proved" methods.

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*Chapter 5*

# THE USE OF MATHEMATICAL MODELLING IN LAKE AND RESERVOIR MANAGEMENT

Models are being used increasingly in environmental management, primarily because they are a good tool for quantitatively relating the impacts on an ecosystem with the consequences for the state of the ecosystem. Aquatic ecosystem models have particularly been developed during the last few decades, and it is not surprising they have found wide application in lake and reservoir management efforts.

The extensive use of mathematical models in water quality management can be seen from the survey by Alasaarela et al. (1993). They assembled questionnaires from 100 institutions. The results document the use of 105 models applied in 800 situations and requiring approximately 500 person work years. The average modelling team consisted of four persons.

This chapter describes what the models represent and how they are structured (Section 5.1), and how they can be used as management tools (Section 5.2). Section 5.3 presents modelling methodology, followed by guidance on how to select appropriate management models (Section 5.4). The bulk of the chapter presents to the reader models applicable to various management problems (Section 5.5), with a focus on eutrophication models (Section 5.5.1), toxic substance models (Section 5.5.2), and then successively on models of acidification (Section 5.5.3), wetlands (Section 5.5.4) and lake fisheries (Section 5.5.5). Special attention is given to one management option, biomanipulation (Section 5.5.6). The complex water quality models most commonly used are discussed in Section 5.5.7. Finally, recent models dealing with environmental risk assessment are discussed (Section 5.5.8).

## 5.1 MODELLING AND ITS ELEMENTS

The rationale for the use of ecological management models is demonstrated in Figure 5.1. Urbanization and industrial development have had increasing impacts on the environment. Energy and pollutants are released into ecosystems, where they may cause rapid growth of algae or bacteria, eliminate species, or alter the entire ecological structure. An ecosystem is extremely complex, and it is an overwhelming task to predict the environmental effects that emissions, for example, will have on it. It is under these conditions that models become useful. With sound ecological knowledge, it is possible to extract the features of an ecosystem involved in a pollution problem, in order to form the basis of the ecological model. As indicated in Figure 5.1, the resulting model can be used to select the environmental technology best suited for the solution of specific environmental problems, or for legislation that reduces or eliminates the emission problems.

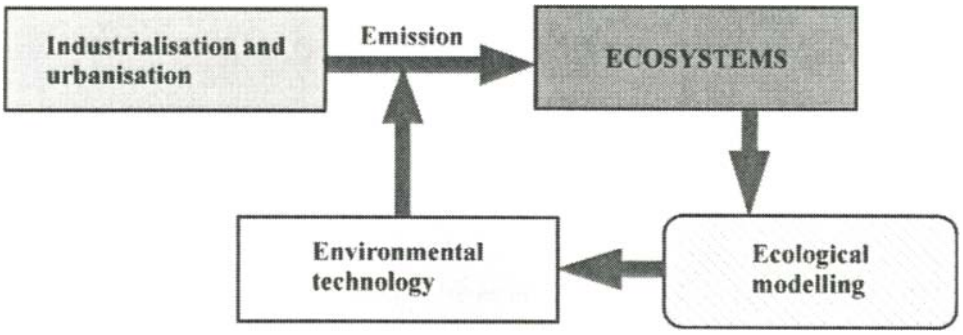



Fig. 5.1. The use of environmental models in management around 1970.

Figure 5.1 represents an introduction to ecological modelling as a management tool around 1970. The environmental management of today is more complex, and must consider such elements as environmental technology, cleaner technology and ecological engineering or ecotechnology (Mitsch and Jørgensen, 1988; Straškraba, 1993) as alternatives to present technology, in combination with environmental legislation. Ecotechnology is applied especially to solve problems of nonpoint or diffuse pollution originating from agricultural runoff and acidification. The importance of nonpoint pollution was only seriously appreciated beginning around 1975. Furthermore, global environmental problems play a more important role today than twenty years ago. The green house effect and ozone depletion, for example, are widely discussed today, and several international conferences on the governmental level have taken first steps toward the use of international standards to solve these and other crucial problems of a global nature. Figure 5.2 attempts to illustrate the more complex picture of environmental management today.

Ecological models may be compared to geographical maps (which are also models). Different types of maps serve different purposes. There are maps for airplanes, for ships, for cars, for railways, for geologists and archeologists, etc. They are all different, because they want to focus on different objects. They are also available at different scales, based on the application of the map and on the underlying knowledge. Furthermore, a map never contains all the details for a considered geographical area, because it would be irrelevant and disrupt the main purpose of the map. If a map contained all details, for example, including the positions of all cars at a given moment, the map would be invalidated very rapidly, since they would have moved to new positions in the meantime. Thus, a good map should contain only the knowledge relevant for the map user.

In the same manner, an ecological model focuses only on the objects and processes of interest for the problem considered. It would detract from the main objectives of a model to include too many irrelevant details. There are many different ecological models of the same ecosystem, as the model edition is selected on the basis of the modelling goals.

The field of ecological and environmental modelling has developed rapidly during the last two decades due essentially to the following factors:

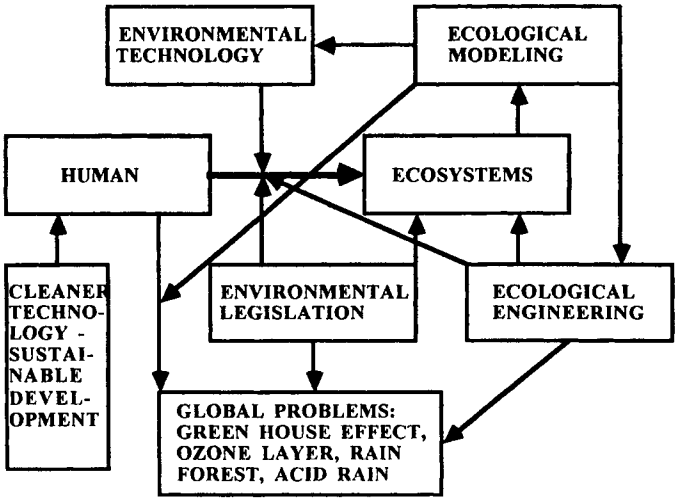



Fig. 5.2. The concept behind the use of environmental models in environmental management. Environmental management today is very complex and must apply environmental technology, alternative technology and ecological engineering or ecotechnology. In addition, global environmental problems have an increasing role. Environmental models can be used to select environmental technology, environmental legislation and ecological engineering.

- The development of computer technology, enabling us to handle very complex mathematical systems.
- A general understanding of pollution problems, including the knowledge that a complete elimination of all pollution ("zero discharge") is not feasible, but rather that a proper pollution control with often-limited economic resources requires serious considerations of the influence of various pollution impacts on ecosystems.
- Our knowledge of environmental and ecological problems has increased significantly. We have particularly gained more knowledge about quantitative relations in ecosystems, and between ecological properties and the environmental factors.

The goal of this chapter is to demonstrate to lake and reservoir managers, limnologists, chemists, sanitary engineers and other interested parties the possibilities that mathematical modelling offers as a tool to support management decisions. More technical modelling issues are discussed by other researchers, including Jørgensen (1983), Orlob (1983), Straškraba and Gnauck (1985), Jørgensen and Gromiec (1989), Jørgensen (1994) and Chapra (1997).

The main focus is a description and survey of various water quality models, and how they can be used to solve different management problems (also see Section 5.5, which presents an overview of models focusing on different problems of management interest). Models are classified in terms of their usefulness in water quality management efforts in Section 5.5, rather than to classical model categories (e.g., see Jørgensen, 1994). Methodological approaches used in different models will be mentioned only when an approach

restricts their use, or to illustrate the spectrum of models applied. Existing models that have been proved useful in decision-making processes are emphasized. Many other research-oriented lake and reservoir models exist, and are discussed elsewhere, including Straškraba (1994), Jørgensen (1994), Straškraba (1995) and Jørgensen et al. (1995).

### *5.1.1 Elements of Environmental Models*

In its mathematical formulation, an environmental science model consists of five primary components:

1. *Forcing functions, or external variables*, which are functions or variables of an external nature that influence the state of an ecosystem. In a management context, the problem to be solved often can be reformulated as follows: If certain forcing functions are varied, how will this influence the state of an ecosystem? The model can be used to predict what properties of the ecosystem will change when the forcing functions are varied with time. The forcing functions under human control are often called *control functions*. The control functions in ecotoxicological models, for example, include the inputs of toxic substances to the ecosystem while, for eutrophication models, the nutrient inputs to a waterbody are control functions. Other forcing functions of interest could include climatic variables, which influence the biotic and abiotic components, and the ecosystem process rates. The latter are forcing functions that usually cannot be controlled by humans.

2. *State variables* describe, as the name indicates, the state of an ecosystem. The selection of state variables is crucial to the model structure, although the choice is sometimes obvious. For example, if we want to model the bioaccumulation of a toxic substance, the state variables should be the primary organisms in the most important food chains and the concentrations of toxic substance in the organisms. For eutrophication models, the state variables would at least include the concentrations of nutrients and phytoplankton. When the model is used in a management context, the values of state variables predicted by changing the forcing functions can be considered as the model results, because the model will contain and utilize the relations between the forcing functions and the state variables.

3. *Mathematical equations* are used to represent the biological, chemical and physical processes that occur in an ecosystem. They describe the relationship between the forcing functions and state variables, and between the state variables. The same types of processes may be found in many different environmental contexts, which implies the same equations can be used in different models. It does not imply, however, that the same process is always formulated by use of the same equation. In fact, a considered process may be better described by another equation because of the influence of other factors. Further, the number of details needed or wanted to be included in the model may differ from case-to-case, due to a difference in complexity of the ecosystem and/or the problem being considered. Some modellers refer to description and mathematical formulation of processes as submodels. A comprehensive overview of submodels is provided by Jørgensen (1994) and Jørgensen et al. (1991).

4. *Parameters* are coefficients in the mathematical representation of processes occurring in an ecosystem. They may be considered constant for a specific ecosystem or part

of an ecosystem. In causal models, the parameter will have a scientific definition (e.g., the excretion rate of cadmium from a fish). Many parameters are not indicated in the literature as constants, however, but rather as ranges. Even these values are of great value in parameter estimation, however, as discussed further in the following sections. Jørgensen et al. (1979) and Jørgensen (1991) provide a comprehensive collection of parameters often considered in environmental science and ecology. Our limited knowledge of parameters is one of the weakest points in modelling, however, as will be discussed further below. Further, the application of parameters as constants in ecosystem models is unrealistic, due to the many feedback processes in real ecosystems. The flexibility of ecosystems is inconsistent, and often not well represented, with the application of constant parameters in a model. A new generation of models that attempts to use parameters that vary on the basis of some ecological principles is a possible solution to this problem, but further work in this area is absolutely needed before an improved modelling procedure reflecting the processes in various ecosystems in realistic manner can be developed.

5. *Universal constants*, such as the gas constant and atomic weights, are also used in most ecosystem models.

Models can be defined as formal expressions, in mathematical terms, of the relations between essential elements of a problem. The first recognition of a problem is often verbal, and may be recognized as an essential preliminary step in the modelling procedure. A verbal model, however, is often difficult to visualize. Thus, it is more conveniently translated into *a conceptual diagram* containing the state variables, the forcing functions, and how they are interrelated by the mathematical formulation of processes.

Figure 5.3 illustrates a conceptual diagram of the nitrogen cycle in a lake. The state variables are nitrate, ammonium (which is toxic to fish in the un-ionized form of ammonia), and nitrogen in phytoplankton, zooplankton, fish, sediment and detritus.

The forcing functions in Figure 5.3 are the inflows and outflows, the nitrogen concentrations in the inflows, solar radiation and temperature (which is not shown on the diagram, but influences all the process rates). The arrows in the diagram illustrate the processes, as formulated with mathematical expressions.

### 5.1.2 Modelling Procedure

Three significant steps in the modelling procedure, calibration, verification and validation, are defined in this section.

- *Verification* is a test of the *internal logic* of a model. Typical questions in the verification phase include: Does the model react as expected? Is the model stable over the long term? Does the model follow the law of mass conservation? Verification is largely a subjective assessment of the behavior of a model. To a large extent, the verification will go on during the use of the model before the calibration phase.
- *Calibration* is meant to find the best agreement between computed and observed data by variation of some selected parameters. It may be carried out by trial-and-error, or by use of software developed to find the parameters that give the best fit between observed

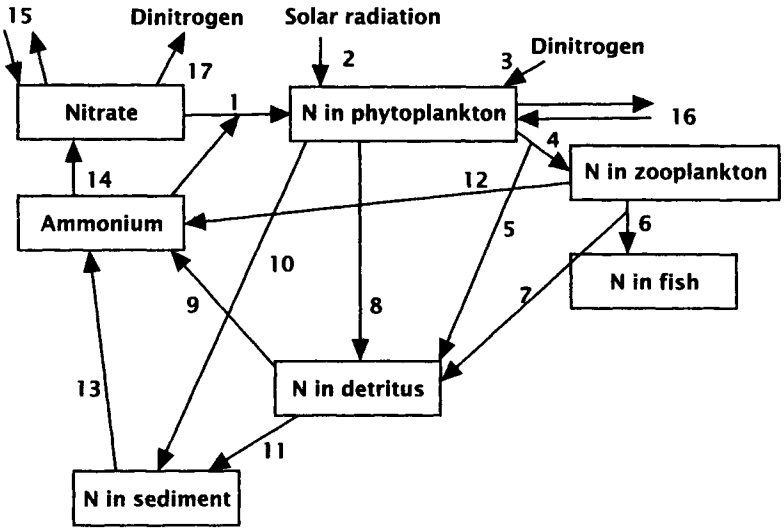



Fig. 5.3. The conceptual diagram of the nitrogen cycle in an aquatic ecosystem. The processes are: 1—uptake of nitrate and ammonium by algae; 2—photosynthesis; 3—nitrogen fixation; 4—grazing with loss of undigested matter; 5, 6 and 7—predation and loss of undigested matter; 8—mortality; 9—mineralization; 10—settling of algae; 11—settling of detritus; 12—excretion of ammonium from zooplankton; 13—release of nitrogen from the sediment; 14—nitrification; 15 and 16—inputs/outputs; and 17—denitrification.

and computed values. In some static models, and in some simple time-varying models which contain only a few well-defined or directly measured parameters, calibration may not be required.

- *Validation.* Validation consists of an objective test on how well the model output fits the data. The selection of possible objective tests is dependent on the scope of the model, although the standard deviations between model predictions and observations, and a comparison of observed and predicted minimum or maximum values of a particularly-important state variable, are frequently used. If several state variables are included in the validation, they may be given different weights.

A validation procedure, if properly carried out, can be used to determine the standard deviation of the predictions or prognosis. Model results used in environmental decisions should always be accompanied by an indication of the standard deviation of the prognosis. It also is important to clarify the meaning of this standard deviation to the decision-maker, for example, by comparison of the expected average results and the worst case situation. The modeller should give recommendations as to how to use the model results and standard deviations in their proper context. Decision-makers often use standard deviations incorrectly when they were presented to them; namely, as numbers which indicate how much the costs can be reduced without any harmful effects on environmental quality.

## 5.2 MODELS AS MANAGEMENT TOOLS

A management problem to be solved can often be formulated as follows: If certain forcing functions (management actions) are varied, what will be their influence on the state of an ecosystem? The model is used to answer this question or, in other words, to predict what will change in a system when the control functions that are managed by humans are varied over time and space.

Typical *control functions* are the consumption of fossil fuel, regulation of water level in a river by a dam, discharge of pollutants, or fisheries policy.

A certain class of environmental management models are called control models. They differ from other such models by the content of the following elements:

- A quantitative description of control processes,
- A formalization of objectives and evaluation of achievements.

The difference between control models and other environmental management models can best be illustrated with an example. The eutrophication model presented in Figure 5.3 can be used as a management model. If the model response to various inputs of nitrogen is obtained, the model outputs for the corresponding scenarios also are obtained. Among these scenarios, the manager can select the one scenario, from among all the possibilities, that he or she prefers from an ecological–economic perspective.The model is used as an environmental management tool. However, this is not a control model. Details of this type of model are presented in Section 5.5.

In the case of eutrophication, the management problem often is: Which control method or combination of methods should be selected from among the existing possibilities? This question can be answered by comparison of the corresponding scenarios obtained by the model runs.

It is important that, to a certain extent, the manager take part in the entire development of a management model, since he will ultimately define the modelling objectives and select the modelling scenarios. The success of the application of a management model, to a large degree, is dependent on an open dialog between the modeller and the manager.

A further complexity is the construction of ecological–economic models. As we gain more experience in constructing ecological and economic models, more and more of them will be developed. It is often feasible to find a relation between a control function and economic parameters. If a lake is a major water resource, an improvement in its water quality will inevitably result in a reduction in the treatment costs of drinking water if the same water quality is to be provided. It is also possible to sometimes relate the value of a recreational area to the number of visitors, and to how much money they spend on average in the area. In many cases, however, it is difficult to assess a relationship between the economy and the state of an ecosystem. For example, how can we assess the economic advantages of an increased transparency in a waterbody? Ecological–economic models are useful in some cases, but should be used with caution, and the relations between the economy and environmental conditions critically evaluated, before the model results are applied.

Data collection is the most expensive component of model construction. For many lake models, it has been found that the data collection needed comprises 80–90% of the total

model costs. Because complex models require much more data than simple ones, the selection of the complexity of environmental management models should be closely related to the costs involved in the environmental problem to be solved.

It is not surprising, therefore, that development of the most complex environmental management models have generally been limited to large ecosystems, where the economic involvement is substantial.

The predictive capability of environmental models can always be improved in a specific case by expansion of the data collection program, and by a correspondingly increased model complexity, provided that the modellers are sufficiently skilled to know in which direction further expansion of the entire program must develop to order to improve the model's predictive capabilities.

The relation between the economy of the project and the accuracy of the model is presented in Figure 5.4. The reduction in the discrepancy between model predictions and reality is lower for the next dollar invested in the project. But it is also clear from the shape of the curve that the associated errors cannot be completely eliminated. All model predictions have a standard deviation associated with them. This fact is not surprising to scientists, but it often is not understood or appreciated by decision-makers, to whom the modeller typically presents his or her results.

Engineers use safety factors to assure that a building or a bridge will last for a certain period of time, with a very low probability of breakdown, even under extreme conditions. No reputable engineer would propose using a smaller, or no, safety factor to save some concrete and reduce the costs. The reason is obvious: Nobody would want to be responsible for even the smallest probability of a building or bridge collapse.

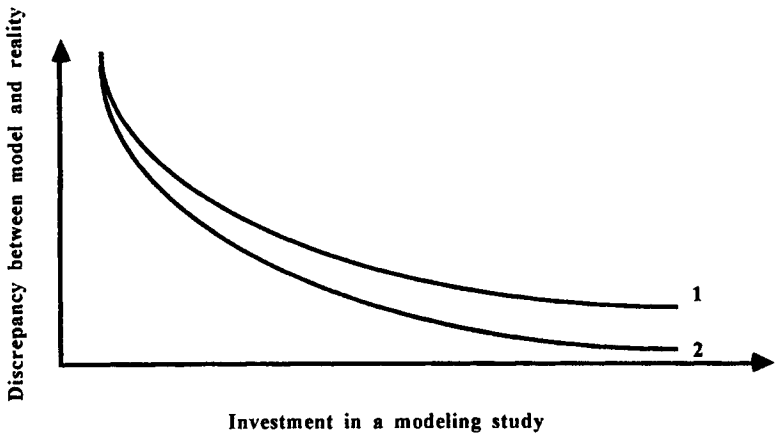



Fig. 5.4. The more a modeller invests in a model and in data collection, the closer he/she will come to realistic predictions. However, the modeller will always gain less for the next dollar invested, and will never be able to give completely accurate predictions. The curve (1) represents a less experienced modeller, while curve (2) gives the same relationship for a more experienced modeller.

When decision-makers are going to make decisions on environmental issues, the situation is strangely different. Decision-makers in this situation want to use the standard deviation to save money, rather than assuring a high environmental quality under all circumstances. It is the modeller's duty, therefore, to carefully explain to the decision-maker all the consequences of the various decision possibilities. A standard deviation of a prognosis for an environmental management model can, however, not always be translated into a probability, because we do not know the probability distribution. It might be none of the common distribution functions, but it is possible to use the standard deviation qualitatively or semi-quantitatively, translating the meaning of the model results with the use of words. Civil engineers are more or less in the same situation, and have been successful in the past in convincing decision-makers of appropriate steps to be taken in various situations. There is no reason that environmental modellers cannot do the same.

It is often advantageous to first attack an environmental problem with the use of simple models. They require much fewer data, and can give the modeller and decision-maker some preliminary results. If the modelling project is stopped at this stage for one or another reason, a simple model is still usually better than no model at all, because it will at least give a survey of the problem.

Simple models, therefore, are good starting points for the construction of more complex models. In many cases, the construction of a model is carried out as an iterative process, and a step-wise development of a complex model is illustrated in Figure 5.5. As previously mentioned, the first step is development of a conceptual model. It is used to get a survey of the processes and state variables for the ecosystem of concern. The next step is development of a simple calibrated and validated model. It is used to establish a data collection program for a more comprehensive effort closer to the final selected version. As shown in Figure 5.5, however, the third model version will often reveal specific model weaknesses, the elimination of which is the goal of the fourth version of the model. At first glance, this may seem to be a very cumbersome procedure. However, because data collection is the most expensive part of modelling, constructing a preliminary model for optimization of the data collection program will ultimately require fewer financial resources. In fact, Figure 5.5 can be considered a formalization of the iterative procedure that many modellers are forced to use under most conditions, and planning these steps at an initial phase of the project always is advantageous.

A first, simple mass-balance scheme is recommended for biogeochemical models. The mass balance will indicate what possibilities exist for reducing or increasing the concentration of a chemical or pollutant, which is a crucial issue for environmental management.

Point sources of pollution are usually easier to control anthropogenic nonpoint sources which, in turn, are more easily controlled than natural pollutant sources, as shown in Table 5.1. Distinction can be made between local, regional and global pollutant sources. Because the mass balance indicates the relative quantities from each source, it is possible to identify which sources should receive the initial attention (e.g., if a nonpoint regional source of pollutants is dominant, it would be pointless to concentration first on eliminating small, local point sources, unless the latter also might have some political influence on regional decisions).

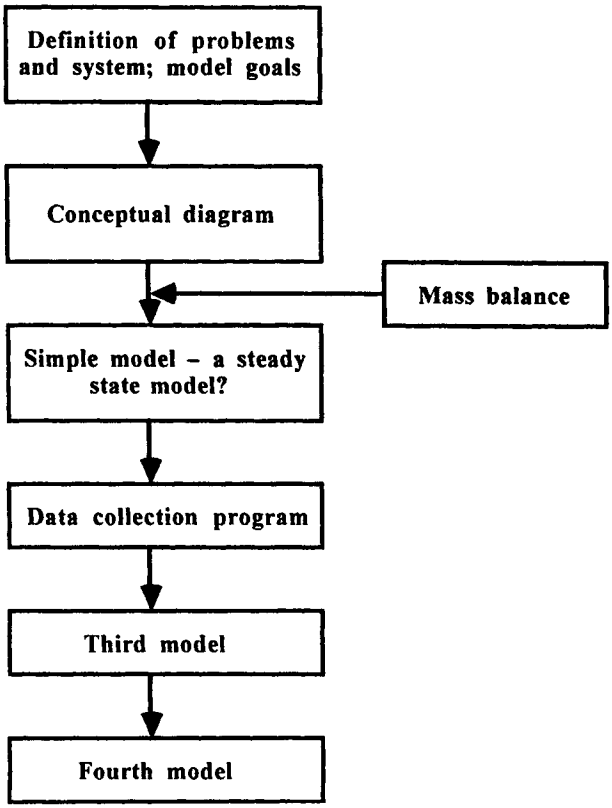



Fig. 5.5. Schematic of the development of a complex management model.

Table 5.1. Examples of water pollutant sources

| Source | Examples |
|---|---|
| Point sources | Wastewater (nitrogen, phosphorus, biochemical oxygen demand); sulphur dioxide from fossil fuels; discharges of toxic substances from industries |
| Nonpoint man-made sources | Agricultural use of fertilizers; deposition of lead from vehicles; contaminants in rain water |
| Nonpoint natural sources | Runoff from natural forests |

It has already been recognized that the modeller and the decision-maker should communicate with each other. It is recommended, in fact, that the decision-maker be invited to follow the model construction process from its very first phases, in order to become acquainted with the model strength and limitations. It is also important that the modeller and the decision-maker formulate together the model objectives and interpret the model results.

Holling (1978) has demonstrated how such teamwork can be developed and used phase-by-phase. Although his recommendations are not reiterated here, it is recommended that a modelling team should acquaint themselves well with the procedures outlined by him. The conclusions are clear: The modeller and decision-maker should work together in all phases of the modelling exercise. Having the modeller first build a model, and then transfer it to a decision-maker accompanied by a small report on the model, is not recommended.

Communication between the decision-maker and the modeller can be facilitated in many ways, and it often is the primary responsibility of the modeller to do so. If a model is built as a menu system, as presented by Mejer (1983), it might be possible to teach the decision-maker how to use the model in only a few hours, thereby also increasing his or her understanding of the model and its results. If an interactive approach is applied, it also is possible for the decision-maker to visualize a wide range of possible decisions. The effect of this approach is increased by the use of various graphic methods to illustrate the best possible decision in regard to what happens with the use of various management strategies. It is recommended under all circumstances that time be invested in developing a good graphic presentation of the model results for the decision-maker. Even if he or she has not been clearly informed about a model project through all its phases, the decision-maker will not necessarily understand the background and assumptions of all the model components. Thus, it is important that the model results, including the main assumptions, shortcomings and standard deviations underlying them, are carefully presented with the use of an illustrative method.

The use of expert systems and decision-support systems to facilitate communication between modellers and managers is discussed in Section 5.4. It is clear that we are not yet sufficiently advanced in environmental modelling efforts to solely use model results to define management programs, even utilizing expert systems and decision-support systems. A model should never be used in this way by a decision-maker, but rather should be considered only one useful tool in the management decision-making process. This implies that modelling results should be clearly and illustratively presented, and be considered a significant component in discussions about selecting specific courses of action. Other elements to be considered in such discussions would include potential side effects, interpretation of model predictions, and the implications of the accuracy of the prognosis.

A good environmental model can be a powerful tool in the decision-making process for management actions. A wide range of environmental problems has been modelled to varying degrees over the last 10–15 years. They have generally been of major assistance to decision-makers. With the continuing rapid growth in the use of environmental models, the situation will only improve in the future. However, we have not achieved the same level of experience for all environmental problems.

The use of models in environmental management is definitely growing. They have been widely used in several European Countries, in North America and in Japan. Further, environmental agencies in more and more countries are making use of model applications. Through such journals as *Ecological Modelling* and the *International Society for Ecological Modelling* (ISEM), it is possible to follow the progress in the field. This "infrastructure" of the modelling field facilitates communication and accelerates the exchange

of experiences, thereby enhancing the growth of the entire field of ecological modelling. It appears there will soon be a need for a model "bank" or data base, where users can obtain information about existing models, their uses, characteristics, etc. At the same time, it is not possible to blindly transfer a model from one case study to another, as is reiterated in this book. It also is difficult to transfer a model from one computer to another unless it is exactly the same type of computer. However, it is often very helpful to learn from experiences of others involved in similar modelling situations in other parts of the world. Journals such as those mentioned above facilitate such exchanges.

The issue of the generality of models also deserves discussion. Few models have been used on multiple case studies, in order to gain wide experience on this important matter. The eutrophication model presented by Jørgensen (1994) has been used on a wide range of lakes in the temperate and the tropical zone, as well as for shallow and deep lakes, and even fjords. The experience gained in these case studies is illustrative, but does not necessarily represent general properties of models. A further discussion on the generality of models is presented in Section 5.4.

Management models are used in water quality management of lakes and reservoirs for the purposes outlined below, with reference to the type of management models:

*General—*

- To estimate pollution sources and loads in the watershed, using simple calculation models.

*For lakes and existing reservoirs—*

- To predict possible future water quality conditions when control functions in the watershed are altered by human activities,
- To provide estimates for decisions between different water quality options for use in long-term planning,
- To support short-term operational management decisions regarding water quantity and quality,
- To optimize sampling schedule investigations and control of water quality,
- To assess the environmental risks associated with discharge of micro pollutants.

*Prior to reservoir construction—*

- To estimate budgets of major water quality components of rivers that will enter the reservoir, as well as in the reservoir and the reservoir outflows,
- To provide reasonable estimates between alternative construction sites, dam heights, and outflow and outlet structures so that management decisions are supported,
- To predict conditions in future reservoirs, and the consequences of different management options on water quality within the watershed.

## 5.3 METHODOLOGY

It is clear from the previous discussions in this chapter that selection of model complexity is a matter of balance. On the one hand, it is necessary to include the state variables and the processes essential for the problem being addressed. On the other hand, it is important

not to make the model more complex than the data set can accommodate. Our knowledge of ecosystem processes and state variables, together with the relevant data sets, will determine the selection of model complexity. If the knowledge is poor, the model will be unable to give many details, and will have a relatively high level of uncertainty. If we have much knowledge of the system and the problem we want to model, a more detailed model can be developed, with a relatively lower degree of uncertainty. A modeller always should identify and present the shortcomings and uncertainties of a model, and not pretend that the model is a complete picture of reality in *all* its details. In the hands of a competent researcher, a model can be a fruitful instrument to test hypotheses, but only if the limitations of the model are fully acknowledged.

It should not be forgotten that models have always been applied in science. The difference between previous and present models is that today, with modern computer technology, we are able to work with very complex models. At the same time, however, it is a continuing temptation to construct models that are too complex. It is easy to add more and more equations and state variables to a computer program, but much harder to get the data needed for calibration and validation of the model. Even if very detailed knowledge about a problem is available, it is not possible to develop a model capable of accounting for the complete input–output behavior of a real ecosystem and that is valid for all frames (Ziegler, 1976). This type of model is called "the base model" by Ziegler, and would be very complex and require such a large number of computational resources that it would be almost impossible to simulate. The base model of an ecological problem will never be fully known because of the complexity of natural ecosystems, and our inability to observe all its states. Thus, a model may be made more realistic up to a point, by adding more connections. However, addition of new parameters after that point is reached does not contribute further to an improved simulation capacity. Indeed, more parameters imply more uncertainty because of possible lack of information about the flows the parameters quantify.

For a given quantity of data, the addition of new state variables or parameters beyond a certain model complexity does not further our ability to model an ecosystem, but only adds to the unaccountable uncertainty, as visualized in Figure 5.6. The relationship between knowledge gained through a model and its complexity is shown for two levels of data quality and quantity. The question that can be formulated with relation to this figure is: *How can we select the complexity and the structure of the model to assure the optimum for knowledge gained or the best answer to the question posed by the model*?

Costanza and Sklar (1985) have examined 88 different models, and were able to show that the theoretical discussion behind Figure 5.6 is actually valid in practice. Selection of the right degree of complexity is of great importance for environmental and ecological models. This is possible on the basis of presented models and literature discussions (e.g., see Jørgensen, 1994). However, the selection will always require that the application of these methods is combined with a good knowledge of the system being modelled. The methods must work hand-in-hand with an intelligent answer to the question: Which components and processes are most important for the problem of concern? Such an answer is important even in the right use of the aforementioned methods. Thus, the conclusion

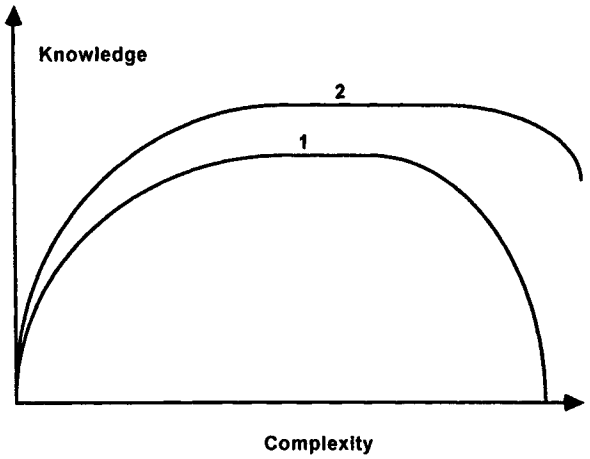



Fig. 5.6. Knowledge plotted versus model complexity measured, based on the number of state variables. The knowledge increases up to a certain level. Increased complexity beyond this level will not add to the knowledge gained about the modelled system. At a certain level, the knowledge might even be decreased, due to uncertainty caused by a too-high number of unknown parameters. (2) corresponds to an available data set, which is more comprehensive or has better quality than (1). Thus, the knowledge gained and the optimum complexity is higher for data set (2) than for (1). Reproduced from Jørgensen.

is: Know the ecosystem and the problem being addressed before a model is selected, including the model complexity. We do not need all details to get a proper overview and a holistic picture of a given ecosystem. Some details are needed, as is an understanding of how the ecosystem works on the systems level. Thus, the conclusion is that, although we can never know everything about an ecosystem needed to make a complete model (i.e., inclusion of all details), good, workable models that expand our knowledge of ecosystems can be produced, particularly of their properties as systems. Ulanowicz (1979) points out that a correct, very precise predictive model cannot be constructed. Thus, it would be most fruitful to build a model that illustrates general trends, taking into account the probabilistic nature of the environment.

Further, models are, and can be applied as, management tools (e.g., see Straškraba and Gnauck, 1985; Jørgensen, 1986b; Chapra and Canale, 1991; Straškraba et al., 1995). All in all, models should be considered only as tools—tools to provide an overview of complex ecosystems. For a complex system, a few interactive state variables already make it impossible to get an intuitive or logical overview on how a system may react to perturbations or other changes. In such cases, models are good tools to help obtain an overview or picture of the properties of an ecosystem on the systems level.

There are only two possibilities for getting around the dilemma rooted in the model complexity issue: Either limit the number of state variables in the model or describe the system by the use of holistic methods and models, preferably by using higher-level sci-

entific laws. *The trade off for the modeller is between knowing much about little, or little about much*!

More complex models require more data and more knowledge about a lake or reservoir. They imply higher costs, but they should be justified by the importance of the ecosystem being studied. Thus, it is recommended to proceed in a step-wise manner toward a more complex model, learning from simpler models before the complex model is constructed. A procedure for this step-wise development could proceed along the following lines:

- *Mass balances and lake morphology* is the first component to be considered in this procedure. Assessment of the sources of nutrient inputs, and the area, depth and water retention time of a lake will help focus attention on the main problems a lake or reservoir.
- *Set up a simple model*, an example being a phosphorus loading model of the type presented by Vollenweider (1976; OECD, 1982; Ryding and Rast, 1989) for eutrophication. This type of model gives guidance on the possible directions to solutions to the problem. For example, what nutrient limits or controls the maximum algal biomass? Which are the primary nutrient sources in the drainage basin? Will treating or conveying wastewaters produce the expected results? It may be feasible to first develop hypotheses, even at this early stage, on how to solve the problems, and even make the first coarse predictions. They will probably help identify the important factors in lake and reservoir management, and indicate which processes further modelling work should consider. The results, for example, will help define whether the point or nonpoint nutrient sources are of most importance for achievement of the desired results and, therefore, which of the two sources it is necessary to model with the greatest accuracy. The results also may indicate the relative importance of the nutrients or toxic substances stored in the lake bottom sediments, or whether or not the water retention time is an important factor that should be determined very accurately. Of course, it is important to compare these initial hypotheses and predictions with the results obtained with the use of more elaborate models at a later stage. If the first predictions are not confirmed, it should be possible to properly explain the discrepancies.
- *Set up a conceptual diagram.* The procedures for developing models are presented in Section 5.2. It includes defining the modelling goal, and its confinement in time and space. The next step is to develop a conceptual diagram, which should ideally contain all the important state variables and processes. The selection of the important components can be made on the basis of mass balances (i.e., the important processes are selected on the basis of their contribution to the mass balance of the primary components). For a eutrophication model, this means the balance of the nutrients, or at least the limiting nutrient, is of concern. It also is necessary to consider the characteristic features of a lake or reservoir (i.e., Which species are dominant? What is the dominant hydrological pattern?).
- It is usually important to discuss and *consider the model complexity* at this stage, although it should be reconsidered when the model has been calibrated and validated.
- *Assess the data requirements.* This step is in accordance with the procedure previously presented above in Section 5.2. However, the economic considerations should be added in this context. Which scale should be used for a given problem, knowing the economic

implications? It is possible to assess the optimum funding level that should be invested in a modelling project, including the data collection, if all the economic parameters of the project are known, such as wastewater treatment costs, the economic value of a lake or reservoir as a recreational area, etc. It would be to speculative to guess the economic value of recreational areas in this context. Further, how the recreational value can be calculated is also often an open question. *Experience has shown that 2–10% of the investment in wastewater treatment, for example, could be used for developing a model* that could be used to select the right management strategy. Based on this rough rule-of-thumb, it is now possible to assess whether the model selected (according to points 3 and 4 below) could be justified from an economic perspective, or whether it is possible to expand the project or if it is necessary to reduce it. When the model has been developed, calibrated and validated, the standard deviation of the model will be determined, and the model can then be tested to determine whether or not it has sufficient accuracy to be used to help choose between two or more management strategies. Modelling is an iterative process and, therefore, the possibility that a developed model cannot be used in a given management context, and that a more complex model may be needed, cannot be excluded. It is important to confront managers with this fact in the initial phase of model development. The manager, therefore, also should be prepared to accept that a project may have to be enlarged, or the model cannot be used, if its accuracy is not sufficient for the purposes. On the other hand, while an experienced modeller will be able in most cases to estimate the appropriate model complexity at an early stage, unforeseen difficulties during the model development also may lead to a model of insufficient accuracy, even for an experienced modeller. Generally, it is strongly recommended to have an ongoing dialog between the modeller and the manager during the entire development of the model. It is absolutely necessary that the manager is aware of the strengths and limitations of the modelling process, including the expected accuracy of any model being considered. Further, the modeller should attempt to apply the model interactively, in order to develop demonstrations for the manager of the simulation results obtained from different management strategies. This also will open the necessary discussion between the manager and modeller on all the possibilities that a model can offer, including alternative changes in the management strategy.

- *Strictly follow the recommended modelling procedure.* The procedure outlined in Section 5.2 should be carefully followed. All the steps, including verification, calibration and validation, should be followed without exception. Model verification is particularly important. The modeller should take sufficient time to "play" with the model, and test the feasibility of the parameters. Are the parameters in accordance with the literature? Introducing "ecological constraints" into the model also should be considered, as discussed further in Section 5.4. The ability of an ecosystem to introduce other species better suited to the model (i.e., other properties of the biological model components) may provoke the use of a structural dynamic model (see Section 5.6.7). Using the model to optimize the data collection efforts is often recommended, particularly when data collection has a high cost. Kettunen (1988) presented a method to assess this optimum situation, which means that the first complex model is only used to find the optimal data

collection scheme (i.e., to assess which data most beneficial to collect at a given cost) to offer the best support to the model development. When additional data resulting from the first provisional model are available, the final model can be developed, using all the available data.

Another crucial problem is associated with model generality. Can a model used in one lake, for example, be used without change for other case studies? The answer is not a simple yes or no. However, experience shows that simpler models can be used more generally than more complex models. They contain a description of the basic processes characterizing all lakes and reservoirs (e.g., nutrient uptake by phytoplankton, which is dependent on the nutrient concentration and the concentration of phytoplankton; mineralization of detritus). The more complex models inevitably will contain more site-specific process descriptions, which may not be important for all lake and reservoir case studies. Thus, more complex models will generally have to be modified from case to case. This is illustrated in Table 5.2, in which the experience gained with the general use of a eutrophication model containing 17–20 state variables is presented.

Table 5.2. Survey of eutrophication studies, based upon the application of a modified Glumsø model

| Ecosystem | Modification | Level* |
|---|---|---|
| Glumsø, version A | Basis version | 6 |
| Glumsø, version B | Nonexchangeable nitrogen | 6 |
| Ringkøbing Firth | Boxes, nitrogen fixation | 5 |
| Lake Victoria | Boxes, thermocline, other food chain, herbivorous fish | 4 |
| Lake Kyoga | Other food chain | 4 |
| Lake Albert | Boxes, thermocline, other food chain | 4 |
| Lake Fure | Boxes, nitrogen fixation, thermocline | 3 |
| Lake Esrom | Boxes, silicon cycle, thermocline | 4 |
| Lake Gyrstinge | Level fluctuations, sediment exposed to air | 4–5 |
| Lake Lyngby | Basis version | 6 |
| Lake Bergunda | Nitrogen fixation | 2 |
| Broia Reservoir | Macrophytes, 2 boxes | |
| Lake Great Kattinge | Resuspension | 5 |
| Lake Svogerslev | Resuspension | 5 |
| Lake Bue | Resuspension | 5 |
| Lake Kornerup | Resuspension | 5 |
| Lake Balaton | Adsorption to suspended matter | 2 |
| Roskilde Fjord | Complex hydrodynamics | 4 |
| Stadsgraven, Cph† | 4–6 interconnected basins | 5 (Level 6: 93) |
| Internal lakes of Cph† | 5–6 interconnected basins | 5 |

*Level 1: Conceptual diagram selected; Level 2: Verification carried out; Level 3: Calibration; Level 4: Calibration using additional intensive measurements; Level 5: Validation by use of an independent data set; Level 6: Validation of a prognosis.

†Cph—Copenhagen.

Table 5.3. Lake models for different environmental problems

| Problem | Modelling effort |
|---|---|
| Hydrodynamical problems | 5 |
| Eutrophication | 5 |
| Oxygen balance | 4–5 |
| Bacteriological water quality | 3–4 |
| Heavy metal pollution | 4 |
| Pesticide pollution | 4 |
| Acid rain | 4–5 |
| Fishery | 3 |
| Wetlands | 4 |

Problems of interest for lake and reservoir management include: (1) eutrophication, (2) ecotoxicological effects, (3) acidification, (4) fishery management, (5) the oxygen concentration of lake water, (6) management of wetlands surrounding, (7) the bacteriological quality of the lake water, and (8) hydrodynamical problems related to water discharge, or to the use of lake water for cooling or for production of drinking water.

All eight of these problems have been modelled. Table 5.3 provides an overview of the modelling activity for each of these eight problems on a scale from 0 to 5, where 5 means very intense modelling efforts, 4—intense modelling effort, 3—some modelling effort, 2—a few models have been studied, 1—one or a few studies with not well-calibrated or validated models, and 0—no modelling efforts have been done. The table indicates that all 8 problems have been modelled to some extent, with most of the problems being intensely or very intensely modelled.

Thus, a spectrum of models is available for all eight major lake problems. Selection of the appropriate complexity dictated by the problem, the characteristics of the ecosystems, and the quality and quantity of the available data, as discussed in Section 5.4, is crucial for the modelling results. Thus, it is beneficial that lake ecosystems have been modelled intensely, and that a wide experience with models of differing complexity is available.

## 5.4 SELECTION OF AN APPROPRIATE MANAGEMENT MODEL

The first step in selecting a model is specification of the goals for which the model will be used. The selected model must be one intended to answer the questions of interest, as defined by the lake manager. Another consideration in selecting a model is the specific data available for use with the model. Using the most advanced models is impossible, for example, if the quantity and quality of the water inflows have not been measured. In some cases, it may be possible to obtain data required for the model, but this inevitably involves expenditures of money and time, since most lake inflows vary considerably over time. The availability of personnel skilled in the use of models also can be a significant limitation. Learning the modelling procedure, and understanding the applied basis for model development, can be both difficult and time consuming. Thus, appropriate model selection requires

balancing the importance of the problem, money and time invested in the model development, the available personnel, and the existence of appropriate models.

As emphasized in previous sections, models only provide a gross simplification of reality. Thus, caution is always necessary when considering model results. Moreover, three levels of uncertainties influence even the best models (Hilborn, 1987), including:

- Noise—the natural variability that occurs sufficiently frequently to be routine (various sampling schemes and statistical analyses are required to accommodate this uncertainty),
- The state of nature that is not well known,
- Surprises—unanticipated events (flexible management strategies cope with unanticipated events more effectively than rigid, dogmatic strategies).

These uncertainties are inherent in any complex system, and may occur in any specific case. By proper use of validation techniques, it is possible in most cases to quantify the uncertainty, which is very valuable for interpretation of the model predictions.

Models useful for water quality management may be classified as follows:

- *Simple static calculation models* consisting of algebraic equations or graphs. These models are based predominantly on the statistical elaboration of data sets, sometimes large ones. Thus, they are limited by the extent of the materials covered in the analysis. For example, many models based on empirical relationships between phosphorus and chlorophyll presented in the literature. Examples are presented in Section 5.5, which focuses on applicable models. Thornton et al. (1999) provides details on models for estimating nonpoint source pollution loads.
- *Complex dynamic models* providing analysis of the timing of water quality conditions. Several of the models presented in Section 5.5 belong to this class, and represent a wide spectrum of complexity.
- *Geographical information systems* (GIS) used for problems requiring spatial resolution. This is typical for estimating pollutant loads, particularly for nonpoint source pollution. The basis of GIS is computerized maps and procedures for entry and treatment of spatial data. For example, a specific watershed can be included in the database and corresponding pollution sources indicated. Through the use of models (usually more complex dynamic models), the expected pollution input can be calculated. Several attempts to apply GIS in watershed management (DePinto and Rodgers, 1994), or combine watershed models with reservoir models, have emerged in recent years.
- *Prescriptive models* can be used to calculate water quality conditions without indicating appropriate management options for a given situation. By means of scenario analysis, it is possible to test management alternatives and to predict the potential consequences of water quality. This can be useful in selecting the most appropriate management possibilities. These models are based either on simple static calculations or dynamic simulations, depending on whether a time-independent or time-dependent solution is required.
- *Optimization models* incorporate selection procedures to choose the most suitable option, based on a set of criteria. Such models, which are often based on complex dynamic models, can allow simultaneous analysis of several management alternatives or goals. The major component of an optimization model is called a goal function, which is a function the user seeks to minimize or maximize, including critical water quality

variables (e.g., oxygen concentration in a lake), or the money spent on attaining a specified level of water quality improvement. Optimization with constraints means that some, or all, of the management parameters are limited (i.e., they are forced to remain within specified limits due to natural conditions, management limitations, etc.). The following examples illustrate these kinds of constraints:

1. Inflow phosphorus levels cannot be reduced more than the capabilities of treatment plants or other available reduction methods,
2. It is impossible to mix a waterbody beyond its greatest depth; thus, there is a natural depth of mixing,
3. There are no feasible reasons to reduce the chlorophyll concentrations, or increase the oxygen concentrations, above a certain limit.

The *standard optimization* procedure is used when a management decision involves only one variable (e.g., reservoir outflow), and is determined by one characteristic of this variable (e.g., outflow rate by turbines). The goal in this case, for example, can be to keep the reservoir water level within certain limits. *Multiparameter* optimization also concerns one variable, although the optimal performance is searched for in a multidimensional parameter space. It can be asked, for example, to what extent should the bottom outflow gate of the reservoir be opened, and a turbine run, to keep the water level in the reservoir within certain limits. The constraints are given by the maximum capacity of the bottom outlets and the turbines, and by the minimum and maximum flow rates prescribed for the river downstream of the reservoir. The model formulation with respect to water quality, for example, may be retaining some water quality variable below a certain limit, while also modifying several parameters characterizing the use of various management options. As an example, the model GIRL OLGA is intended for making dynamic, time-dependent estimates of the best combination of the time sequence of using five different management options, each characterized by one parameter that can be manipulated within certain limits. *Multivariable* optimization simultaneously takes into account several variables. Examples would include keeping the oxygen concentration, quantity of algae and the content of organic matter in a lake within certain limits. *Multigoal* optimization is the most complex, with several goals having to be simultaneously achieved. In this latter case, a compromise set is sought, rather than a unique optimal solution, with the model user driven to make a proper selection. A combination of multiparameter, multivariable and/or multigoal formulation also is possible, although such a solution has not yet been attempted for water quality modelling. Up to the present time, more complex formulations have generally been used to address water quantity problems (e.g., optimal operation of a reservoir cascade). Nevertheless, progress in developing optimization techniques is moving forward rapidly, and useful water quality formulations will doubtless be available in the future. Table 5.4 provides a listing of optimization water quality management models.

It is emphasized that optimization procedures only allow selection among the possibilities included in the model, and is limited by the validity of the model, including its assumptions and formulations and its imposed constraints.

Thus, the model conclusions should be used with caution, with the user considering the model limitations, possible inadequacies, and possible insufficiency of input data. The

Table 5.4. Optimization water quality management models

- Dynamic optimization of eutrophication by *phosphorus removal*—used for a Japanese lake (Matsumura and Yoshiuki, 1981).
- Optimal control by *selective withdrawal* (Fontane et al., 1981).
- Optimizing reservoir operation for *downstream aquatic resources*—applied to Lake Shelbyville, IL, USA (Sale et al., 1982).
- GIRL OLGA for cost minimization of eutrophication abatement, using time-dependent selection from five management options (Kalčeva et al., 1982; Schindler and Straškraba, 1982)—applied for several reservoirs in the Czech Republic.
- Stochastic optimization of *water quality* (Ellis, 1987).
- COMMAS for prediction of environmental multiagent system (Bouron, 1991).
- DELWAG-BLOOM-SWITCH for management of eutrophication control of shallow lakes (van der Molen et al., 1994).
- GFMOLP, a fuzzy multiobjective program for the optimal planning of reservoir watersheds (Chang et al., 1996).

model can be run before any decision is taken, and several alternatives investigated. Alternatively, the model can be connected to automatic devices that activate water quality management options. As examples, chlorophyll concentrations and meteorological variables can be automatically recorded and put into a computer model. Short-term predictions by the model can be used to switch water mixing devices on or off, or to specify the intensity of phosphorus purification (see Fig. 5.7).

*Expert systems* use qualitative and quantitative expressions to guide the model user toward relevant answers to complex water quality questions. The major advantage of expert systems is their ability to consider qualitative characteristics, in addition to quantitative characteristics, and to handle complex decision rules. The name of this model group originated from the basis in which these answers are obtained; namely, the judgment of experts in a given field. An interactive mode is available, in which the user interacts with the computer software, selecting questions offered by the computer, obtaining answers and answering the questions. Each expert system is devoted to a particular problem. They are called *empty expert systems* or *expert shells* directed to handling the *knowledge base*, which is the specific decision tree on what to do, how to decide in the given instance, etc. A general review of expert systems applicable to different environmental problems is provided by Hushon (1990), Davis (1993) and Davis and Guariso (1994).

*Decision support systems* (DSS) represent a further extension of expert systems. They incorporate other computer software products relevant for a specific water quality decision problem for which the system was constructed (e.g., simple models, simulation models, optimization models, GIS systems). A graphics package that generates explanatory drawings and texts can be an integral part of DSS, and all the model types mentioned above can be incorporated in the system. The entire decision support system is driven by questions offered by the computer and answers provided by the model user.

DSS were named for their ability to *support* decisions. They are not intended to *make* decisions. Wise, experienced people must always make the final decisions. In order to make

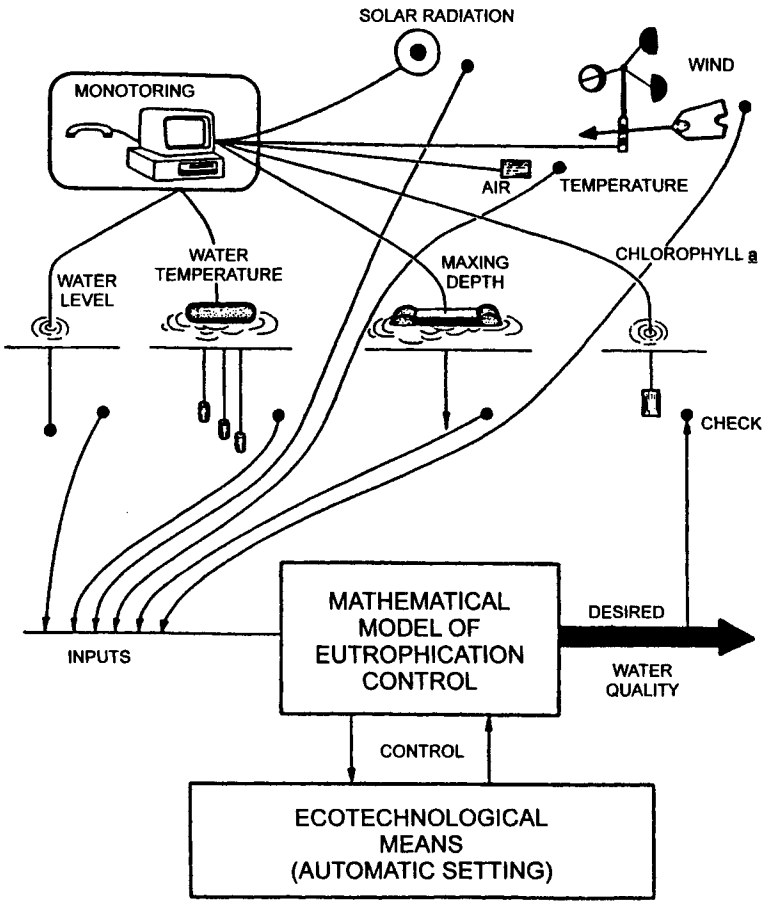



Fig. 5.7. The use of automatic monitoring, in connection with an online mathematical model to control eutrophication.

acceptable decisions, however, people need varying amounts of information that is often not easily obtainable. For complex systems such as water quality systems, it is difficult to preview the consequences of different options because there are many nonlinear relationships and complicated interactions among their components (Loucks and da Costa, 1991; Simonovic, 1996).

DSS are tools that provide managers with necessary information regarding the potential consequences of various management decisions. DSS use both the experiences of numerous experts, and the capabilities of the computer to rapidly calculate many complex relationships. The interactive function allows the user to assess the various versions of decisions under different possible situations.

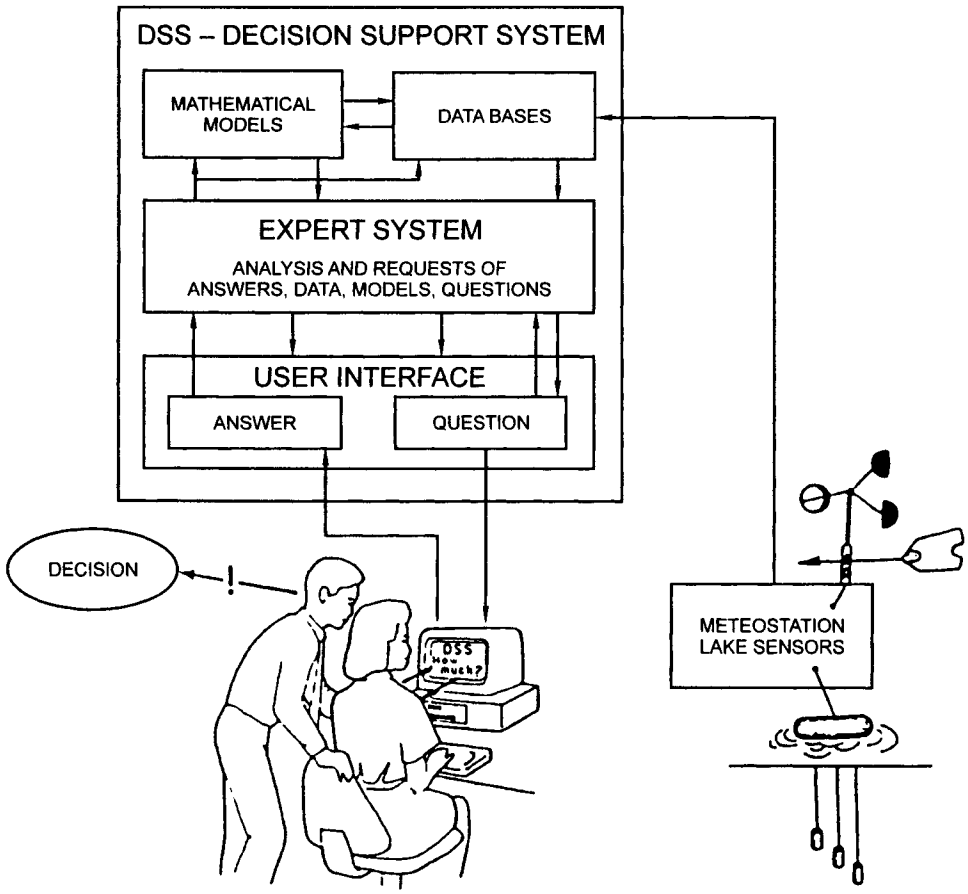



Fig. 5.8. A Decision Support System for water quality, combined with automatic recording.

Figure 5.8 provides an example of the structure and organization of a DSS. This DSS is connected to a meteorological monitoring station and to an in-lake monitoring system that measures some water quality variables in a lake or reservoir. Thus, actual information and past records of the situation are available for inspection in a transparent form. The database stores the past data, both recorded automatically or taken by hand sampling and analyzed biologically or chemically. It also contains a knowledge base of the expert system, which gives the questions what should be specified by the user in order to get the correct management advice. The latter comprise the decisions that experts would suggest in a particular situation. The DSS also incorporates a selection of water quality models that can be automatically set into operation, or suggested to the model user to be set into operation, since they will assist the manager in getting an idea of the consequences of specific management decisions before they are implemented.

Table 5.5. Expert and decision support systems for water quality management

| Model name | Purpose | Author |
|---|---|---|
| • No name | Selection of control strategy for lake eutrophication | Grobler et al., 1987 |
| • REH | Localization of pollution sources on a river that cause minimum deterioration of water quality | Gnauck et al., 1988 |
| • No name | DSS for environmental decisions, combining a modelling system with GIS | Fedra, 1990a, 1990b |
| • CMSS | Analysis of environmental catchment policies | Davis et al., 1991; Davis and Farley, 1997 |
| • AQUATOOL | Water resources management | Andreu et al., 1991 |
| • DELAQUA | Prototype expert system for lake water quality management | Recknagel et al., 1991 |
| • HEC-3 | Multipurpose quantitative operation of reservoir systems (no water quality) | Haestad Methods, 1993 |
| • GEO-WAMS | Modelling support system for integrating GIS with watershed analysis | DePinto and Rodgers, 1994 |
| • MASAS | Evaluation of micropollutants | Ulrich et al., 1991, 1995 |
| • WATERWARE | Water resource planning in the European Union, based on the No name DSS by Fedra given above, as well as on object-oriented simulation models for predicting the impact of water quantity and quality management strategies like IRAS (Loucks et al., 1995), RIBASIM and PCRSS | Fedra and Jamieson, 1996; Jamieson and Fedra, 1996a, 1996b |
| • BEKWAAM | Expert system for reservoir design and management; not directly water quality but using relevant information | Benoist et al., 1998 |
| • ODE | Software tool for rapid building of DSS which can handle multiple issues across different scales | Reed et al., 1999 |
| • RMU | Development of a catchment management DSS for Thailand | Scoccimarro et al., 1999 |

## 5.5 OVERVIEW OF MODELS APPLICABLE FOR VARIOUS MANAGEMENT PROBLEMS

### *5.5.1 Eutrophication Models*

The complexity of eutrophication problems was previously discussed in Section 2.2.2—*Eutrophication*, and the relations with nonpoint-source nutrient loads in Section 4.2. Eutrophication models represent a particularly wide spectrum of complexity. Table 5.6 reviews various eutrophication models representing the spectrum of complexity and the characteristics of the models (i.e., number of state variables, nutrients considered, number of lake segments or number of water layers, whether constant stoichiometric or independent nutrient cycles were applied, whether the model has been calibrated and validated,

Table 5.6. Range of eutrophication models

| Model name or developer | Number of state variables per layer or segment | Nutrients | Segments | Dimension (d) or layer (l) | CS or NC* | C and/or V† | Number of case studies |
|---|---|---|---|---|---|---|---|
| Vollenweider | 1 | P (N) | 1 | 1 l | CS | C + V | Many |
| Imboden | 2 | P | 1 | 2 l, 1 d | CS | C + V | 3 |
| O'Melia | 2 | P | 1 | 1 d | CS | C | 1 |
| GIRL | 3 | P | 1 | 1–2 d | CS | C + V | Many |
| Larsen | 3 | P | 1 | 1 l | CS | C | 1 |
| Lorenzen | 2 | P | 1 | 1 l | CS | C + V | 1 |
| Thomann 1 | 8 | P, N, C | 1 | 2 l | CS | C + V | 1 |
| Thomann 2 | 10 | P, N, C | 1 | 2 l | CS | C | 1 |
| Thomann 3 | 15 | P, N, C | 67 | 2 l | CS | — | 1 |
| Chen and Orlob | 15 | P, N, C | Sev. | 2 l | CS | C | 2 min. |
| Patten | 33 | P, N, C | 1 | 1 l | CS | C | 1 |
| Di Toro | 7 | P, N | 7 | 1 l | CS | C + V | 1 |
| Biermann | 14 | P, N, Si | 1 | 1 l | NC | C | 1 |
| Canale | 25 | P, N, Si | 1 | 2 l | CS | C | 1 |
| Jørgensen | 17 | P, N, C | 1 | 1–2 l | NC | C + V | 21 |
| CLEANER | 40 | P, N, C, Si | Sev. | Sev. l | CS | C | Many |
| Lavsoe | 7 | P, N | 3 | 1–2 l | NC | C + V | 25 |
| ASTER/MELODIA | 10 | P, N, Si | 1 | 2 l | CS | C + V | 1 |
| Baikal | > 16 | P, N | 10 | 3 l | CS | C + V | 1 |
| Chiemsee | > 14 | P, N, C, S | 1 | Profile | CS | C + V | Many |
| 3DWFGAS | > 70 | P, N, C, S | Many | Many | CS | C + V | 135 |
| Reward | 16 | P, N | 1 | Profile | CS | C + V | 5 |
| Mahamah | 8 | C, P, N | 1 | 1–2 l | CS | C + V | 2 |
| LAKE ECOSYSTEM | 11 | C, P, N | 1 | 1 | CS | C + V | 1 |
| MINLAKE | 9 | P, N | 1 | 1 | CS | C + V | > 10 |
| SALMO | 17 | P, N | 1 | 2 l | CS | C + V | 16 |

*CS means constant stoichiometric and NC means independent nutrient cycle.

†C means calibrated and V means validated.

number of case studies to which the model has been applied, etc.). It is assumed, particularly for more complex models, that some modifications from case to case to reflect specific lake conditions or properties should be reflected in the model (also see discussion in Section 5.4 on model generality).

The simple calculation modelling approach is exemplified by the relation between phosphorus, which can easily be calculated on the basis of the relation between the phosphorus load (corrected by the fraction trapped in the sediment) and chlorophyll (e.g., see OECD, 1982; Ryding and Rast, 1989; Rast and Thornton, 1996). Such empirical relations do have shortcomings. They are not necessarily general and, therefore, should be used with proper caution. The differences between the various empirical relationships indicates that a high standard deviation can be expected by application of empirical relationships.

As seen in Table 5.6, the complex dynamic models represent a range of complexity, given that the different models include a different number of submodels. Three possible core submodels that should be considered for inclusion in dynamic models are presented here. They will be discussed further when it is necessary to apply these submodels in an overall eutrophication model. The characteristics of each case study, as discussed above, should determine whether these three submodels should be included or not. They represent typical considerations for selecting the complexity for eutrophication models.

*The application of independent nutrient cycles* inevitably increases the complexity of a model, as this situation requires that nitrogen, phosphorus, carbon and perhaps silica may be included as state variables in each trophic level. In most models that include independent nutrient cycles, however, only one state variable is considered for zooplankton and fish. The application of independent nutrient cycles implies that the growth of phytoplankton is described as a two-step process, as follows:

- Phytoplankton nutrient uptake, in accordance with Monod's kinetics,
- Phytoplankton growth determined by the internal substrate concentration.

This complication obviously requires that the underlying data are of sufficient quality and quantity. Di Toro (1980) has shown that the application of independent nutrient cycles is particularly important when the model is used for shallow, very eutrophic lakes. In contrast, this complication can be omitted for deep, mesotrophic or oligotrophic lakes.

Because *lake bottom sediments* accumulate nutrients, it is important to describe quantitatively the processes determining the mass flows from sediment to water. This is of particular importance when the mass flows sediment to water are significant, compared with the other mass flows. As the relative amount of nutrients stored in the sediment is most significant for shallow, eutrophic lakes, the more detailed description presented below should always be included in models for such types of lakes.

The sediment–water submodel attempts to answer the following crucial question: To what extent will accumulated compounds in the sediment be redissolved into the lake water? Because these processes are important for lake eutrophication, the phosphorus and nitrogen exchange processes between mud and water have been extensively studied. Chen and Orlob (1975) ignored the exchange of nutrients between mud and water; as pointed out by Jørgensen et al. (1975), this will inevitably give a false prognosis. Ahlgren (1973) applied a constant flow of nutrients between sediment and water, while Dahl-Madsen and Strange-Nielsen (1974) used a simple first-order kinetic to describe the nutrient exchange rate.

A more comprehensive submodel (Fig. 5.9) for the exchange of phosphorus has been developed by Jørgensen et al. (1975). The settled material, $S$, is divided into $S_{\text{detritus}}$ and $S_{\text{net}}$, the first being mineralized by microbiological activity in the lake, and the latter being the material actually transported to the sediment. $S_{\text{net}}$ can also be divided into two flows, as follows:

$$S_{\text{net}} = S_{\text{net,s}} + S_{\text{net,e}}, \tag{5.1}$$

where $S_{\text{net,s}}$ = flow to the stable nonexchangeable sediment,
$S_{\text{net,e}}$ = mass flow to the exchangeable unstable sediment.

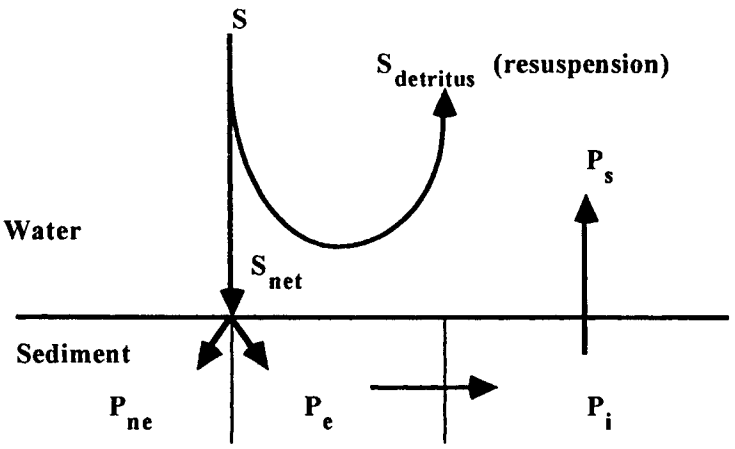



Fig. 5.9. Sedimentation, $S$, divided into $S_{\text{detritus}}$ and $S_{\text{net}}$; $P_{\text{ne}}$ nonexchangeable phosphorus in unstabilized sediment: $P_{\text{e}}$ exchangeable phosphorus in unstabilized sediment; $P_{\text{i}}$ phosphorus in interstitial water; $P_{\text{s}}$ dissolved phosphorus in water.

Correspondingly, the nonexchangeable, $P_{\text{ne}}$, and exchangeable, $P_{\text{e}}$, phosphorus concentrations, both based on the total dry matter in the sediment, also can be distinguished. Increases in the stabilized sediment can be found by numerous methods. The analysis of lead isotopes, for example, is a fast and reliable method.

The exchangeable phosphorus is similarly mineralized to detritus in a lake, and a first-order reaction gives a reasonably good description of the conversion of $P_{\text{e}}$ into interstitial phosphorus, $P_{\text{i}}$.

Finally, the interstitial phosphorus, $P_{\text{i}}$, will be transported by diffusion from the pore water in the sediment to the lake water. This process, which has been studied by Kamp-Nielsen (1975), can be described by means of the following empirical equation (valid at 7°C):

$$\text{Phosphorus release} = 1.21(P_{\text{i}} - P_{\text{s}}) - 1.7 \text{ (mg P m}^{-2} \text{ day)}, \tag{5.2}$$

where $P_{\text{s}}$ = the dissolved phosphorus in the lake water.

This submodel of water–sediment exchange was validated in three case studies (Jørgensen et al., 1975), based on examining sediment cores in the laboratory. Kamp-Nielsen (1975) has added an adsorption term to these equations.

A similar submodel for sediment nitrogen release has been developed by Jacobsen and Jørgensen (1975). The nitrogen release from sediment is expressed as a function of the nitrogen concentration in the sediment and the temperature, under both aerobic and anaerobic conditions.

Figure 5.10 shows a sediment profile from these examinations, illustrating the interpretation of the profile that can be used in the model to distinguish between exchangeable and nonexchangeable sediment phosphorus.

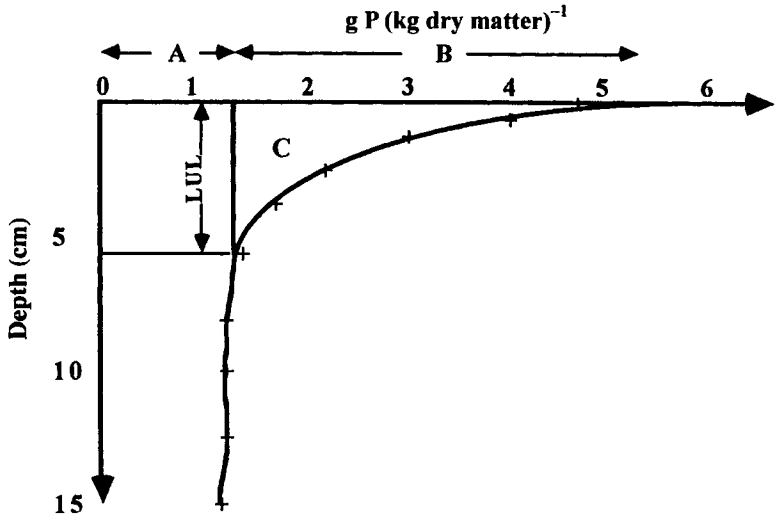



Fig. 5.10. Analysis of sediment core from Lake Esrom. Phosphorus concentration (mg P $g^{-1}$ dry matter) is plotted against the depth. The area $C$ represents exchangeable phosphorus. $f = (B \cdot A - 1)$, LUL is the unstabilized layer.

*Grazing of phytoplankton by zooplankton* ($Z$), and the predation of zooplankton by fish ($F$) are both expressed by a modified Monod expression, which considers a threshold concentration, $KT$, below which grazing or predation does not occur. The expression for phytoplankton grazing by zooplankton (Steele, 1976) is expressed as follows:

$$\mu Z = \mu Z_{\max} \cdot \frac{\text{Phyt} - KT}{\text{Phyt} + KM}, \tag{5.3}$$

where $KM$ = Michaelis–Menten constant.

A zooplankton carrying-capacity must often be introduced to give a better simulation of zooplankton and phytoplankton. Although carrying capacities are often observed in ecosystems, the need to introduce them in this case may be due to a too-simple representation of the grazing process. Phytoplankton might not be grazed, for example, by all the zooplankton species present, and some zooplankton species might use detritus as a food source. The zooplankton growth rate, $\mu Z$, is computed in accordance with these modifications as:

$$\mu Z = \mu Z_{\max} \cdot \text{FPH} \cdot \text{FT2} \cdot \text{F2CK}, \tag{5.4}$$

where FPH = the expression in equation (5.4),
FT2 = a temperature regulation expression,
F2CK = accounts for the carrying capacity.

Now,

$$\text{F2CK} = \frac{CK - \text{ZOO}}{CK}, \tag{5.5}$$

where $CK$ = carrying capacity.

If the data are not sufficient to include the three important submodels presented above, or other relevant submodels that an individual case study might require, it is recommended that the needed data be obtained with an intensive measuring effort that can provide high-quality data for the specific time period when the eutrophication processes are most dynamic.

An intensive measuring period, with several sets of measurements each week during the spring and summer bloom period, can first of all be applied to improve parameter estimation, which is often a focal problem in developing ecological models. The experience from conducting intensive measuring periods has identified the following advantages:

- Different optional expressions of simultaneously-limiting factors were tested, and only two gave an acceptable maximum growth rate for phytoplankton and an acceptably low standard deviation. These were (i) multiplication of the limiting factors, and (ii) averaging the limiting factors.
- The previously applied expression for the influence of temperature on phytoplankton growth gave unacceptable parameters, with standard deviations that were too high. A better expression (Equation 1) was introduced as a result of the intensive measuring period.
- It was possible to improve the parameter estimation, which gives more realistic values for some parameters. Whether this would give an improved validation when observations from a period with drastic changes in the nutrients loading are available could not be determined.
- The other expressions applied for process descriptions were confirmed.

It is important to validate models against independent set of measurements. Table 5.7 gives the results of a typical validation for a model with a medium to high complexity, developed on the basis of good data. $R$ and $A$ are the standard deviations for the mean and maximum value, respectively, of the considered state variables. $Y$, $R$ and $A$ give the errors in relative terms. Multiplying them by 100 gives the errors expressed as percentages (%). As shown in the table, the standard deviation, $Y$, for all measured state variables is 16%. It is the standard deviation for one comparison of modelled and measured values. As the standard deviation for a comparison of $n$ sets of modelled and measured values is $\hat{u}n$ times smaller, and $n$ is in the order of 200, the average picture of the lake is provided, with a very acceptable standard deviation of about 1%. $Y$ is generally 5–10 times larger, for example, for hydrological models (WMO, 1975).

The relative errors of the mean values, $R$, are 3% for production, 8% for phytoplankton and 2% for nitrogen, all of which are very acceptable values. The relative error for total

Table 5.7. Numerical validation of the described model

| Validation criterion | State variable | Value |
|---|---|---|
| *Y* | All | 0.16 |
| *R* | P total | 0.18 |
| *R* | P soluble | 0.16 |
| *R* | N total | 0.02 |
| *R* | N soluble | 0.14 |
| *R* | Phytoplankton | 0.08 |
| *R* | Zooplankton | 0.20 |
| *R* | Production | 0.03 |
| *A* | P total | 0.12 |
| *A* | P soluble | 0.15 |
| *A* | N total | 0.07 |
| *A* | N soluble | 0.03 |
| *A* | Phytoplankton | 0.15 |
| *A* | Zooplankton | 0.00 |
| *A* | Production | 0.08 |

phosphorus, however, is 18%, and that for zooplankton is 20%, both of which must be considered too large. The relative errors of the maximum values, *A*, range from 0% to 15%, an acceptable range. The ability of the model to predict maximum production and maximum phytoplankton concentrations has special interest within the context of a eutrophication model, with the relative errors being 8% and 15%, respectively—both of which are very acceptable.

A prognosis for the development of eutrophication in Lake Glumsø (the validation results referred to in Table 5.7 are based on this case study) by different removal efficiencies for phosphorus, nitrogen, or phosphorus and nitrogen simultaneously, have been made. The validation of this prognosis is presented here to illustrate the reliability of prognoses made on the basis of well-developed eutrophication models.

It was previously stated that nitrogen removal had little or no effect on the lake, while phosphorus removal would give substantial reductions in the phytoplankton concentration. The results of two cases are summarized in Table 5.8, as follows:

Case A. The treated wastewater has a concentration of 0.4 mg P $l^{-1}$, corresponding to about 92% removal efficiency, which should be achieved with proper chemical precipitation.

Case B. The treated wastewater has a concentration of 0.1 mg P $l^{-1}$, corresponding to about 98% removal efficiency, which will require chemical precipitation, for example, in combination with ion exchange.

As seen in Table 5.8, the water quality will improve significantly, in accordance with the predictions. Case B, with a 98% removal of phosphorus, is obviously preferred. In the third year, Case B will give a reduction in production from 1100 g C $m^{-2}$ yr to 500 g C $m^{-2}$ yr, with the water transparency increasing from a minimum value of 20–60 cm. The 9th year would even result in reduction of the production to 320 g C $m^{-2}$ yr, corresponding to a

Table 5.8. Model predictions for two concentrations of treated wastewater: Case A—0.4 mg P $l^{-1}$; Case B—0.1 mg P $l^{-1}$

| | Third year | | Ninth year | |
|---|---|---|---|---|
| | Case A | Case B | Case A | Case B |
| g C $m^{-2}$ yr | 650 | 500* | 500 | 320* |
| Minimum Transparency (cm) | 50 | 60 | 60 | 75 |

*An error of 3% on this value can be expected if the validation results hold, see *R* for production in Table 5.7.

mesotrophic lake, which is an acceptable improvement for a shallow lake situated in an agricultural area.

The prognosis gives a pronounced effect of 98% phosphorus removal, which could therefore be recommended to the appropriate environmental authorities. Further improvements after nine years should not be expected with this case study.

Conveyance of the wastewater also was considered, but has the following disadvantages:

- It is slightly more expensive than the Case B solution, taking interests, depreciation and running costs into consideration.
- The phosphorus is *not* removed but only transported to the downstream Susaa River, where its effects have not been considered.
- The sludge produced at the biological treatment plant will be less valuable as a soil conditioner, since the phosphorus concentration will be lower than when phosphorus removal is included.
- The freshwater is not retained in the lake, from which it could have been reclaimed, if needed, after storage for some time. Freshwater supply is not presently a problem in this area, but it is foreseen that it might be in 20–40 years.

In spite of these observations, the community in this case study chose to convey its wastewater to the Susaa River, due to a preference for traditional methods. The pipeline was constructed in 1980 and began operation in April 1981, which has enabled a validation of the presented prognosis.

Lake Glumsø was ideal for these studies due to its limited depth and size, but also because a reduced nutrient input to the lake could be foreseen. The limited retention time (about six months) makes it realistic to obtain a validation of a prognosis within a relatively short time interval (a few years). On April 1, 1981, the direct input of wastewater to the lake was stopped. Because the capacity of the sewage system is still too small, however, a minor input of mixed rain water and wastewater is discharged into the lake from time to time through an upstream tributary. Thus, the phosphorus loading is not reduced by 98%, but rather only by 88% (determined by a phosphorus balance). The prognosis in Case A, therefore, can be used for comparison.

During the third year after the reduction in phosphorus load occurred, a pronounced effect on the lake was observed. Table 5.9 compares some of the most important data of the prognosis. It also includes data obtained during the first two months of the third year. The

Table 5.9. Comparison of predicted and measured data

| | Predicted (Case A, 92% P reduction) | Measurements (approximately 88% reduction) |
|---|---|---|
| *Minimum transparency* | | |
| First year | 20 cm | 20 cm |
| Second year | 30– | 25– |
| Third year | 45– | 50– |
| *Production*, g C $m^{-2}$ day, maximum | | |
| First year | $9.5 \pm 0.8$ | $5.5 \pm 0.5$ |
| Second year | | |
| (spring) | $6.0 \pm 0.5$ | $11 \pm 1.1$ |
| (summer) | $4.5 \pm 0.4$ | $3.5 \pm 0.4$ |
| (autumn) | $2.0 \pm 0.2$ | $1.5 \pm 0.2$ |
| Third year | | |
| (spring) | $5.0 \pm 0.4$ | $6.2 \pm 0.6$ |
| *Chlorophyll in spring*, mg $m^{-3}$, maximum | | |
| First year | $750 \pm 112$ | $800 \pm 80$ |
| Second year | $520 \pm 78$ | $550 \pm 55$ |
| Third year | $320 \pm 48$ | $380 \pm 38$ |

table identifies errors as ± for g C $m^{-2}$day and chlorophyll maximum (mg $m^{-3}$). For the predicted values, the results from Table 5.7 (8% for production and 15% for phytoplankton concentration) are used to determine standard deviations. For the measured values, an error of 10% was estimated.

The lake was previously dominated by *Scenedesmus*. After the conveyance of the wastewater, it was dominated by diatoms, which have a lower optimum temperature and, therefore, typically bloom earlier in the spring than *Scenedesmus*. This seems to explain the discrepancy between the predicted and measured values regarding this parameter. The model, therefore, might improve its predictions if it was possible to account for shifts in species composition. Results published by Jørgensen (1981), Jørgensen and Mejer (1981a, 1981b) and Jørgensen (1992a, 1992b) indicate this would be possible by introducing a maximum growth rate for phytoplankton (see Section 5.6.7). Such models are called "structurally dynamic models" (Jørgensen, 1986a). They have given such promising results in recent times that Section 5.5.9 is devoted to discussion of this type of model.

In the Lake Glumsø case, however, since diatoms take up silica, it would probably also be necessary to introduce a silica cycle into the model. The other production and chlorophyll values are well predicted, except the spring production in the second year (Table 5.9). The predictions of minimum transparency are acceptable, as they show a difference of 5 cm or less. The general trends in the nutrient concentrations exhibit a good correlation between predicted and measured values.

This case study shows that it is possible to make reliable predictions using eutrophication models, provided that data of sufficient quality and quantities are available. On the other hand, an attempt should always be made to develop a eutrophication model, even if

the available data are not sufficient to construct a complex model, because the model results can be used to compare different management strategies with an indication of the uncertainty resulting from the validation. If the uncertainties are not acceptable, the model can be used to assess the weak points and help identify modifications that should be introduced to improve the model results.

Thus, the model should be considered a tool that is able to synthesize the knowledge available on a lake ecosystem of concern and, by following the recommendations in Section 5.4, should be able to obtain the best possible model predictions under given circumstances.

### 5.5.2 *Toxic Substance Models*

Over approximately the last 20 years, models of toxic substances have emerged because of the increasing interest in the environmental management of toxic substances that can cause water pollution. The topic of ecotoxicology was discussed in Section 3.2.6.

Toxic substance models attempt to model the fate and/or effects of toxic substances in ecosystems. Toxic substance models are most often biogeochemical models because they are meant to describe the mass flows of the toxic substances being considered. There also are models of population dynamics, which include the influence of toxic substances on the birth rate, growth rate and/or mortality and, therefore, which should also be considered toxic substance models.

Toxic substance models have some characteristic properties:

- The need for parameters to cover all possible toxic substance models is great. Thus, general estimation methods are widely used. Methods based on the chemical structure of chemical compounds are developed; the so-called QSAR and SAR methods.
- The safety margin should be high, for example, when expressed as the ratio between the actual concentration and the concentration that causes undesired impacts.
- The possible inclusion of an effect component, which relates the output concentration to its effect. It is easy to include an effect component in the model. However, it often is a problem to find a well-examined relationship upon which to base it. Rather, it will require the use of good knowledge of the properties, particularly the toxicological properties of the component of concern.
- The possibility and need of simple models, based on points 1 and 2 above, and of our limited knowledge of the processes, parameters, sublethal effects, antagonistic and synergistic effects.

The decision regarding which model class to apply is based on the ecotoxicological problem that one wishes to solve with the model. Figure 5.11 shows a model of lead contamination, with bioaccumulation through the food chain. This model requires detailed data about the lead concentrations in the various biological components as a function of time, which are rarely available.

However, it often is only the toxic substance concentration in one trophic level that is of concern. This includes the zero trophic level, which is understood to be the medium—lake

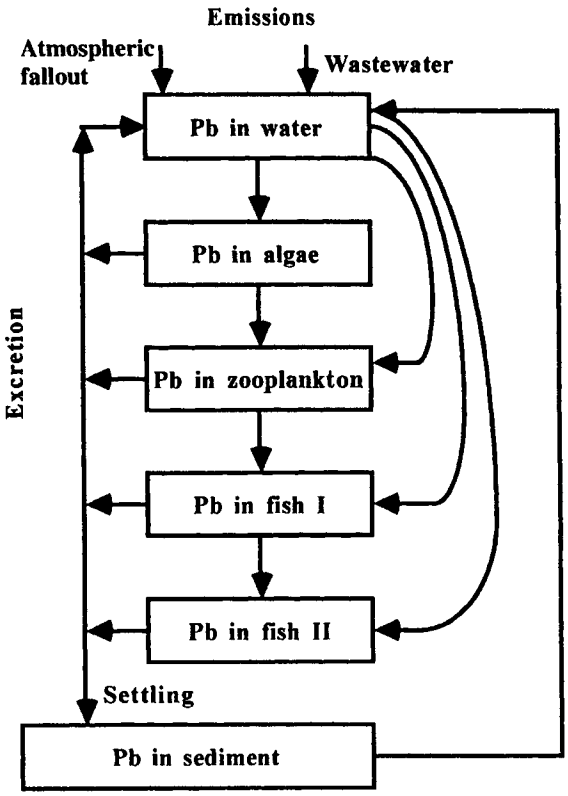



Fig. 5.11. Conceptual diagram of the bioaccumulation of lead (Pb) through a food chain in an aquatic ecosystem.

water, as illustrated with the example in Figure 5.12. It is a model of copper contamination in a lake ecosystem. The main concern is the copper concentration in the water, since it might reach a toxic level for the phytoplankton. Zooplankton and fish are much less sensitive to copper contamination, so the initial concern focuses on the concentration level harmful to phytoplankton. However, only the ionic form of copper is toxic and, therefore, it is necessary to model the partitioning of copper in its ionic form, complex-bound form, and adsorbed form. The exchange between copper in the water phase and the sediment also is included because the sediment is able to accumulate relatively large amounts of heavy metals. The amount released from the sediment may be significant under certain circumstances (e.g., low pH).

The acknowledgment of the uncertainty is of great importance for all models, but it is particularly important for ecotoxicological models. It may be taken into consideration either qualitatively or quantitatively. Another problem is: Where does one take the uncertainty into account? Should the economy or the environment benefit from the uncertainty?

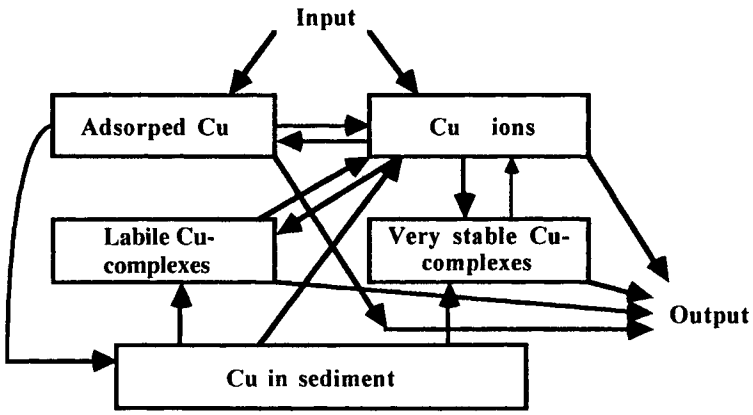



Fig. 5.12. Conceptual diagram of a simple copper (Cu) model.

Unfortunately, most decision-makers up to now have used the uncertainty to the benefit of the economy. This is a completely unacceptable approach, since the same decision-makers would, for example, never consider whether uncertainty should be used for the benefit of the economy or the strength of a bridge in a civil engineering project. Table 5.10 gives an overview of a number of ecotoxicological models applied to lake and reservoir management.

Ecotoxicological models are widely used to perform environmental risk assessments, *ERA* (see Section 5.5.8). This is of particular interest for lakes and reservoirs used as drinking water supplies, or for which the fisheries are significant. *ERA* typically answers such questions as: What is the risk of contaminating drinking water or fish for a given application of pesticides in an agricultural project adjacent to a lake or reservoir?

### *5.5.3 Acidification Models*

Lake acidification is caused by acid rain, which originates from emissions of sulfur and nitrogen oxides into the atmosphere (see Section 2.2.2—*Acidification*). Thus, assessment of acidification for management purposes requires a chain of models, linking the emissions to the atmosphere to the effects of acid rain on soil chemistry in the lake's catchment area, and further to the pH changes in a lake. This is illustrated in Figure 5.13, which illustrates the model linkages between energy policy and lake and reservoir acidification.

In this context, only the last submodels of the chain (i.e., lake models and soil models) will be mentioned. These latter models determine the composition of the drainage water flowing into lakes and reservoirs. Those interested in entire model chains are referred to Alcamo et al. (1990) and Jørgensen (1994).

Several models have been developed to translate the emissions of sulfur and nitrogen compounds into changes in soil chemistry (in the first hand to changes in the pH of soil

Table 5.10. Examples of ecotoxicological lake models

| Toxic subst. | Model characteristics | Reference |
|---|---|---|
| • Cadmium, mercury | Food chain as in a eutrophication model 6 state variables: water, sediment, suspended matter, invertebrates, plant and fish | Thomann et al., 1974<br>Miller, 1979 |
| • Vinyl chloride | Chemical processes in water | Gillet, 1974 |
| • Methyl parathion | Chemical processes in water, microbial degradation, adsorption, 2–4 trophic levels | Lassiter |
| • Heavy metals | Concentration factor, excretion, bioaccumulation | Ayoma et al., 1978 |
| • Pesticides in fish: DDT and methoxychlor | Ingestion, concentration factor, adsorption on body, defecation, excretion, chemical decomposition, natural mortality | Leung, 1978 |
| • Lead | Hydrodynamics, precipitation, toxicity of ionic lead on algae, invertebrates and fish | Lam and Simons, 1976 |
| • Radionuclides | Hydrodynamics, decay, uptake and release by various aquatic surfaces | Gromiec and Gloyna, 1973 |
| • Polycyclic aromatics | Transport, degradation, bioaccumulation hydrocarbons | Bartell et al., 1984 |
| • Cadmium, PCB | Hydraulic overflow rate (settling), sediment interactions, steady state food chain submodel | Thomann, 1984 |
| • Hydrophobic organic | Gas exchange, sorption/desorption, hydrolysis compounds, photolysis, hydrodynamics | Schwarzenbach and Imboden, 1984 |
| • Mirex | Water–sediment exchange processes, adsorption, volatilization, bioaccumulation | Halfon, 1984 |
| • Toxins (aromatic hydrocarbons, Cd) | Hydrodynamics, deposition, resuspension, volatilization, photooxidation, decomposition, adsorption, complex formation (humic acid) | Harris et al., 1984 |
| • Persistent organic chemicals | Fate, exposure and human uptake | Paterson and Mackay, 1989 |
| • pH, calcium and aluminium | Survival of fish populations | Breck et al., 1988 |
| • Pesticides and surfactants | Fate in rice fields | Jørgensen et al. |
| • Toxicants | Migration of dissolved toxicants | Monte, 1998 |
| • Growth promoters | Fate, agriculture | Jørgensen |
| • Toxicity | Effect on eutrophication | Legovic, 1997 |
| • Pesticides | Mineralization | Fomsgaard et al., 1997 |
| • Mecoprop (3) | Mineralization in soil | Fomsgaard et al., 1999 |

water). Kauppi et al. (1984) used knowledge about buffer-capacity and velocity to relate the emissions to soil water pH. A critical pH value of 4.2 is usually applied to interpret the results.

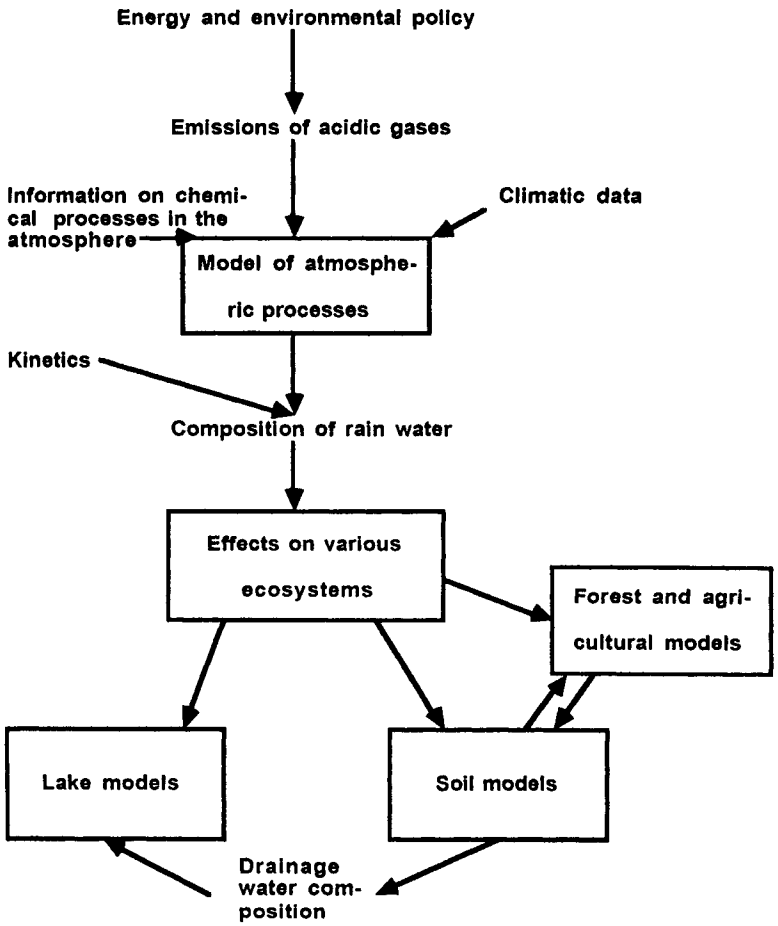



Fig. 5.13. A series of models must be used to relate energy and environmental protection policy to effects on soil, plants, water chemistry and aquatic ecosystems, which serve as feedback to political issues.

From the results of the atmospheric deposition calculations, it is possible to estimate the acid load in equivalents of hydrogen ions per square meter per year. In the European model, RAINS, the acid load is computed from the deposition, after accounting for forest filtering and atmospheric deposition of cations (for details, see Alcamo et al., 1990).

The Soil Map of the World classifies European soils into 80 soil types. The fraction of each soil type within each grid element is computerized to an accuracy of 5%. The resolution of the European RAINS model is such that each grid element includes one to seven soil types, with a mean number of 2.2. The model goal is to keep track of the development of soil pH and buffer capacity. Further development of soil models is still needed, as the soil model must be considered the weakest model in the chain.

It can be concluded that models of soil processes and chemical composition, including the concentrations of various ions and pH of the drainage water, are complex. It is not the intention here to present the models in detail—they are far too complex to do so—but rather to mention the difficulties and a few of the basic ideas behind them. Further details can be found in the references given in this section.

Henriksen (1980) and Henriksen and Seip (1980) developed an empirical model for the relationship between lake water pH and sulfur load. They plotted related values of pH and sulfur loads for a given catchment area with specific sensitivities for acidification. The resulting curves look like a titration curve. The sensitivity is dependent on the calcium concentration (low calcium concentration, high sensitivity due to low buffer capacity). For a sensitive area, a sulfur load of less than 0.7, or even 0.5 g $m^{-2}$ · year is needed to insure a pH of 5.3 or higher.

These simple empirical approaches are used without consideration of the drainage water. Weathering processes, however, can be incorporated in the empirical approaches, although the results have little generality since they are based on regression analysis of local measurements. The presented modelling approach has been used on Scandinavian lakes with good results.

Most biological models of lake acidification focus on fish. On the basis of data from 719 lakes, Brown and Salder (1981) developed an empirical model relating the fish population to the pH of the water. For their study lakes in southern Norway, they found that a 50% reduction of the sulfate emissions will give an average increase in pH of 0.2, which will only improve the fish populations in 9% of the lakes. However, a criticism of the model is that it underestimates the relationship between the reduction of sulfate emissions and pH.

Muniz and Seip (1982) developed another empirical model, in which they distinguished between lakes of different conductivity. Chen et al. (1982) developed a very comprehensive pH-effect model, which considers the effects on all levels in the food chain and the total effects on the ecosystem.

### *5.5.4 Wetland Models*

Nitrogen and phosphorus balances have shown that agricultural runoff and other nonpoint sources contribute significantly to overall lake and reservoir pollution, and particularly to the eutrophication problem. The results imply that environmental technology alone is not sufficient to cope with nonpoint pollutant sources, but must be supplemented with other methods. The results of comparative studies have shown that the use of wetlands is often a very effective method for reducing pollution (see Section 4.2.4).

Mitsch (1976, 1983) provided a more comprehensive review of wetland models than can be presented here. He distinguished between energy/nutrient models, hydrological models, models of spatial ecosystem, models of tree growth, process models, causal models and regional energy models. Mitsch et al. (1988) also reviewed several types of wetland models.

A nitrogen balance for agricultural regions revealed that nitrogen from nonpoint sources plays a major role in lake pollution, and that a solution to the eutrophication problem of freshwater and marine ecosystems cannot be realized without solving the problems asso-

ciated with nonpoint sources of pollution. The entire spectrum of ecological engineering methods identified in Chapter 4 has been implemented in various cases to solve the problems. As indicated above, the methods may have different effects in different situations, dependent on the actual mass balance and general properties of the lake or reservoir in question. In this context, there obviously is a need for a model that can be used to make predictions on the nutrient removal capacity of a wetland on the basis of certain information about an existing or a planned wetland. The goal is to develop as general a model as possible. However, because ecological models only have a certain generality, it is necessary to distinguish between the general relations and the more site-specific parameters and forcing functions. Thus, it is not possible to achieve a complete generality for wetland models.

Jørgensen (1994) presents a relatively general wetland model able to predict nitrogen removal by wetlands. A wide spectrum of wetland models has been proposed lately (e.g., see Jørgensen et al., 1995). The various models that only focus on nitrogen removal can easily be expanded to also cover heavy metal and phosphorus removal by adsorption. A simple adsorption isotherm can be added to consider the removal of phosphorus and nitrogen by harvest at day $m$ of the wetland. The climatic information and some site specific properties of the focal wetland(s) (e.g., content of organic matter, plant species, hydraulic conductivity of the soil) must be assessed before a calibration, a validation and a prediction with use of the model can occur.

### 5.5.5 *Fisheries Models*

Augmented exploitation of freshwater fish resources by sport and commercial fishery (see Section 2.1.4), and the deterioration of water quality, has stimulated concern about the depletion of fish stocks.

This reality has intensified the development models which take into account the effects of harvesting and water quality on the fish population. Models with a wide spectrum of complexity attempt to provide a management tool to assess an optimum fishery strategy.

For lakes with important commercial fisheries, landing records are often available for periods of multiple decades, and may open the possibilities of developing a statistical fisheries model. However, a statistical model generally does not take into account a number of important factors, such as interactions among species, water quality and changes in the concentrations of fish food. A statistical model would build on the assumption that present and past properties of the environment and the fish populations will be maintained. Statistical models will not be discussed further here, primarily because they do not consider the influences of water quality. It should be mentioned, however, that the International Lake Environment Committee (ILEC) has developed a software that contains among several models, a simple eutrophication model with four state variables. It also computes the fish population with a simple regression equation relating primary production and the fish population.

The simplest approaches assume that the entire fish population is homogeneous, and does not consider the population dynamics and related age structure, which is essential for

fishing policy. The more complex approaches consider the influences of water quality on the population dynamics and the age structure of the fish populations.

### 5.5.6 *Models of Biomanipulation*

Success with biomanipulation varies considerably from case to case, depending on the specific circumstances (see Section 4.3.1). Thus, modelling seems an obvious tool to use in selecting biomanipulation as a control measure, since the primary task of management models is always to predict the results of alterations in an ecosystem. However, biomanipulation implies that the structure of the ecosystem (i.e., a lake) is changed, which is much more difficult to simulate with a model than is a simple change in forcing functions. On the other hand, structurally dynamic models are emerging (e.g., see Jørgensen, 1992a, 1992b). Models are increasingly used as an experimental tool that can sometimes also be used to explain ecosystem behavior associated, for example, with chaos and catastrophe theories (Jørgensen, 1992b).

Models based on case studies with structural changes are still rare. However, we know that ecosystems have the ability to adapt to changed forcing functions, and to shift to species better suited to emerging conditions. Because radical changes are imposed on ecosystems, which inevitably will lead to the most pronounced changes in ecosystem structures, it is particularly interesting to apply models with dynamic structures, which are becoming increasingly important in environmental management. Thus, this type of model is discussed in this section in relation to lake and reservoir biomanipulation.

Biomanipulation is based on changes in ecosystem structure imposed by intentional changes in fish populations. In this context, it is of interest to explain when and why top-down control measures are working (i.e., when the imposed structural changes can be predicted to work). This is also possible with the use of catastrophe theory on lake eutrophication models.

The short-term results of biomanipulation have been encouraging. It is unclear, however, whether or not a manipulated ecosystem will ultimately return to the initial eutrophic and turbid conditions. Some observations suggest that, if low nutrient concentrations are combined with a relatively high concentration of predatory fish, a stable steady state will be attained, while high nutrient concentration and high predatory concentration will lead to an unstable clear water state (see Hosper, 1989; Van Donk et al., 1989). On the other hand, turbid conditions may prevail even at medium nutrient concentrations, provided that the predatory fish concentration is low. By introducing more predatory fish, however, the conditions may improve significantly, even at medium nutrient concentrations.

Willemsen (1980) distinguishes in the temperate region into two possible conditions:

- A "bream state" characterized by turbid water, and a high degree of eutrophication relative to the nutrient concentration. Submerged vegetation is largely absent from such systems. Large amounts of bream are found, while pike are rarely found.
- A "pike state" characterized by clear water, and a low degree eutrophication relative to the nutrient level. Pike are abundant, while significantly fewer breams are found, compared with the "bream state". Willemsen's work shows that the pike/bream ratio

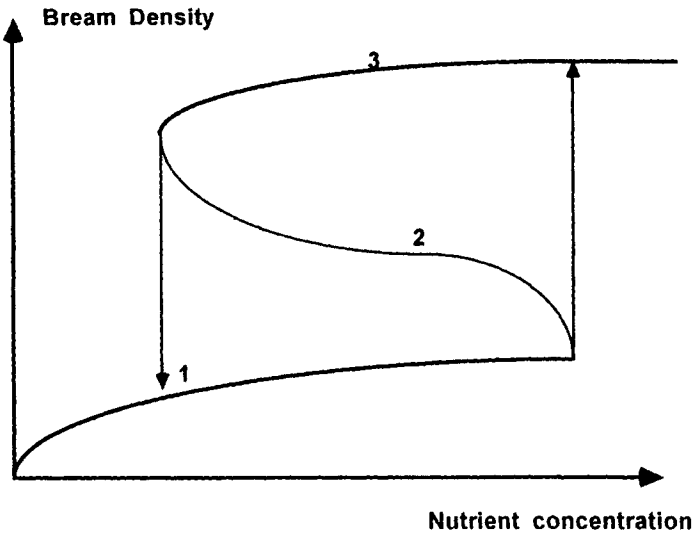



Fig. 5.14. A catastrophe fold shown by means of a projection of the intersection line of isocline planes of bream and pike on the nutrient–bream plane.

is strongly correlated with water transparency and that the separation between the two states is relatively distinct.

This behavior is clearly analogous to other examples described by catastrophe theory applied to biological systems (Jørgensen, 1992a, 1994). Figure 5.14 shows the catastrophe fold, where a bream isocline is plotted in relation to the nutrient concentration, assuming that the pike population is in a steady state. The isocline consists of the stable parts, 1 and 3. The unstable part, 2, corresponds to the "jump" between the two stable points.

The discontinuous response to increasing and decreasing nutrient levels implies that decreased nutrient levels will not cause a significant decrease in lake or reservoir eutrophication, and a significant increase in water transparency, before a rather low level has been attained. It may be possible, however, to "push" the equilibrium from point 3 to point 1 with the addition of predatory fish. This modelling example illustrates that two different concentrations of planktivorous fish can coexist with a certain nutrient concentration, which explains the hysteresis reaction shown in Figure 5.14. The general modelling experience is that a given set of forcing functions will give a certain set of state variables, including the concentration of planktivorous fish. However, when the set of equations that describe the ecosystem has a formulation causing catastrophic behavior in the mathematical sense, the reactions described above are observed.

To further explain these observations, models have been used as an experimental tool in the sense that the model description is in accordance with well-working lake models, but for which simulations with forcing functions for which there are no data available to control the model output, are carried out. The model used for these experiments is shown in Figure 5.15. Only phosphorus is considered as a nutrient in this case, although it is

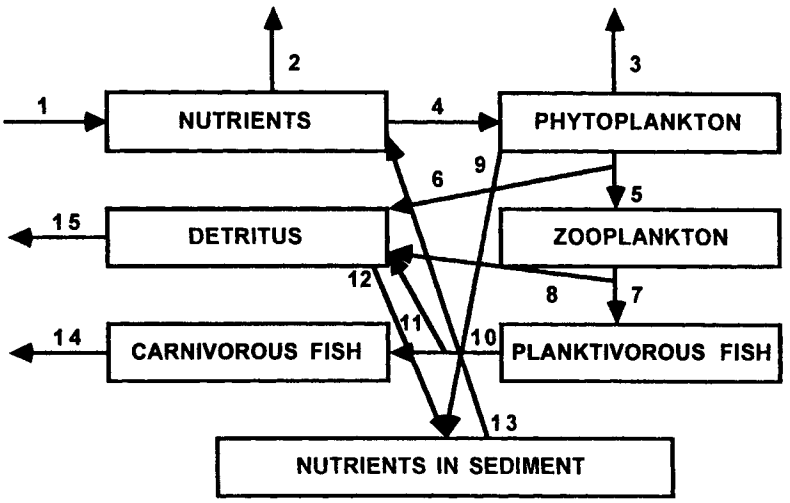



Fig. 5.15. The model used as an experimental tool. The state variables are nutrients (only phosphorus is considered in this case), phytoplankton, zooplankton, planktivorous fish, carnivorous fish, nutrients in sediment and detritus. The processes are: 1—inflow of nutrients, 2—outflow of nutrients, 3—outflow of phytoplankton, 4—uptake of nutrients by phytoplankton for growth, 5—grazing, 6—loss of detritus by grazing, 7—predation on zooplankton, 8—loss of detritus by this predation, 9—settling of phytoplankton, 10—predation by carnivorous fish, 11—loss of detritus by this predation, 13—release of nutrients from the sediment, 14—catch of fish, 15—outflow of detritus.

feasible to consider both nitrogen and phosphorus. The model encompasses the entire food chain. The modelling problem in the case of different inputs of nutrients, however, is that the phytoplankton, zooplankton, planktivorous and carnivorous fish are all able to adjust their growth rates within certain ranges. Thus, it is necessary to test a series of simulations with different combinations of growth rates to find which combination can give the highest probability of survival for all four classes of species.

The results of these simulations can be summarized in the following points:

- Between a total phosphorus (TP) concentration of 60–125 $\mu g\ l^{-1}$, two levels of planktivorous fish give a stable situation, with a high probability of survival for all four classes of species. The level with the lowest level of planktivorous fish give the highest concentration of zooplankton and carnivorous fish, while the phytoplankton concentration is lowest. This interval from 60–125 $\mu g\ l^{-1}$ is, of course, dependent on the model descriptions of the lake and must not be taken as fixed values for all lakes. Rather, they only indicate that there is an interval corresponding approximately to mesotrophic conditions and two stable situations.
- The stable situation with the lowest level of planktivorous fish corresponds to the lowest growth rate of zooplankton and phytoplankton, which normally implies species that are larger in size (see Peters, 1983).

- Only one stable level is achieved at total phosphorus concentrations below 60 $\mu g\ l^{-1}$ or above 125 $\mu g\ l^{-1}$. The zooplankton, relative to the level of phytoplankton, are present in highest levels at total phosphorus concentrations below 60 $\mu g\ l^{-1}$ and at relatively low levels above 125 $\mu g\ l^{-1}$.
- When the planktivorous fish are present at low concentrations either below total phosphorus concentrations of 60 $\mu g\ l^{-1}$, or between 60–125 $\mu g\ l^{-1}$, the phytoplankton is controlled by the relatively high zooplankton level (i.e., the grazing pressure), while phytoplankton at a high planktivorous level above total phosphorus concentrations greater than 60 $\mu g\ l^{-1}$ is controlled by the nutrient concentration.

As already underlined, this exercise should only be considered qualitatively. However, the results suggest that the use of biomanipulation only seems to be successful at an intermediate nutrient level, which is consistent with the results of many biomanipulation experiments (see Section 4.3.1). The range at which biomanipulation can be used successfully is most probably dependent on the specific conditions in the considered lakes, which unfortunately makes it problematic to use results described in this section more qualitatively. The model example here uses a test of many combinations of growth rates to find the best combination of parameters from a survival perspective. This adjustment of the parameters corresponds to adaptation within some ranges, and for more radical changes of the parameters to a change in the species composition in the lake. If models are to be used as management tool in cases where significant changes in nutrient levels will occur, or where biomanipulation is being considered, it is necessary to develop models that can account for structurally dynamic changes. However, this type of model development is still in its infancy, and only limited experience is available.

Our present models have generally rigid structures and a fixed set of parameters, indicating that no changes or replacements of the components are possible. However, it is necessary to introduce parameters (properties) that can change according to changing general conditions for the state variables (components). The current idea is to test if a change of the most crucial parameters produces a higher so-called goal function of the system and, if so, to use that set of parameters.

The type of models that can account for changes in species composition, as well as for the ability of the species (i.e., the biological components in the models) to change their properties (i.e., to adapt to the prevailing conditions imposed on the species), are as indicated in the introduction to this section; namely, the so-called *Structurally Dynamic Models* (see Fig. 5.16). They also may be called the next or fifth generation of ecological models, in order to emphasize that they are radically different from previous modelling approaches and can do more; namely describing changes in species composition.

It can be argued that the ability of ecosystems to replace present species with better-fitted species, can be considered by constructing models that encompass all the actual species for the entire period the model attempts to cover. However, this approach has two essential disadvantages. First, the model becomes very complex, since it will contain many state variables for each trophic level. It implies the model will contain many more parameters that have to be calibrated and validated, and which will introduce a high uncertainty to the model predictions, rendering the application of the model very case-

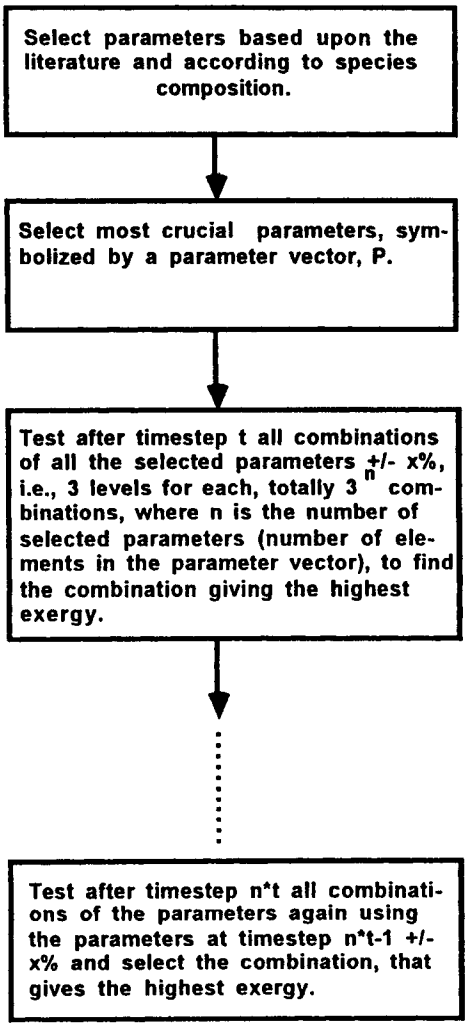



Fig. 5.16. The procedure used to develop structurally dynamic models.

specific (Nielsen, 1992a, 1992b). In addition, the model will still be rigid and not give the model the ecosystem property of having continuously changing parameters even without changing the species composition (Fontaine, 1981).

*Structurally Dynamic Models* can be developed by a current change of some of the most crucial parameters, reflecting the change of species. These changes may be based on expert knowledge, or may be determined with the use of a goal function. In the first case, the changes of parameters are made in accordance with good knowledge of the dominant species under different conditions (i.e., different combinations of forcing functions).

Reynolds (1996) illustrates this approach. In the latter case, the parameters currently are changed according to goal functions. Several goal functions have been proposed, but only a few models accounting for changes in species composition, or for the ability of species to change their properties within some limits, have been developed. Straškraba (1979) used a maximization of biomass as the governing principle (i.e., the above-mentioned goal function). The model computes the biomass and adjusts one or more selected parameters to achieve the maximum biomass at every instance. The model has a computation program that computes the biomass for all possible combinations of parameters within a given realistic range. The combination that gives the maximum biomass is selected for the next time step, etc. However, biomass can hardly be used in models with more trophic levels, and it will lead to biased results by, for example, adding together the biomass of fish and phytoplankton. The thermodynamic variable, exergy, has been used most widely as a goal function in ecological models; see Jørgensen (2002) and Section 5.5.9.

The modelling experience shows that models can help explain why biomanipulation works under some circumstances, and why it does not work under other circumstances. The applications of catastrophe theory and/or structurally dynamic models are able to explain the appearance of hysteresis in the relations between nutrient level and eutrophication (see Fig. 5.17). The results can be used to help explain that this hysteresis relationship exists in an intermediate nutrient level, which explains why biomanipulation in this range of nutrient concentrations has worked properly (i.e., with long-term effects), but not above or below this intermediate range.

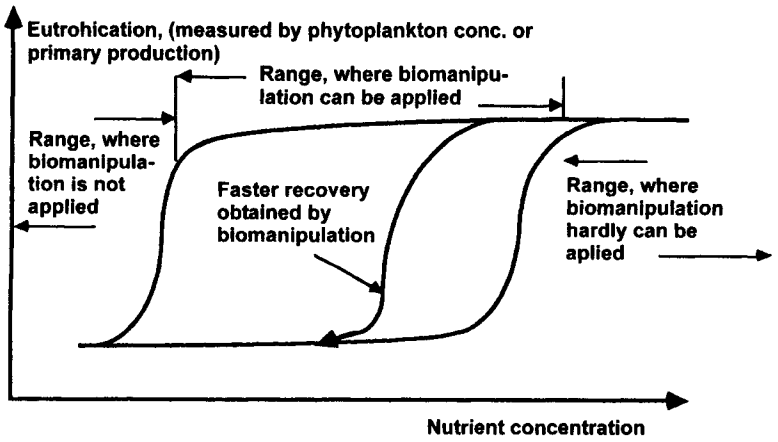



Fig. 5.17. The hysteresis relation between nutrient level and eutrophication measured by the phytoplankton concentration or the primary production. The possible effects of biomanipulation are shown. As shown, will it be able to short cut between the high eutrophication structure and the low eutrophication structure? An effect of biomanipulation can hardly be expected above a certain concentration of nutrients, as indicated on the diagram. Below a certain nutrient concentration, biomanipulation will not be applied, since eutrophication is too low to expect any effect.

Models have been an applicable tool in efforts to understand biomanipulation results. The results may be summarized in the following recommendations for using models for managing biomanipulation:

- Attempt either to use a structurally dynamic model or catastrophe theory to examine the possibilities of using biomanipulation.
- Consider the results only as approximations, unless very good data for applying structural dynamic models are available.
- Consider intermediate nutrient levels, between approximately 60 $\mu g\ l^{-1}$ and 125–140 $\mu g\ l^{-1}$ of total phosphorus, or 7 times higher concentrations for nitrogen if it is the limiting nutrient, as the range within which biomanipulation will most probably work.

The results obtained by modelling and in-lake biomanipulation experiments may be explained by the presence of ecological buffer capacities and time lags in structural changes. When the nutrient level is increased (see Fig. 5.17), the degree of eutrophication will initially not increase significantly. The lake ecosystem has an ecological buffer capacity, which can be explained by increased grazing and settling of the phytoplankton resulting from increased nutrient levels. At a certain nutrient level, however the zooplankton can no longer control the phytoplankton concentration, which becomes limited only by the nutrient level. The result is that the eutrophication level increases considerably with increased nutrient levels. The entire structure of the ecosystem is changed because phytoplankton is now limited by nutrients, zooplankton are limited by predation from planktivorous fish rather than by the available food source (phytoplankton), and the planktivorous fish are now limited by the food source (zooplankton) rather than by the carnivorous fish.

This implies that other properties are selected (e.g., other growth rates). Further, the carnivorous fish will be scarce since they mostly hunt by sight, with the increasing eutrophication causing an increased turbidity and, therefore, making hunting more difficult.

If nutrient levels are now decreased, for example, by discharging wastewater with significantly lower nutrient concentrations, will this "high eutrophication structure", characterized with a low concentration of carnivorous fish, high concentration of planktivorous fish, low concentration of zooplankton, and high concentration of phytoplankton, change immediately?

Thus, the low transparency will still make an increase in the carnivorous fish difficult to achieve, and grazing will still not able to control the phytoplankton concentration. This is because they are present in relatively low concentrations, since they are still under stress from considerable predation pressure by planktivorous fish. In this situation, will biomanipulation either by removal of planktivorous fish and/or introduction of more carnivorous fish, give a clear effect, as shown in Figure 5.17? Biomanipulation simply provides a faster recovery to the "low eutrophication structure", where grazing controls the phytoplankton concentration. Outside the intermediate nutrient level, the forcing functions (input of nutrients) determine the state of eutrophication.

Models are a useful tool for trying to understand the function of biomanipulation, including the fact that it sometimes gives positive results, and sometimes negative results. Models generally should be used as a tool to try to qualitatively understand the behavior of complex systems (e.g., lake ecosystems). When a good data base is available, mod-

Table 5.11. Some sediment oxygen demand values (g $O_2$ $m^{-2}$ day)

| Bottom type | Range | Average |
|---|---|---|
| Filamentous bacteria (10 g $m^{-2}$ dry wt.) | 5–10 | 7 |
| Municipal sewage sludge—out fall vicinity | 2–10 | 4 |
| Municipal sewage sludge—downstream of outfall | 1–2 | 1.5 |
| Estuarine mud | 1–2 | 1.5 |
| Sandy bottom | 0.2–1.0 | 0.5 |
| Mineral soils | 0.05–0.1 | 0.07 |

els can sometimes be used quantitatively in environmental management, an example being the control of eutrophication by wastewater management (e.g., see Jørgensen and Vollenweider, 1988; Jørgensen, 1986b, 1994). The use of models in a quantitative sense for biomanipulation efforts is possible by applying structural dynamic models. However, because our experience with this latter type of models is limited, the modelling results should not be the sole management decision criteria, although the results may be used in a qualitative way. Thus, there is a clear need for the use of models in management and selection of biomanipulation techniques to try to answer such questions as: Can biomanipulation give the desired results? How many planktivorous fish must be removed and/or how many carnivorous fish must be added? How many times, and with which frequency, should biomanipulation be used to ensure the anticipated results? Such questions can only rarely be answered today with the use of models, primarily because the data requirements would be too great for most case studies. Thus, it is recommended that more case studies on the use of models under such conditions be developed and supported with good data sets. Only with increased experience in using models for biomanipulation purposes can the urgently needed management tools be developed.

### 5.5.7 *Most Commonly-Applied Dynamic Water Quality Models and DDS*

Some of the commonly-applied models for practical water quality management of lakes and reservoirs, particularly deep, stratified ones, belong to the category of coupled hydrodynamic–ecological models (Orlob, 1983, 1984; Varis, 1994). The reason for coupling detailed hydrodynamics (Martin and McCutcheon, 1999) and basic chemical and biological processes is that hydrodynamic conditions vary widely, depending on changes in inflows and meteorology. The variability is particularly great in reservoirs under various operational regimes. The depth and time distribution of the chemical compounds in them is highly affected by currents and other movements of water masses, and the biological processes react on physical conditions both directly and indirectly via chemistry.

The coupled hydrodynamic–ecological models are used to simulate time-dependent water quality conditions under different meteorological conditions, eutrophication or its abatement, and increasing or decreasing pollution, in order to evaluate the consequences of different management strategies. Further, the conditions existing shortly after a reservoir is filled (the aging process—see Section 6.5.4) can be simulated, and also the conditions in reservoirs subject to different operation regimes, undergoing dam reconstruction and/or

turbine repair. With modern personal computers, the simulation of one-year cycles can be done in a few minutes, and the *scenario analysis* (comparative analysis of a number of model runs) can be used as a basis for making decisions. The manager can see, with the graphical model outputs, the probable consequences of different variants of management actions under different hydrological and meteorological situations.

The input conditions needed to run the models are lake morphometry, annual cycles of hydrology and meteorology, and chemical and biological conditions of the inflows. Particularly important is the knowledge (or reasonable assumptions about) the inflow water density, particularly as determined by temperature and salinity. The morphometry is characterized particularly by the area and volume curves. In the case of reservoirs, it also is characterized by the slope of the bottom and sides of the reservoir. The character of the outflow structure and operation regimes are important as well.

The *hydrodynamic part* of the model provides information on the physical structure of the water masses under different external conditions. Its final output is usually the water temperature stratification. However, for the chemical and biological components, the intensity of mixing and exchange among different strata is the decisive output used. The hydrodynamic models are categorized according to the space dimensions being considered, as one-, two- or three-dimensional.

*One-dimensional models* provide the depth distribution of the physical properties in the deepest part of the waterbody and the water movements. In addition, *two-dimensional models* provide the longitudinal profile, which is important for reservoirs. The most complicated are the *three-dimensional models*, which treat the full volume of a waterbody in both the horizontal and vertical directions. The one-dimensional models are adequate for most management purposes, but two-dimensional models may be necessary if the knowledge of the longitudinal distribution in a reservoir is important for a manager. The use of a three-dimensional model may be necessary for large lakes, as well as lakes and reservoirs with complicated shapes.

The *ecological part* of the model provides information on the chemical and biological components of interest to the manager. The biological part need not be directly used by the manager, but must be considered because of the close interactions between water chemistry and biology. Only the conservative chemical components (e.g., salinity) do not strongly interact with the biological components, although most chemical variables are either directly used and/or released by organisms, or are affected by other chemical changes produced during photosynthesis, respiration and decomposition. As an example, changes in pH and oxygen conditions in a lake or reservoir can affect most chemical reactions. There also is a feedback between lake thermics and phytoplankton, in that higher radiation adsorption by phytoplankton can lead to more heat accumulation in the upper water layer, surface temperatures increase and the thermocline will be located at a shallower depth. Phytoplankton growth is enhanced by the higher temperature (Straškraba, 1998a).

Few of the existing models have found broad application (Table 5.12). In Russia and Ukraine, similar models have been developed by Oleg Vasiliev and co-workers (Vladivostok), mainly from the hydrodynamic perspective, and by Vladimir Lavrik (Kiev) mainly from the mathematical perspective (e.g., Lavrik et al., 1990). In addition to the

Table 5.12. Examples of coupled hydrodynamic–ecological models

- LAKE ONTARIO model (Scavia and Chapra, 1977) is a multi-purpose ecological model developed specifically for the Great Lakes, particularly Lake Ontario.
- SALMO (Benndorf and Recknagel, 1982) includes schematic distinction into epilimnion and hypolimnion, interactions between phytoplankton biomass and changing external nutrient load, temperature, light, mixing and zooplankton. SALMOSED (Recknagel and Benndorf, 1995) includes exchange of phosphorus between water and sediments.
- RESQUAL by Stefan and Cardoni (1982) was applied to Lake Chicot and was the basis for further models designed by the group associated with Heinz Stefan at the University of Minnesota (see MINLAKE).
- CE-QUAL-RIV1 is a derivative of the model CE-QUAL for water quality in larger rivers (Bedford et al., 1983; Anonymous, 1986—manual for application). The model was originally destined "for crude planning analysis of reservoir eutrophication" (Wlosinski and Collins, 1985), but was continuously extended into two-dimensional forms. The model is distributed free by the USEPA and is broadly used, particularly in the Americas (e.g., Anonymous, 1995; Diogo and Rodrigues, 1997).
- FINNISH THREE DIMENSIONAL WATER QUALITY–TRANSPORT MODEL by Virtanen et al. (1986) concentrates first of all on longitudinal flows in complex shaped reservoirs (Koponen et al., 1998).
- WASP4 (Ambrose et al., 1988) includes hydrodynamics, conservative mass transport, eutrophication, oxygen and toxic chemical-sediment dynamics. The model is also distributed free by USEPA, including documentation. The model is used in the USA for many predictions, a recent list mentioned, for example, by Tufford et al. (1998).
- MINLAKE (Riley and Stefan, 1988) was recently extensively modified to allow estimation of the effect of climate changes on USA lakes (Stefan and Fang, 1994a) and dissolved oxygen conditions for regional lake analysis (Stefan and Fang, 1994b).
- MIKE (Anonymous, 1998a) is originally a model with a simplified two-layer representation. A new version MIKE11 was presented by the Danish Hydraulic Institute (Agern Alle 5, Horsholm) and represents a two-dimensional vertical hydrodynamic flow model coupled with an eutrophication model covering 12 state variables, including inorganic nutrients, phytoplankton, zooplankton, detritus and dissolved oxygen. Used in Denmark and England, but also applied elsewhere. A number of additional models for estimating the loads from watershed (CATCHMENT, LOAD, WETLAND) and other water quality processes are also available (Anonymous, 1998a).
- ASTER and MELODIA (Salençon and Thèbauld, 1994a, 1994b, 1995) were applied in France to model stratification and biological conditions in Pareloup Reservoir. The hydrodynamic part represents a separate model, EOLE (Salençon, 1997).
- DYRESM–WQ, the DYnamic REServoir Model–Water Quality (Schladow et al., 1994; Hamilton and Schladow, 1997; Schladow and Hamilton, 1995, 1997; Hamilton and De Stasio, 1998; Hamilton, 1999) is based on the hydrodynamic model DYRESM by Imberger and Patterson (1981). The water quality model simulates stratification and flow conditions, phytoplankton, nutrients (different forms of phosphorus and nitrogen), oxygen, biochemical oxygen demand (BOD), iron and manganese, and sedimentation. It is being systematically extended, with CE-QUAL the most widely used water quality model. However, the cost of the model available from the Water Research Centre (University of Western Australia, Nedlands, Western Australia) is high. A standard one-dimensional, a two-dimensional hydrodynamic (Hocking and Patterson, 1988, 1991), and most recently a three-dimensional water quality version (Romero and Imberger, 1999), also exist. An extension of the model to simulate the consequences of artificial destratification was formulated by Schladow and Hamilton as used, for example, by Lindenschmidt and Chorus (1997) and Schlenkhof (1997).

Table 5.12 (*continued*)

- BEEKWAM is intended for nonstratified embankment reservoirs in The Netherlands. It combines an empirical equilibrium temperature model with differential equations for nutrients and algal growth. Matlab® and Simulink® are used as simulation tools (Benoist et al., 1998).
- PROTECH–C (from Phytoplankton RespOnses To Environmental Change) is a two-dimensional model directed to detailed representation of species specific phytoplankton reactions on external effects, including the meteorological, hydrological and anthropogenic effects (Anonymous, 1998b). The model from the Algal Modelling Unit (Institute of Freshwater Ecology, Far Sawrey, Ambleside, Cumbria, UK) is based on extensive scientific phytoplankton studies by Reynolds (1987, 1992, 1998, 1999a, 1999b) and is used by a number of water quality agencies in UK.

models listed in Table 5.12, some of the more complex ecological models without elaborate hydrodynamics have been used for water quality management decisions, particularly for the more simple, shallow waterbodies, including:

- The model of Jørgensen et al. (1978), which was first applied to Lyngby Lake during 1952–1958. The model also closely matched the conditions which later changed during 1959–1975.
- AQUAMOD, the AQUAtic MODel by Straškraba (1976) is a simple eutrophication-oriented model. In the extended version, the model GIRL OLGA (mentioned below) was used for Czechoslovakian reservoirs. The model also exists in a two- and three-layer version, including sediments (Straškraba and Gnauck, 1985; Dvorkov and Kozerski, 1980). Gnauck et al. (1990) have used the modelling system SONCHES to apply AQUAMOD in Germany.
- BLOOM, particularly in the form of BLOOM II (Los and Brinkman, 1988; Los, 1991), examines the eutrophication of shallow lakes, and was systematically extended into DELWAG-BLOOM-SWITCH for control of eutrophication of lake and channel systems in The Netherlands (van der Molen et al., 1994).
- The CONSTANT SEDIMENT FEEDBACK PHOSPHORUS MODEL and WATER–SEDIMENT MODEL by Chapra and Canale (1991) concentrates on phosphorus conditions, particularly its exchange with the sediments. Thus, they are relevant for evaluating the consequences of phosphorus abatement strategies.
- PCLAKE, the shallow lake model designed specifically by Janse et al. (1993) for use on personal computers is systematically being extended to increase its usefulness for managing shallow lakes, particularly with respect to eutrophication and attempts to treat it using biomanipulation (Janse et al., 1995). It is also being applied to the problem of two multiple stable states of shallow lakes—the phytoplankton state and macrophyte state (Janse, 1997; Janse et al., 1998).
- The BALATON model by Kutas and Herodek (1986) is a complex model specifically designed to aid in the management of shallow Lake Balaton, characterized by high calcium content low amounts of zooplankton and high importance of bacterial processes.

These complex models are often framed into decision support systems (DSS), as described in Section 5.4. Presented below is a list of some DSS which may prove useful to a lake or reservoir water quality manager (Table 5.5). One part of a DSS often is the

GIS-based determination of pollutant loads, as well as evaluation of a lake water quality model, as discussed further in a following section.

### 5.5.8 *Environmental Risk Assessment (ERA)*

ERA can be defined as the process of assigning magnitudes and probabilities to the adverse effects of human activities. The process involves identification of hazards (e.g., the release of toxic chemicals to the environment) by quantification of the relationship between an activity associated with an emission to the environment and its impacts. The entire ecological hierarchy is considered in this context, implying that the impacts on the cellular level, organism level, population level, ecosystem level and the entire ecosphere should be considered.

The application of environmental risk assessment is rooted in the recognition that:

- The cost of elimination of all environmental effects is impossibly high,
- The decision in practical environmental management must always be made on basis of incomplete information.

Humans use about 100,000 chemicals in such quantities that they may threaten the environment. However, we only know a small fraction about what we must know about these chemicals to be able to make a proper and complete environmental risk assessment of them.

ERA is a process complementary to environmental impact assessment, EIA (Jørgensen, 1991), with the latter used to assess the impacts of a human activity. EIA is predictive, comparative and concerned with all possible effects on the environment, including secondary and tertiary (indirect) effects, while ERA attempts to assess the probability of a given (defined) adverse effect resulting from a considered human activity.

Both ERA and EIA use models to find the *expected environmental concentration* (EEC), which is translated into impacts for EIA, and to risks of specific effects for ERA.

Uncertainty plays an important role in risk assessment. Risk is the probability that a specified harmful effect will occur or, in the case of a graded effect, the relationship between the magnitude of the effect and its probability of occurrence.

Risk assessment has emphasized risks to human health, ignoring ecological effects to a certain extent. However, it has now been acknowledged that some chemicals with little or no risk to human health can cause severe effects to aquatic and other organisms. Examples include chlorine, ammonia and certain pesticides. An up-to-date risk assessment, therefore, comprises consideration of the entire ecological hierarchy, which is the ecologist's perspective in terms of level of organization. Organisms interact directly with the environment, and they are exposed to toxic chemicals. The reproducing population is the smallest meaningful level in the ecological sense. However, populations do not exist in vacuum, but rather within a community of other organisms of which the population is a part. The community occupies a physical environment, with which it forms an ecosystem.

Moreover, both the various adverse effects and the ecological hierarchy have different scale in time and space, which must be included in a proper environmental risk assessment (Fig. 5.18). For example, oil spills occur at a spatial scale similar to those of populations,

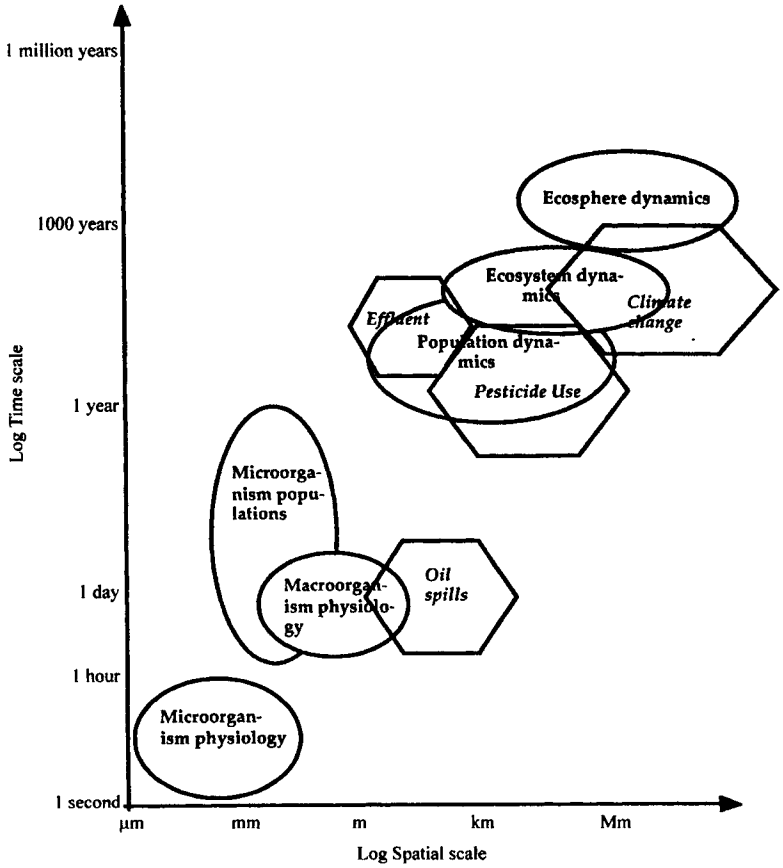



Fig. 5.18. The spatial and time scale for various hazards (hexagons, italic) and for the various levels of the ecological hierarchy (circles, nonitalic).

but they are briefer events than population processes. Thus, a risk assessment of a oil spill requires considerations of reproduction and recolonization that typically occur on a longer time scale than the spill, and that determine the magnitude of the population responses and its significance to natural population variance safety factors. Uncertainties have three basic causes:

- The inherent randomness of the world (i.e., stochasticity),
- Errors in execution of assessment,
- Imperfect or incomplete knowledge.

The inherent randomness refers to the uncertainty that can be described and estimated, but cannot be reduced because it is a characteristic of the system. The meteorological factors (e.g., rainfall, temperature, wind) are effectively stochastic at levels of interest for risk

assessment. Many biological processes, such as colonization, reproduction and mortality, also must be described stochastically.

Human errors are inevitably attributes of all human activities. This type of uncertainty includes incorrect measurements, data recording errors, computational errors, etc.

Lack of knowledge results in undefined uncertainty that cannot be described or quantified. It is a result of practical constraints on our ability to accurately describe, count, measure or quantify everything relevant to an estimate of risks. Prominent examples are an inability to test all toxicological responses of all species exposed to a pollutant, and simplifications needed in a model to predict the expected environmental concentration.

The most important feature distinguishing risk assessment from impact assessment is the emphasis in risk assessment on characterizing and quantifying uncertainty. Thus, it is of particular interest in risk assessment to be able to analyze and estimate the analyzable uncertainty. They include natural stochasticity, parameter errors and model errors. Statistical methods may provide direct estimates of uncertainties, and are widely used in model development (e.g., see Jørgensen, 1994).

The use of statistics to quantify uncertainty is complicated in practice by the need to consider errors in both the dependent and independent variables, and to combine errors when multiple extrapolations should be made. Monte Carlo analysis is often used to overcome these difficulties (e.g., see Bartell et al., 1984).

Model errors include inappropriate selection or aggregation of variables, incorrect functional forms and incorrect boundaries. The uncertainty associated with model errors are usually assessed by field measurements utilized for calibration and validation of the model (Jørgensen, 1994).

Risk assessment of chemicals can be divided into nine steps (Fig. 5.19). The nine steps correspond to questions the risk assessment attempts to answer in order to quantify the risk associated with the use of a chemical, as follows:

Step 1. Which hazards are associated with the application of the chemical? Answering this question involves gathering data on the types of hazards, including possible environmental damages and human health effects. The health effects include congenital, neurological, mutagenic and cancerogenic effects. It may also include characterization of the behavior of the chemical within the body (interactions with organs, cells or genetic material). What is the possible environmental damage, including lethal effects and sublethal effects on growth and reproduction of various populations?

A variety of toxicity tests has been devised to attempt to quantify the potential danger posed by chemicals. Some recommended tests involve experiments with subsets of natural systems (i.e., microcosms) or with entire ecosystems. The majority of the testing of new chemicals for possible effects, however, has been confined to laboratory studies on a limited number of test species. Results from these laboratory assays provide useful information for quantifying the relative toxicity of different chemicals. They are used to forecast effects in natural systems, although their justifications have been seriously questioned.

Step 2. What is the relation between dose and responses of the type defined above in Step 1? It implies knowledge of no effect concentration (NEC), $LD_x$-, $LC_y$- and

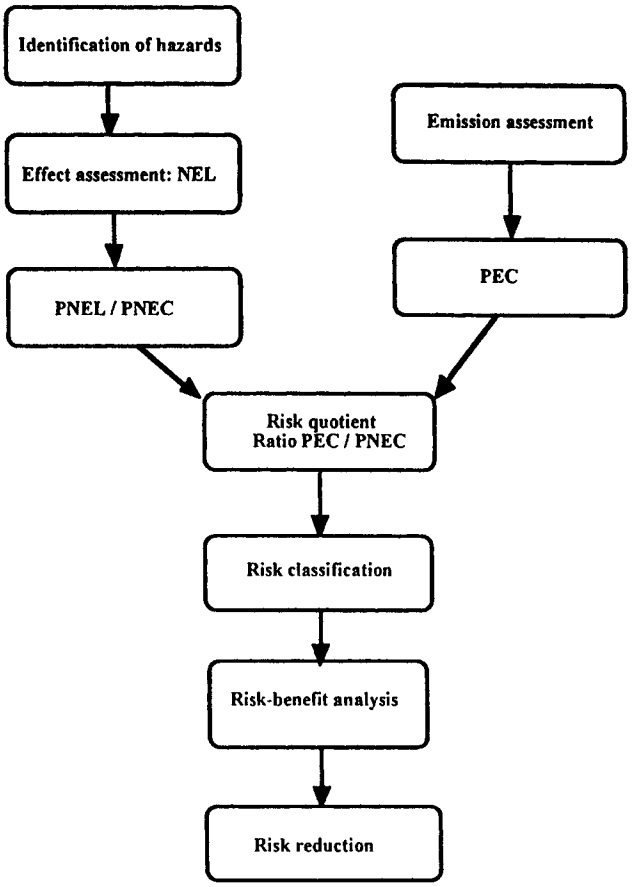



Fig. 5.19. The procedure for a nine-step risk assessment of chemical compounds. Steps 1–3 require extensive use of ecotoxicological handbooks and ecotoxicological estimation methods to assess the toxicological properties of the considered chemical compounds, while Step 5 requires a selection of a proper ecotoxicological model (PEC—predicted environment concentration; PNEL—predicted non-effect level; PNEC—predicted no effect concentration).

$EC_z$-values, where $x$, $y$ and $z$ express a probability of harm. The answer can be found by laboratory examinations or estimation methods. Based on these answers, a most probable *level of no effect*, *NEL*, is assessed.

Data needed for Steps 1 and 2 above can be obtained directly from scientific libraries. They also are increasingly being found via online data searches in bibliographic and factual databases. Data gaps should be filled with estimated data (e.g., see Jørgensen et al., 1997).

Step 3. Which uncertainty (safety) factors reflect the degree of uncertainty that must be taken into account when experimental laboratory data or empirical estimations methods

are extrapolated to real situations? Safety factors ranging from 10 to 10,000 are typically used. If adequate knowledge about a chemical is available, a safety factor of 10 may be applied. On the other hand, if the available information has a high uncertainty, a safety factor of 10,000 may be appropriate. Safety factors ranging between 100–1000 are most frequently applied. The result of $NEL_x$, the safety factor, is called the *predicted non-effect level, PNEL*. The complexity of environmental risk assessment is often simplified by deriving *predicted no effect concentration, PNEC*, for different environmental compartments (e.g., water, soil, air, biota, sediment).

Step 4. What are the sources and quantities of emissions? The answer to this question requires a complete knowledge of the production and use of the chemical compounds of concern, including an assessment of how much of the chemical is wasted to the environment by production and use? The chemical also may be a waste product, making it very difficult to determine the amounts involved. The very toxic dioxins, for example, are waste products of the incineration of organic waste.

Step 5. What is (are) the actual exposure concentration(s)? The answer to this question is called the *predicted environmental concentration, PEC*. Measuring environmental concentrations can assess exposure. It also may be predicted with a model when the emissions are known. The use of models is necessary in most cases, either because we are considering a new chemical, or because the assessment of environmental concentrations require a very high number of measurements to determine the variations in concentrations over time and space. Further, it always provides an additional degree of certainty to compare model results with measurements, implying that it is always advisable both to develop a model and to make at least a few measurements of concentrations in the ecosystem components, where the highest concentrations are expected. Most models will require an input of parameters which describe the properties of the chemicals and the organisms, and which also will require an extensive application of handbooks and a wide range of estimation methods.

The development of an environmental, ecotoxicological model, therefore, requires an extensive knowledge of the physical–chemical–biological properties of the considered chemical compound(s). The selection of a proper model is discussed in Section 5.4.

Step 6. What is the ratio *PEC/PNEC*? This ratio is often called the risk quotient. It should not be considered an absolute assessment of risks, but rather a relative ranking of risks. The ratio is usually found for a wide range of ecosystems (e.g., aquatic ecosystems, terrestrial ecosystems, ground water).

Step 7. How will one classify the risk? The valuation of risks are made in order to decide on risk reductions (Step 9). Two risk levels are defined:

- The upper limit (*maximum permissible level, MPL*), and
- The lower limit (*negligible level, NL*). It may also be defined as a percent of the *MPL*.

The two risk limits create three zones: A black, unacceptable, high risk zone > *MPL*, a gray, medium risk level, and a white, low risk level < *NL*.

Step 8. What is the relation between risk and benefit? This analysis involves examining socioeconomic, political and technical factors, which is beyond the topic of this volume.

The cost-benefit analysis is also difficult, because the costs and benefits are often of a different order.

Step 9. How can the risk be reduced to an acceptable level? The answer to this question requires a thorough technical, economic and legislative investigation before it can be given. Assessment of alternatives is often an important aspect in risk reduction.

Steps 1–3 and 5 require knowledge of the properties of the chemical compounds of concern, which again implies an extensive literature search and/or selection of the best feasible estimation procedure.

### *5.5.9 Structurally Dynamic Models*

Our present models are built on generally-rigid structures and a fixed set of parameters. No change or replacement of components is usually possible. However, we also may need to introduce parameters (properties) which can vary according to changing general conditions for the state variables (components). The idea currently is to test whether a change of the most crucial parameters produces a higher so-called goal function of the system and, if so, to use that set of parameters.

As indicated in the introduction to this section, models that can account for change in species composition, and for the ability of the species (i.e., the biological components of the models) to change their properties (i.e., to adapt to the prevailing conditions imposed on them), are called *Structurally Dynamic Models*. (This model type was mentioned in Section 5.5.6 as they can explain success and failure of biomanipulation.) They also may be called the *next* or *fifth generation* of ecological models, in order to underline that they are radically different from previous modelling approaches and can do more, including describing changes in species composition.

It could be argued that the ability of ecosystems to replace current species with others that are better adapted to the ecosystem can be addressed by constructing models which incorporate all possible species for the entire study period. However, there are two essential disadvantages. First, such models become very complex, since they will contain many state variables for each trophic level. Thus, many more of them need to be calibrated and validated, which will introduce a high uncertainty, and render application of the model very case-specific (Nielsen, 1992a, 1992b). In addition, the model will still be rigid and not replicate the ecosystem property of parameters that are continuously changing without changing species composition (Fontaine, 1981).

Several goal functions have been proposed, but only a few models have been developed that can account for change in species composition, or for the ability of the species to change their properties within some limits. Straškraba (1979) used maximization of biomass as the governing principle (the goal function). The model adjusts one or more selected parameters, in order to achieve maximum biomass at every instance. A modelling software routine is included which computes biomass for all possible combinations of parameters within a given realistic range. The combination that gives the maximum biomass is selected for the next time step, etc. However, the biomass can hardly be used in models with more

trophic levels. To add the biomass of fish and phytoplankton together, for example, will lead to biased results.

The thermodynamic variable, exergy, has been used as a goal function in some ecological models. One of the few lake case studies available is discussed below. There are two pronounced advantages of exergy as a goal function, compared to entropy and maximum power (Odum, 1983). These are that (1) it is defined far from thermodynamic equilibrium, and (2) it is related to state variables which are easily determined or measured. However, because exergy is not a common thermodynamic function, it is first necessary to describe it.

Exergy expresses energy with a built-in measure of quality, quantifies natural resources, and can be considered as fuel for any system that converts energy and matter in a metabolic process (Schrödinger, 1944). Thus, ecosystems convert exergy, with exergy-flow through the system being necessary to keep it functioning. Exergy (as the amount of work the system can perform when brought into equilibrium with its environment) measures the distance from the "inorganic soup" in energy terms, as will be explained further below. Thus, exergy is dependent both on the environment and the system, rather than the system alone. This means, for example, that it is not a state variable like free energy and entropy.

If we assume a reference environment representing an ecosystem at thermodynamic equilibrium, the configuration illustrated in Figure 5.19 is valid. All components are inorganic, at the highest possible oxidation state, and homogeneously distributed in the system. Thus, as much free energy as possible is utilized to perform work (no gradients). As the chemical energy embodied in the organic components and the biological structure contributes the most by far to the exergy content of the system, there seems to be no reason to assume a (minor) temperature and pressure difference between it and the reference environment. Under these circumstances, we can calculate the exergy content of the system provided entirely by chemical energy as $\Sigma_c(\mu_c - \mu_{ceq})N_i$. We can also determine its exergy, compared with that of a similar system at the same temperature and pressure, but in the form of an inorganic soup without any life, biological structure, informational or organic molecules. Because $(\mu_c - \mu_{ceq})$ can be found from the definition of the chemical potential (replacing activities by concentrations), the following expressions for the exergy can be obtained:

$$Ex = RT \sum_{i=0}^{n} C_i \ln \frac{C_i}{C_{i,eq}}. \tag{5.6}$$

As seen, exergy measures the difference between free energy (given the same temperature and pressure) of an ecosystem, and that of the surrounding environment. If the system is in equilibrium with its surroundings, exergy is zero, with $n = 0$ accounting for inorganic compounds, and $n = 1$ corresponding to detritus.

Since the only way to shift systems away from equilibrium is to perform work on them, and since the available work in a system is a measure of its ability to do so, we need to distinguish between the system and its environment (or thermodynamic equilibrium alias the inorganic soup). Thus, it is reasonable to use the available work (i.e., exergy) as a measure of the distance from thermodynamic equilibrium.

Survival implies the maintenance of biomass, while growth means its increase. Exergy is needed to construct biomass, which then possesses exergy transferable to support other processes. Thus, survival and growth can be measured by use of the thermodynamic concept exergy, which may be understood as *the free energy relative to the environment* (see Equation 9.31).

Darwin's theory of natural selection (1859), therefore, may be reformulated in thermodynamic terms and expanded to an ecosystem level as follows: *The prevailing conditions of an ecosystem steadily change. The system will continuously select those species which contribute most to the maintenance, or even to the growth of the exergy of that system.*

Notice that the thermodynamic translation of Darwin's theory requires *populations* to possess properties of reproduction, inheritance and variation. The selection of species that contribute most to the exergy of the system under the prevailing conditions requires that there are sufficient individuals with different properties for selection to take place. Reproduction and variation must be high and, once a change has taken place, it must be conveyed to the next generation via better adaptation. It is also noted also that a change in exergy is not necessarily $\geqslant 0$, but rather depends on the resources of the ecosystem. However, the above proposition claims that an ecosystem attempts to reach the highest possible exergy level under the given circumstances with the available genetic pool (Jørgensen and Mejer, 1979).

It is not possible to measure exergy directly. However, it is possible to compute it if the composition of the ecosystem is known. $C_i$ represents the $i$th component expressed in a suitable unit (e.g., for phytoplankton in a lake, $C_i$ mg nutrient in the phytoplankton/liter of lake water) and $C_{i,\text{eq}}$ is the concentration of the $i$th component at thermodynamic equilibrium. For detritus, exergy can be found on basis of equilibrium constants, which give the ratio between the concentration of detritus in the ecosystem and that at thermodynamic equilibrium. The exergy content of detritus is approximately 18.7 kJ $g^{-1}$, which can be compared with the exergy (chemical energy) content of mineral oil (about 42 kJ $g^{-1}$).

For more complex compounds (e.g., unicellular organisms), $C_{i,\text{eq}}$ will be smaller. The probability of forming one *coli* bacterium or a simple phytoplankton cell can be estimated from the number of genes, which again determine the sequence of amino acids. It has been found by this method (Jørgensen et al., 1994) that the exergy content of phytoplankton is 3.4 times greater than that of detritus, because of their information content. Of course, the probability of forming multi-cell organisms at thermodynamic equilibrium is even lower, in that additional exergy is required to make up the more complex structure and to provide the information embedded in its structure. Thus, the exergy content of zooplankton is provisionally estimated to be around 35 times that of detritus, and fish to be more than 300 times. Such conversion factors for several classes of organisms are listed in Table 5.13.

Of course, proposals for the concentrations of various biological components at thermodynamic equilibrium do not lead to any exact value of exergy, or even of its exact relative change. However, they will account for relative changes caused by variations in the properties (parameters) of *organisms*. The inorganic constituents of an ecosystem do not create similar computational difficulties, but the thermodynamic equilibrium concentrations will be the total concentrations of the various elements, corresponding to the fact that all chem-

Table 5.13. Approximate number of nonrepetitive genes (Cavalier-Smith, Li and Grauer, Lewin)

| Organisms | Number of information genes | Conversion factor* |
|---|---|---|
| Detritus | 0 | 1 |
| Minimal cell (Morowitz, 1992) | 470 | 2.7 |
| Bacteria | 600 | 3.0 |
| Algae | 850 | 3.9 |
| Yeast | 2000 | 6.4 |
| Fungus | 3000 | 10.2 |
| Sponges | 9000 | 30 |
| Moulds | 9500 | 32 |
| Plants trees | 10,000–30,000 | 30–87 |
| Worms | 10,500 | 35 |
| Insects | 10,000–15,000 | 30–46 |
| Jellyfish | 10,000 | 30 |
| Zooplankton | 10,000–15,000 | 30–46 |
| Fish | 100,000–120,000 | 300–370 |
| Birds | 20,000 | 390 |
| Amphibians | 120,000 | 370 |
| Reptiles | 130,000 | 400 |
| Mammals | 140,000 | 430 |
| Human | 250,000 | 740 |

*Based on number of information genes and the exergy content of the organic matter in the various organisms, compared with the exergy contained in detritus; 1 g detritus has about 18 kJ exergy (= energy which can do work).

ical compounds in the primeval soup were in inorganic form, which we often use as the reference state for our exergy calculations.

The last few pages have presented the theoretical background for the application and development of structurally dynamic models. These are important tools in environmental management, as they account for current changes of species composition and the properties of the organisms in the focal ecosystem. The idea of the new generation of models presented here is to find a new set of parameters (limited for practical reasons to the most crucial, i.e., the most sensitive) better suited to the prevailing conditions of the ecosystem, as defined in the Darwinian sense by the ability of the species to survive and grow. As indicated above, this may be measured by the use of exergy (Jørgensen, 1986b, 1994; Jørgensen and Mejer, 1977, 1979). Figure 5.16 illustrates the proposed modelling procedure which has been applied in the cases presented.

The use of exergy calculations continuously to vary parameters has only been employed in nine case studies of biogeochemical modelling. One of these (Søbygaard Lake) is described here as an illustration of what can be achieved with this approach. The results (Jeppesen et al., 1989) are particularly fitted to test its applicability to structural dynamic models.

Søbygaard is a shallow lake (depth of 1 m) with a short retention time (15–20 days). The phosphorus nutrient load was significantly lowered in 1982, from 30 g to 5 g P $m^{-2}$ yr.

However, the decreased load did not result in reduced nutrient and chlorophyll concentrations during the period 1982–1985 because of internal loading of stored nutrients in the sediment (Jeppesen et al., 1989).

Radical changes were then observed during the period 1985–1988. Recruitment of planktivorous fish was significantly reduced during the interval 1984–1988 because of a very high pH. The zooplankton increased, and the phytoplankton decreased in concentration, with the average summer chlorophyll-a concentration reduced from 700 $\mu g\ l^{-1}$ in 1985 to 150 $\mu g\ l^{-1}$ in 1988. The phytoplankton population even collapsed during shorter periods because of extremely high zooplankton concentrations.

Simultaneously, phytoplankton species increased in size. Their growth rates declined and higher settling rates were observed (Kristensen and Jensen). In other words, this case study illustrates that pronounced ecosystem structural changes were caused by biomanipulation-like events. However, the primary production was not higher in 1985 than in 1988, because of pronounced self-shading by smaller algae. Thus, it was very important to include a self-shading effect in the model. Simultaneously, sloppier feeding of zooplankton was observed, with a shift from *Bosmina* to *Daphnia* taking place.

The model contains six state variables, all of which represent different forms of nitrogen, including fish, zooplankton, phytoplankton, detritus nitrogen, soluble nitrogen and sedimentary nitrogen. The model equations are given in Table 5.14. Because nitrogen is the limiting nutrient for eutrophication in this particular case, it may be sufficient to only include this element in the model.

The aim of the study is to describe, by use of a structural dynamic model, the continuous changes in the most essential parameters, using the procedure shown in Figure 5.16. The data from 1984–1985 were used to calibrate the model. The two parameters which it was intended to change for the period 1985 to 1988 received the following values:

| | |
|---|---|
| Maximum phytoplankton growth rate: | 2.2/ day |
| Phytoplankton settling rate: | 0.15/day |

The state variable, fish nitrogen, was kept constant at 6.0 during the calibration period. During the period 1985–1988, however, an increased fish mortality was introduced to reflect the increased pH. Thus, fish stock was reduced to 0.6 mg N $l^{-1}$; notice the equation "mort = 0.08 if fish > 6 (may be changed to 0.6) else almost 0". A time step of $t = 5$ days and $x\% = 10\%$ was applied (Fig. 5.16). This means that nine runs were needed for each time step, in order to select the parameter combination giving the highest exergy. Changes in parameters from 1985 to 1988 (summer) are summarized in Table 5.15. It may be concluded that the proposed procedure (Fig. 5.16) can approximately simulate the observed change in ecosystem structure.

The maximum phytoplankton growth rate is reduced by 50% from 2.2/day to 1.1/day, approximately in accordance with to the increase in size. It was observed that the average size was increased from a few 100 $\mu m^3$ to 500–1000 $\mu m^3$, a factor of 2–3 (Jeppesen et al., 1989). This would correspond to a specific growth reduction by a factor $f = 2^{2/3} - 3^{2/3}$ (Jørgensen, 1994; Jørgensen et al., 2000).

Table 5.14. Model equations for Søbygaard Lake

fish = fish + $dt$ × (−mort + predation)
INIT(fish) = 6
$na = na + dt$ × (uptake − graz − outa − mortfa − settl − setnon)
INIT($na$) = 2
$nd = nd + dt$ × (−decom − outd + zoomo + mortfa)
INIT($nd$) = 0.30
$ns = ns + dt$ × (inflow − uptake + decom − outs + diff)
INIT($ns$) = 2
nsed = nsed + $dt$ × (settl − diff)
INIT(nsed) = 55
$nz = nz + dt$ × (graz − zoomo − predation)
INIT($nz$) = 0.07
decom = $nd$ × (0.3)
diff = (0.015) × nsed
exergy = $\text{total}_n$ × (Structural-exergy)
graz = (0.55) × $na$ × $nz/(0.4 + na)$
inflow = 6.8 × $qv$
mort = IF fish > 6 THEN 0.08 × fish ELSE 0.0001 × fish
mortfa = (0.625) × $na$ × $nz/(0.4 + na)$
outa = $na$ × $qv$
outd = $qv$ × $nd$
outs = $qv$ × $ns$
pmax = uptake × 7/9
predation = $nz$ × fish × $0.08/(1 + nz)$
$qv$ = 0.05
setnon = $na$ × 0.15 × (0.12)
settl = (0.15) × 0.88 × $na$
Structural-exergy = ($nd$ + $\text{nsed/total}_n$) × (LOGN($nd$ + $\text{nsed/total}_n$) + 59) + ($ns/\text{total}_n$) × (LOGN($ns/\text{total}_n$) − LOGN($\text{total}_n$)) + ($na/\text{total}_n$) × (LOGN($na/\text{total}_n$) + 60) + ($nz/\text{total}_n$) × (LOGN($nz/\text{total}_n$) + 62) + ($\text{fish/total}_n$) × (LOGN($\text{fish/total}_n$) + 64)
$\text{total}_n = nd + ns + na + nz$ + fish + nsed
uptake = $(2.0 - 2.0 \times (na/9)) \times ns \times na/(0.4 + ns)$
zoomo = 0.1 × $nz$

Thus:

$$\text{the growth rate in 1988} = \text{the growth rate in 1985}/f, \tag{9.32}$$

where $f$ is between 1.58 and 2.08. In the above table, the value of 2.0 is found with the use of the structurally dynamic modelling approach. Kristensen and Jensen observed that settling was 0.2 m day$^{-1}$ (range 0.02–0.4) during 1985, but 0.6 m day$^{-1}$ (range 0.1–1.0) in 1988. Using the structurally dynamic modelling approach, the increase was found to be 0.15 m to 0.45 m day$^{-1}$, a slightly lower set of values. However, the same (3) phy-

Table 5.15. Parameter combinations giving the highest exergy

| | Maximum growth rate ($day^{-1}$) | Settling rate ($m\ day^{-1}$) |
|---|---|---|
| 1985 | 2.0 | 0.15 |
| 1988 | 1.2 | 0.45 |

toplankton concentration as chlorophyll-a was simultaneously reduced from 600 $\mu g\ l^{-1}$ to 200 $\mu g\ l^{-1}$, approximately in accord with observations.

In this case, it may be concluded that structurally dynamic modelling gave an acceptable result. Validation of the model, and the procedure in relation to structural changes, was positive. Of course, the approach is never better than the model applied, and the model presented here may be criticized for being too simple, and not accounting for changes in zooplankton.

For further elucidation of the importance of introducing parameter shifts, an attempt was made to run data for 1985 with parameter combinations for 1988, and vice versa. These results (Table 5.16) show that it is of great importance to apply the appropriate parameter set to given conditions. If those for 1985 are used for 1988, significantly less exergy is obtained, and the model behaves chaotically. The parameters for 1988 used under 1985 conditions give significantly less exergy.

Experience mentioned previously in this chapter shows that models can be applied to explain why biomanipulation may work under some circumstances, and not others. Qualitatively, the results can be used to explain that hysteresis exists over an intermediate range of nutrient loadings, so that biomanipulation has worked properly over this range, but not above or below it.

Another hysteresis behavior obtained with the use of structurally dynamic models for lakes have recently been published (Zhang et al., 2003a). It focuses on the structurally change between a dominance of submerged vegetation and phytoplankton in shallow lakes. The model results show that between about 100 and 250 $\mu g\ P\ l^{-1}$, both structures can exist—they show hysteresis in this range. This result is in accordance with observations from many shallow lakes.

Ecosystems are very different from physical systems, due mainly to their enormous adaptability. Thus, it is crucial to develop models that are able to account for this property, in order to derive reliable model results. The use of exergy as goal functions to cover the concept of fitness seems to offer a good possibility for developing a new generation of models, which are able to consider the adaptability of ecosystems and to describe shifts in species composition. The latter advantage is probably the most important, because a description of the dominant species in an ecosystem is often more essential than assessing the level of the focal state variables.

The structurally dynamic approach has also recently been used to calibrate eutrophication models. It is known that the different phytoplankton and zooplankton species are dominant in different periods of the year. Thus, a calibration based upon one parameter set for the entire year will not capture the succession which did take place over the year. By using exergy optimization to capture the succession (i.e., the parameter giving the best

Table 5.16. Exergy and stability by different combinations of parameters and conditions

| Year | Parameter 1985 | Conditions 1988 |
|---|---|---|
| 1985 | 75.0—Stable | 39.8 (average)—Violent fluctuations, chaos |
| 1988 | 38.7—Stable | 61.4 (average)—Only minor fluctuations |

survival for phytoplankton and zooplankton over the year), it has been possible to improve the calibration results (see Jørgensen et al., 2002).

The structurally dynamic modelling approach generally has been most widely applied in eutrophication models. A software package named Pamolare, and launched by the International Lake Environment Committee (ILEC) and the United Nations Environment Programme's International Environment Technology Centre (UNEP-IETC) also contains a structurally dynamic model, in addition to a conventional two-layer model. A test of this model has shown that it is calibrated and validated faster than the conventional model and gives better results, understood as a smaller standard deviation.

In other words, models are an appropriate tool in our efforts to understand the results of structural changes to ecosystems. In addition to the use of a goal function, it also is possible to base the structural changes on knowledge, for example, of what conditions under which specific classes of phytoplankton are dominant. This knowledge can be used to select the correct combination of parameters, as well illustrated by Reynolds (1996). That the combined application of expert knowledge and the use of exergy as a goal function will offer the best solution to the problem of making models work more in accordance with the properties of real ecosystems, cannot be ruled out. Such combinations would draw upon the widest possible knowledge at this stage.

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*Chapter 6*

# MANAGEMENT OF RESERVOIRS

Knowledge of the characteristics of reservoirs (man-made lakes, artificially constructed to address beneficial human needs) is important for the water quality manager, in order to properly apply management approaches to them. In the best case, ignoring the limnological and operational differences between reservoirs and natural lakes may cause some unexpected consequences; in the worst case, it can cause significant environmental and related problems. We know more about natural lakes than we do about reservoirs and a number of features of reservoirs do not differ significantly from lakes. Thus, this chapter stresses those characteristics that distinguish reservoirs from lakes. This is not an easy task, because the multitude of specific purposes of reservoirs and their connected features makes any generalization subject to numerous restrictions that determine whether a generalization is valid. Restrictions can be made in regard to some types of reservoirs, or to some of their geographic, hydrologic, morphometric and trophic characteristics. In each case, the particular features of the given waterbody must be taken into account, thus making the rules conditional. Thus, a theoretical basis for reservoir management is clearly needed (Tundisi et al., 1999b; Kennedy, 1999).

This chapter initially classifies reservoirs with respect to their water uses (Section 6.1.1), stressing the combinations of reservoir utilization within reservoir systems (Section 6.1.2). The purpose for which a particular reservoir was constructed determines its morphometric and other features, which is reflected in its water quality (Section 6.1.3).

Reservoir water quality and limnological typology is based on a range of geographical, geological, morphometrical, flow and trophic features (Section 6.2). Section 6.3 is devoted to discussing the effects of reservoirs on outflowing rivers and the management of reservoir outflows, primarily because the manager of a reservoir must not only pay attention to the waterbody itself, but also to its outflow. Specific methods for managing reservoirs which cannot be used to manage lakes, because of their different hydraulics, are discussed in Section 6.4.

Even though the level of reservoir construction is lower now than in the past, large reservoirs are still being constructed, particularly in developing countries. Thus, the last section of this chapter is devoted to water quality considerations during the construction of new reservoirs. Problems connected with reservoir construction, and the resulting need for environmental impacts assessments (EIAs), construction guidelines with respect to water quality, and the specific processes of newly-filled reservoirs and their aging, are discussed in Section 6.5.

## 6.1 DIFFERENCES BETWEEN RESERVOIRS AND NATURAL LAKES

In regard to their limnology and water quality, often little distinction is made between natural and man-made lakes (reservoirs). In fact, there are many similarities between these two types of waterbodies. Nevertheless, there also are differences in their limnology and water quality, as well as in the management possibilities, between the two types. Neglecting these differences may cause impacts related to not utilizing management options specific to reservoirs. Reservoirs, particularly larger, riverine reservoirs, behave differently from lakes with regard to several aspects of water quality. Thus, it is not wise to apply knowledge gained on natural lakes to reservoirs without consideration of the specific characteristics of the latter. This distinction also is important in regard to examining lakes and reservoirs separately or comparatively, and will increase the scientific insight into the properties and characteristics of both. It is stressed that this distinction holds true, particularly for larger, deeper reservoirs.

### *6.1.1 Reservoir Construction Types and Uses*

This section discusses the most typical types of reservoir constructions and uses. However, because there are many possible variations between reservoir types, these differences primarily hold true primarily for those defined technically as large reservoirs according to the definition of the International Commission on Large Dams (ICOLD, 1994, 1998); namely, those with a dam height of at least 15 meters and any volume of water, or those with water volumes exceeding 1 million $m^3$ and of any dam height. Shallow, and particularly smaller, pond-like artificial waterbodies like the thousands of ancient reservoirs of Sri-Lanka, are not much different from similarly-shaped natural lakes. An example of a transition between a natural lake and a reservoir is given by what are called *impounded lakes*. For some natural lakes, the water levels are artificially raised with the construction of a dam on the outlet tributary, in order to store more water in the lake for energy generation or irrigation.

Reservoirs can be built by constructing a dam across a river valley—these are called river *valley* or *dam reservoirs*. Dam reservoirs can be further classified into two categories, *main-stem* and *tributary*, on the basis of their location either on major rivers or just on small tributaries. Main-stem reservoirs are most often used for power generation, while water supply reservoirs are most often located on tributaries. Some ancient reservoirs were built in a period of primitive engineering capabilities and, thus, it was not possible to directly dam a river. In these cases, dams were built along the river, and the artificial depression was then filled with water from the river by means of a channel. These are the *off-river reservoirs*. They have variable uses, ranging from irrigation to fish production. Above ground reservoirs, like giant tanks, were built more recently and have similar uses. Well-documented examples of the latter are the system of reservoirs used as drinking water supplies for London (Duncan, 1990) and the Biesbosch system in The Netherlands (see Section 9.5). These are earth, stone and concrete above-ground structures to which water is

pumped from the river Thames or Meuse, respectively. In agreement with the designations described by Thornton and Rast (1996), they are called *embanked reservoirs* in this section, to distinguish them from dam reservoirs with very different water quality and management characteristics (Oskam, 1983; also see Section 9.5). The management options used more commonly for this specific type of reservoir are discussed in Section 4.3.2—*Epilimnetic Mixing* and Section 4.3.5—*Underwater Light Regime*.

The most basic physical force differentiating reservoirs from lakes, and from one another, is their theoretical water retention time, $RT$, where $RT = V/Q$. This is a theoretical property, different from the real time that an individual water parcel stays within a reservoir. Correlated with temperature stratification, some water layers may pass through a reservoir much more rapidly than others. Calculation of $RT$ is usually based on the average inflow and average volume of water in a reservoir. It may vary considerably, however, between years and shorter-time intervals, due to the variability of flow rates and reservoir volumes. The use of the inflow rates for the calculation of $RT$ is based on its use for construction considerations, particularly as the basis for the projected inflow water volumes. The use of inflows to calculate $RT$ for existing reservoirs, and shorter time intervals, is based on the assumption that the volumes of water flowing into and out of a reservoir (or being withdrawn from the reservoir and/or being evaporated) are approximately identical. During periodic imbalances between the inflow and outflow of water, the $RT$ value may be biased.

Table 6.1 lists the most common uses of reservoirs, and some reservoir features, which are to a certain degree connected with these uses. It is clear that throughflow reservoirs are not favorable as a drinking water supply. Their quality will not be improved, compared to its river inflows, which would be the case if the water retention time is long. For navigation purposes, the retention time is of no importance. On the other hand, a large accumulation capability (i.e., long $RT$) is important for flood protection.

Table 6.1. Features of reservoirs constructed for various purposes (Straškraba and Tundisi, 1999)

| Primary use | Size | Depth | Retention time | Outflow depth |
|---|---|---|---|---|
| • Flood protection and flow regulation | Small to medium | Shallow | Regionally dependent | Surface |
| • Water storage | Small to medium | — | Extremely variable | Below surface |
| • Hydroelectricity | Medium to large | Deep | Variable | Near-bottom |
| • Drinking water supply | Small | Preferably deep | High | Intermediate to deep |
| • Fish cultivation | Small | Shallow | Low | Surface |
| • Pump storage | Small to medium | Deep | Extreme variable | Near-bottom |
| • Irrigation | Small | Shallow | Long | Surface |
| • Navigation | Large | Deep | Short | Whole profile |
| • Recreation | Small | Shallow | Long | Surface |

Because of water scarcity in many regions and extensions of water usage, there is a strong tendency to add other water uses to the originally-intended purpose of a reservoir, so that multipurpose reservoir use is now common. Most recent reservoirs were constructed with a multipurpose use in mind. Nevertheless, the main use is usually the dominant consideration in the selection of a reservoir site, and correspondingly in its water quantity and morphometry characteristics.

### *6.1.2 Reservoir Systems*

In many countries, particularly those with limited water reserves in relation to water demands (e.g., Spain, Brazil, Australia), complicated reservoir systems were constructed, often with some features of unified management. Reservoir systems management is first of all dictated by water quantity concerns. Water from several reservoirs is utilized in one place and/or water from one reservoir can be directed to different destinations and uses. The most elaborate system of this type is in the southern populated area of Australia. In this area, water for irrigation and urban water supply (including the 3 million people in Adelaide) is transported over distances of several thousand kilometers, and collected by a system of reservoirs on two sides of the highest mountainous range, the Australian Alps. The water is transported between watersheds belonging to two sea regions through tunnels which act like water siphons through peaks and across valleys.

Reservoir systems can be distinguished into reservoir cascades, reservoir multisystems and pumping schemes.

*Reservoir cascades (paternoster lakes)*

A reservoir cascade consists of a series of reservoirs located on the same river. The situation in Spain provides an example, where there are more than 1000 reservoirs in a territory of 500,000 $km^2$ (Margalef et al., 1976). The reason to build such reservoir cascades is either because of hydroelectricity production, enabling the maximum use of the river's hydropower potential, or, in the case of water deficits, the desire to keep water in the country as long as possible for its continuing use and reuse. Detailed limnological and water quality studies exist for such systems in Spain (e.g., Margalef et al., 1976), Brazil (e.g., Tundisi, 1986; Tundisi et al., 1990, 1991, 1995), the United States (Kennedy et al., 1985) and the Czech Republic (Hrbácek, 1966; Hrbácek and Straškraba, 1973; Straškraba, 1990).

From a water quality perspective, reservoir cascades are favorable. The reason is the retention of phosphorus, organic matter and other polluting elements and components in a given reservoir, so that the next reservoir in the cascade receives less pollution than the one upstream of it, and so on. However, this successional pollution decrease along a cascade will not occur, in spite of the large retention capacity of each reservoir, if the pollutant load from secondary tributaries and shoreline sources exceeds the water retention in the reservoirs. This is happening, for example, in the Barra Bonita Reservoir of the reservoir system on the Tiete River in Brazil, São Paulo State (Padisak et al., 2001; Barbosa et al., 1999—also see Section 9.8).

*Reservoir multisystems*
A reservoir multisystem comprises several reservoirs located in different watersheds (or sub-watersheds) that are operated jointly for water quantity and/or water quality purposes. Because water quantity is usually the primary concern, recent introduced attempts to also manage water quality are fairly difficult. The reason is that the chemical composition and water quality characteristics of the different reservoirs within the system may be markedly different, and the transfer of water between systems also can produce unwanted characteristics and changes within the reservoirs.

*Pumping schemes*
These reservoirs are used to augment hydroelectricity production during periods of peak needs. The water is either pumped into a reservoir built specifically for this purpose, into an upper-lying lake or into a cascade consisting of a large reservoir upstream and a smaller one immediately downstream. During periods of lower electricity demand, the excess energy is used for pumping the water upstream. During periods of high energy demand, the water is released through turbines to produce hydropower. A daily cycle of such operations is possible. On the other hand, using these reservoirs for additional purposes (e.g., drinking water supply), along with power production, is complicated because unwanted water quality deterioration can take place in the upper storage reservoir.

Pumping schemes can be divided into those constructed as a secondary storage in combination with conventional hydroelectric generators, and those which reversibly send water up and down the same turbines.

The effects of water pumping on fish and other organisms have been discussed (e.g., see Schindler et al., 1995). Several studies have demonstrated that there is only a small negative impact of pumping on fish and fisheries (Robins and Mathur, 1976).

### 6.1.3 Water Quality Differences between Natural Lakes and Reservoirs

With respect to water quality, reservoirs, particularly deep, stratified ones, differ from natural lakes in a number of limnological characteristics (Wetzel, 1983, 1990; Kennedy et al., 1985; Tundisi, 1986; Søballe et al., 1992; Thornton et al., 1996; Straškraba et al., 1993; Straškraba, 1999). Although shallow and small reservoirs are much more similar to natural lakes, some differences also have been recognized and summarized by Padisak et al. (1999). This section primarily discusses the characteristics of deep reservoirs, which have considerable consequences for water quality management. The essential differences are summarized in Figure 6.1.

*Pollution of reservoirs.* Pollutant loads to reservoirs are usually greater than for lakes located in drainage basins with similar land uses. This is due primarily to the fact that the drainage basins of reservoirs are generally larger than those of lakes. In a sample of lakes and reservoirs in the United States, the drainage basin to waterbody area ratio for reservoirs was on average 14 times higher than for lakes. This figure may be somewhat biased, however, since there are more reservoirs and fewer lakes in the southern, drier regions of

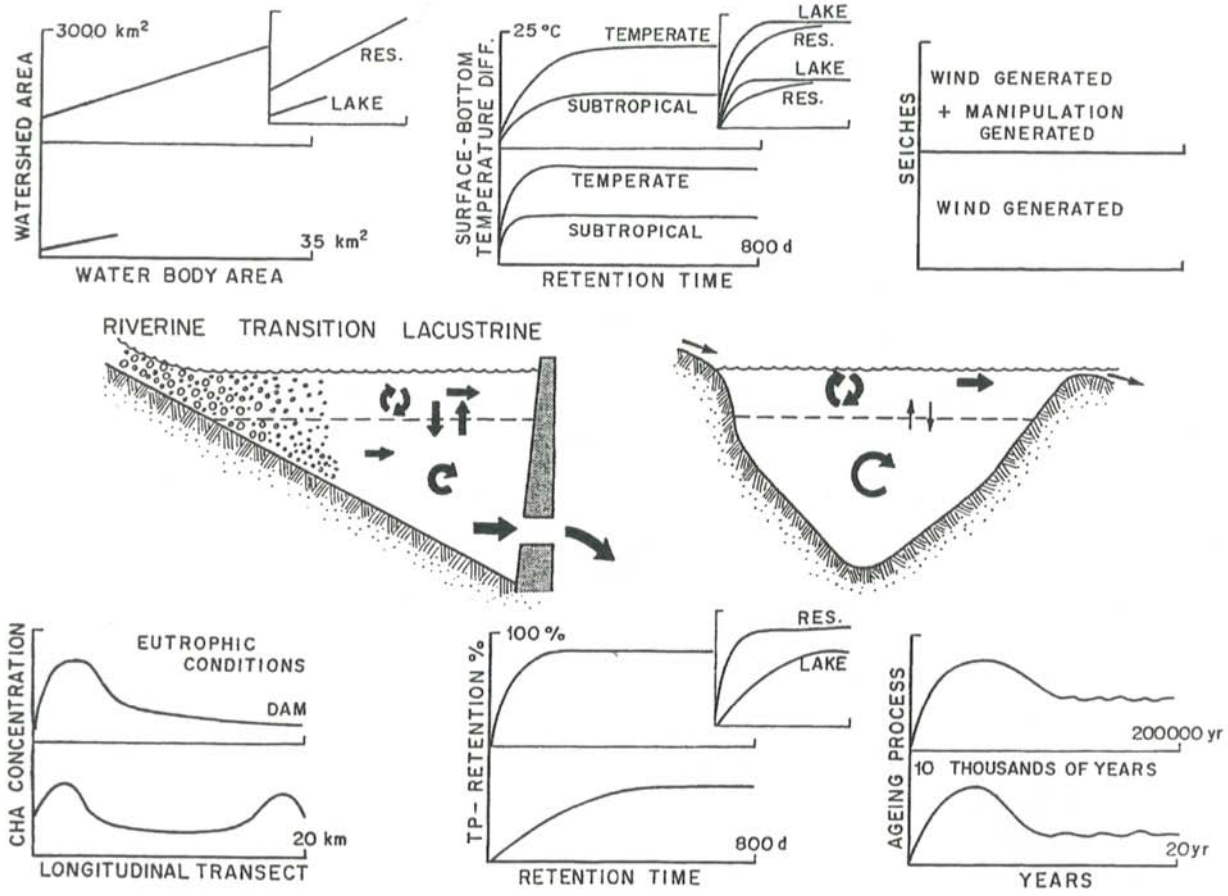

3000 km²
RES.
LAKE
WATERSHED AREA
35 km²
WATER BODY AREA
25 °C
TEMPERATE
SUBTROPICAL
LAKE
RES.
LAKE
RES.
SURFACE-BOTTOM TEMPERATURE DIFF.
TEMPERATE
SUBTROPICAL
800 d
RETENTION TIME
SEICHES
WIND GENERATED
+ MANIPULATION GENERATED
WIND GENERATED
RIVERINE
TRANSITION
LACUSTRINE
EUTROPHIC CONDITIONS
DAM
CHA CONCENTRATION
20 km
LONGITUDINAL TRANSECT
100 %
RES.
LAKE
TP- RETENTION %
800 d
RETENTION TIME
200000 yr
10 THOUSANDS OF YEARS
20yr
AGEING PROCESS
YEARS

Fig. 6.1.

the United States, where larger watersheds are needed to provide water for dry periods. Nevertheless, the difference is still large. As diffuse (nonpoint) pollution enters waterbodies from each type of land use per unit land surface area, and point source pollution based on the number of inhabitants and mass-cultivated animals, and the quantities of various products made in factories, the drainage basin of a reservoir with similar land use and population densities generates more pollution because of its larger area. Thus, it would be expected that the eutrophication of a reservoir, at least of its inflowing part, should generally be higher than that of a lake. Chemical stratification also would be more pronounced in a reservoir, because the increasing sedimentation of organic matter entering it from the drainage basin or produced by phytoplankton within the reservoir due to its higher nutrient load, will be higher than for a lake. However, much depends on the land use, the pollution concentrations in the lake inflows, and the inflow rates. The consequence of increased organic matter sedimentation is increased decomposition and utilization of oxygen by microbial decomposers in the waterbody, leading to low hypolimnetic oxygen concentrations, or even the development of anoxic conditions in the waterbody. When the bottom water layer is deprived of oxygen, water treatment is typically more costly because of higher concentrations of manganese and iron, the increased occurrence of odors and smells, and the presence of hydrogen sulfide and methane. Further, phosphorus accumulated in the sediments is released at an elevated rate, causing increased algal productivity.

*Stratification and mixing of reservoirs.* In the same manner as for lakes, reservoir stratification is determined primarily by the reservoir's geographic location and morphometry (compare Section 1.3.2). For deep valley reservoirs, a characteristic feature is the strong dependence of limnological features like stratification and in-lake phosphorus retention on the theoretical water retention time, *RT* (Straškraba, 1998a, 1998b, 1999). Lakes also are affected by water flow rates; because of the generally higher inflows to reservoirs relative to their volumes, there are many more reservoirs than lakes with short water retention times, with their stratification being strongly affected by flows. Consequently, stratification conditions are much more variable in reservoirs than in lakes. Because of inter-annual variability in weather conditions, the same reservoir may only be weakly-stratified in a

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Fig. 6.1. Schematic representation of the essential limnological differences between deep reservoirs and deep lakes. Reservoirs have uniformly deepening longitudinal transects with eccentric maximum depth and intensive longitudinal flows, while lakes have more central position of the maximum depth. Reservoirs also have larger watershed/waterbody areas and consequently higher throughflow (= shorter water retention times) than lakes. The degree of temperature stratification depends on geographical location and theoretical water retention time, as do the lakes. Due to the saturation character of relationship between stratification and *RT* shown in Figure 6.2, however, many lakes (with longer water retention times) are in the insensitive saturated region of the dependence, while reservoirs are in the sensitive region. Lakes usually have lower areal phosphorus loads and retain less phosphorus than reservoirs of similar retention times. Longitudinal transects of biologically-active variables (e.g., phosphorus, chlorophyll-a) are characterized by a peak near the inflow, while lakes have higher values along the shores (from Straškraba, 1998b).

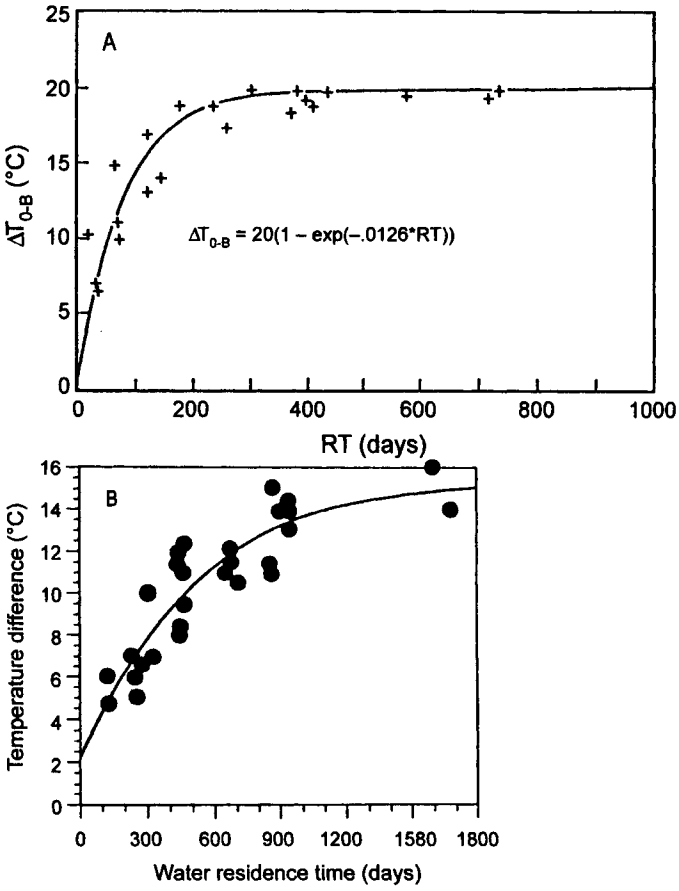



Fig. 6.2. Dependence of the degree of reservoir temperature stratification on theoretical water retention time (*RT*, days). The degree of stratification is expressed by a simple measure, temperature difference in the period of maximum temperatures between the surface and the hypolimnion ($\Delta T_{0\text{-}B}$, °C). A—The relationship developed by Straškraba and Mauesberger (1988) for reservoirs in the Czech Republic and the Bavarian region in Germany; B—The same relationship for Texas Reservoirs by Groeger and Tietjen (1998).

wet year when the flows are high, and exhibit well-developed stratification in another year with drier conditions. This high sensitivity to weather conditions is another feature making reservoir management more difficult than for lakes. Thermal stratification affects a number of chemical and biological processes, and subsequently the water quality for such reservoirs. The relationship between the degree of stratification of deep reservoirs and the *RT* for the period of stratification is asymptotic. Reservoirs with an average *RT* of a few days are not stratified. The degree of stratification strongly increases with the prolongation of *RT*, slowly becoming independent of the retention time. Thus, there is no sharp boundary in

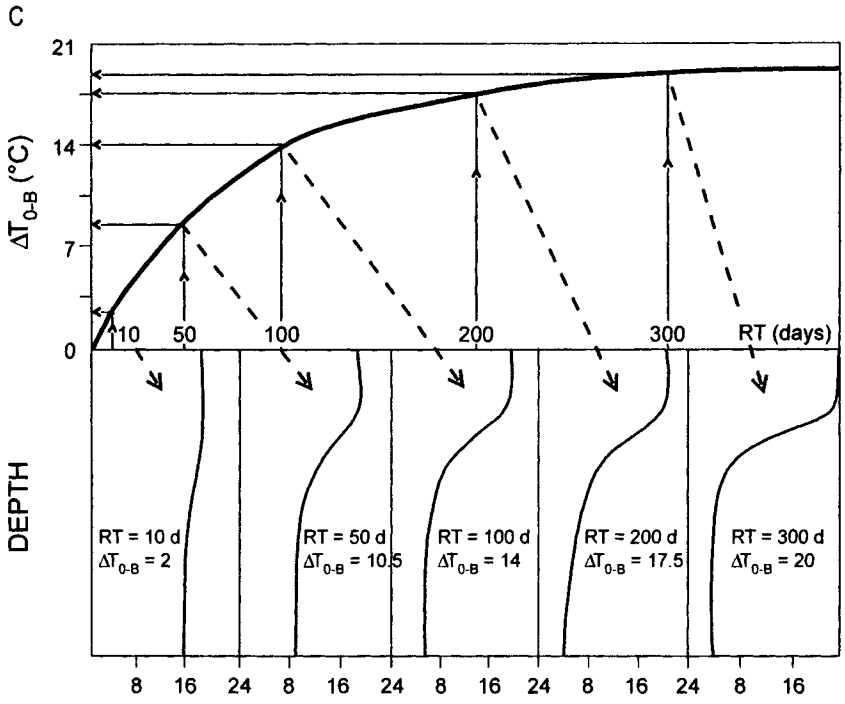



Fig. 6.2 (*continued*). C—The profiles at different $\Delta T_{0\text{-}B}$ (°C) due to different retention times (*RT*) are shown.

a reservoir's *RT* above which no influence on stratification can be observed. Nevertheless, we can arbitrarily set the critical retention time, above which the stratification is only weakly affected by *RT*, at 200 days. Above this value, there is practically no effect at all, at $RT \approx 1$ year (see Fig. 6.2). Stratification of a reservoir with long-term average retention times highly exceeding the critical level of 200 days can still be conditioned by flow during extremely low water levels and high flows. On the other hand, deep, unstratified reservoirs with a retention time of only a few days may still become stratified during drought conditions, or during periods when the water throughflow is significantly reduced for flow operation. Because of variable water flows into and out of reservoirs, their temperature and chemical depth profiles are less smooth than for lakes.

A specific feature of some reservoirs is their *hydraulic stratification* (Tundisi, 1984), which is related to the separation of the deepest water layers within the waterbody, thereby not being part of the reservoir outflows and becoming more stagnant as a result. The separation of the water layers may be caused by the lower edge of the outlets, or by submerged structures such as walls build for protection of the construction site during the building phase. The influence of the outlet location is apparent in power generation reservoirs with large water throughflows (Fig. 6.3). Further, submerged weirs also can cause hydraulic stratification when the lower edge of the outlets is also deep, as is sometimes the case

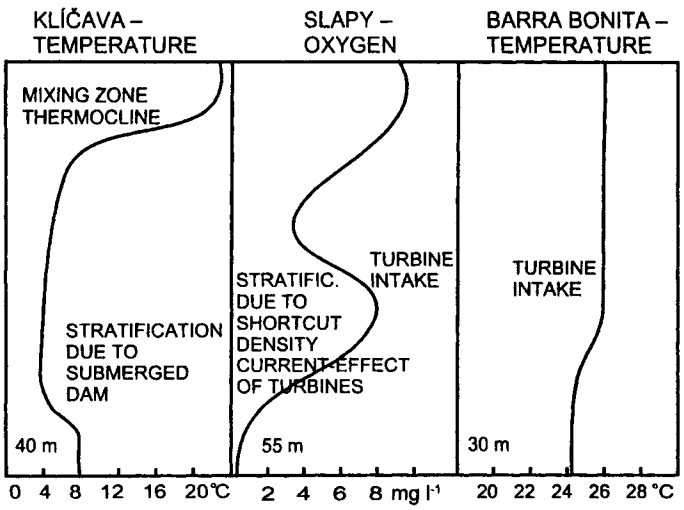



Fig. 6.3. Examples of hydraulic stratification caused by submerged weirs and location of the turbine intakes for different reservoirs.

for drinking water reservoirs. Thus, the submerged dams have to be destroyed, or at least opened sufficiently, to prevention the formation of a stagnant bottom layer of reduced water quality.

Another decisive variable for stratification, and a cause of differences between lakes and reservoirs, is the location of the water outlets. Water flows from a lake primarily from its surface water layers, whereas the outflow from reservoirs can be variable. Most reservoirs have deep outlets related to their intended beneficial use. For some reservoirs, surface outlets are constructed. During floods, the excess water can be discharged by overspills or by bottom outlets. The deep release of bottom water layers causes more admixing of surface water to deeper strata and, hence, decreased stratification.

Mixing within a reservoir depends on stratification, instantaneous water flow rates, and the shape of the reservoir. The water mixing is generally intensive, and extends over the whole depth when stratification is weak. When the temperature is high and, therefore, a density difference exists between the mixed layer and the hypolimnion, the water mixing is dominantly restricted to the upper water layer.

*Longitudinal differentiation.* Another typical feature of deep valley reservoirs is a longitudinal differentiation of water quality (Fig. 6.4). The main causes are changes in water flow rates and mixing along the length of the reservoir, with effects on associated physical and chemical processes. Although water moves on average down the length of a reservoir, considerable longitudinal and local biological and chemical inhomogeneities can exist (Groeger and Kimmel, 1984; Hejzlar and Straškraba, 1989; Ryding and Rast, 1989; Ground and Groeger, 1993). Locally, and at certain depths, water in the reservoir also can

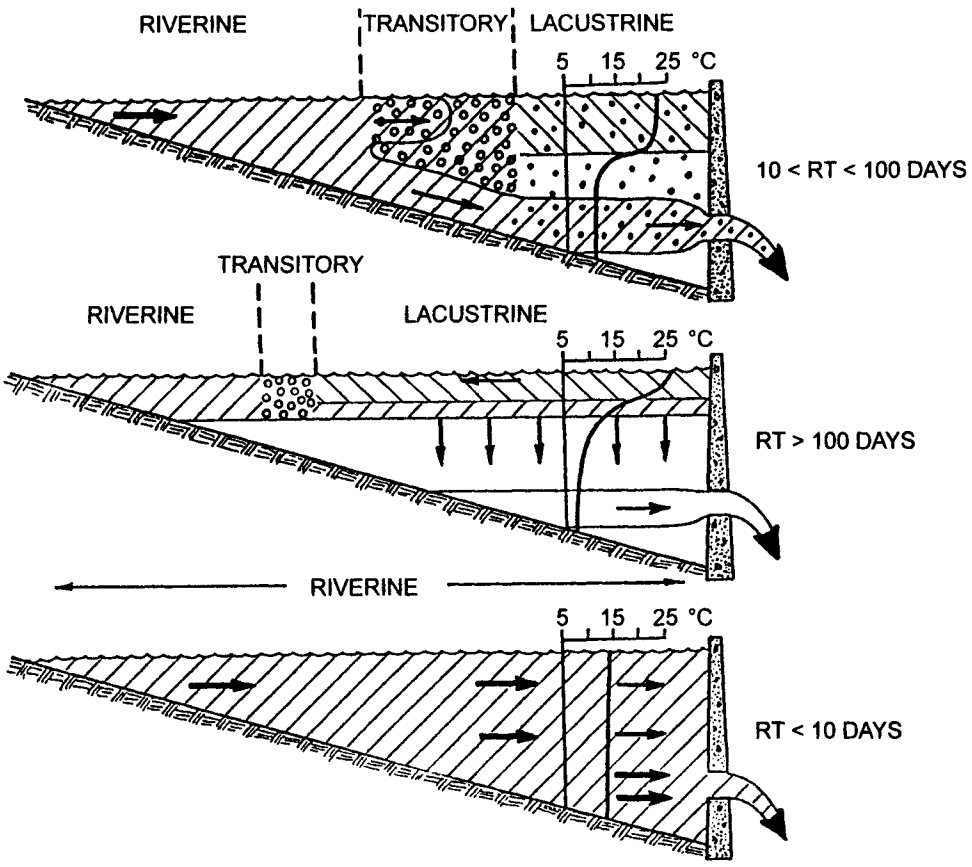



Fig. 6.4. Longitudinal differentiation of reservoirs with different theoretical water retention times. Different situations can be found not only in different reservoirs, but also in the same reservoir during different water levels and/or inflow rates. The typical three zones are the riverine inflow, the transitory zone and the lacustrine zone at the dam. The extent of the individual zones varies from domination of the riverine to domination of the lacustrine zone.

move upstream. Four major longitudinal regions of a deep valley reservoir usually can be distinguished, as follows:

- The inflow region,
- The transition region,
- The "lake-like" region,
- Coves (bays).

Such clear longitudinal differentiation, however, is only valid if a reservoir has one dominant inflow. Reservoirs receiving water from many smaller inflows may not possess such a characteristic, an example being the Tomhannock Reservoir (Melcher et al., 1997).

*The inflow region* is usually shallow and narrow, characterized by rapid, unidirectional water flow, with maxima near the center and close to the surface. The average length of this region is greater in reservoirs (or situations) with short average *RT*. The length of the region at a particular time depends on the actual flow rates and water level in the reservoir, as well as its degree of stratification. It is longer when flows are higher, water levels lower, and stratification less strongly developed.

*The transition region* is where the flowing river water enters the more stagnant reservoir water. It is characterized by major, and often sharp, differences in flow rates and stratification within the transition region. It is this region in which the river water, if colder than the reservoir water, plunges down to a depth of corresponding density. In its passage to a lower depth, its temperature slightly increases by mixing with the surrounding water masses. Depending on the density stratification within the reservoir, the plunged flow can continue as a bottom flow or an interflow. The sharp boundary between the two water masses also may be marked by differences in transparency: the river water may be turbid, with the reservoir water more transparent. If the density difference between the river and reservoir water is very small, or the inflow is warmer, the inflow will predominantly enter the surface layers of the reservoir.

Maximum concentrations of suspended sediments, nutrients and algae, as well as maximum sedimentation, occur in the transition zone. The sedimentation retention of reservoirs is very high, often more than 90% (Petts, 1984), and retention levels even up to 94% being observed. This characteristic limits the operational life time of reservoirs, with some of the shortest operational life times being observed for some Chinese reservoirs. As an example, the Heisong Reservoir lost 20% of its volume in 3 years, and the Sanmanxia Reservoirs 40% in 20 years. Classical estimates of the sediment filling of reservoirs were recently found in South America to be underestimates. The observed rates give up to 60 times the predicted value, with the major reason being human disturbance of soil by improper land practices (e.g., inadequate irrigation, line plantation, fires, increased flows).

The *lake zone* is more balanced and most stagnant, with wind effects also being stronger. Vertical stratification is most pronounced. *Coves* are typical where side streams enter a river. They may exhibit longitudinal differentiation similar to the main body of a reservoir, with modifications occurring because of density conditions and inflow composition. When water levels rise, water from the main body can enter the bays, being layered according to its instantaneous density situation and, simultaneously to a certain degree, to the mixing of inflowing and bay water.

*Water level fluctuations.* Water level fluctuations in many reservoirs are higher than for natural lakes. The extent of variability is related to the natural inflow variability and the purpose and operation of a reservoir. The highest variability is observed in dry regions with a high periodicity of river inflows. The water volume in summers of dry years may drop in such reservoirs to a small fraction of the full water storage volume. This property is typically known prior to the preparatory phase of reservoir construction. For each reservoir, reliable probabilistic estimates of long term changes in flow rates, utilization and release rates and corresponding water level fluctuations are made available. For deep reservoirs, high fluctuations can cause phytoplankton to become the dominant internal source of

organic matter production, which also corresponds to the conditions for reduced development of macrophytes. Water level fluctuations and associated development of macrophytes can have major impacts on the development of fish populations in reservoirs (Ploskey, 1983, 1986; Zalewski and Dobrowolski, 1994).

*Phosphorus retention.* From a water quality perspective, deep reservoirs have one positive feature compared to lakes; namely, they have a very high phosphorus retention capacity, $R_P$. They have a high phosphorus retention even with an average annual theoretical water retention time of about 10 days, with the $R_P$ increasing to the theoretical water retention time *RT* of about 200–300 days (Schreiber and Rausch, 1979; Fiala and Vašata, 1982). Both the particulate and dissolved (both organic and inorganic) phosphorus is retained. Figure 6.5A illustrates the situation in Germany (based on data by Wilhelmus et al., 1978). When the *RT* was greater than 30 days, 70–90% of the reactive phosphorus was retained in the reservoir, whereas when the *RT* was less than 30 days, the total phosphorus retention dropped to less than 10%. For the shallow Lake Talquin, Turner et al. (1983) observed a decreased retention of both total phosphorus and silica during years with a low *RT*. A decrease in the *RT* from 44 to 22 days produced a drop of total phosphorus retention from 65% to only 25%.

Recent re-examination of the phosphorus retention capacity of reservoirs by Straškraba et al. (1995), Kennedy (1999) and Straškraba (1999) have shown that maximum retention capacity for both temperate and tropical reservoirs is on average around 80%, but can increase to above 90% in some situations. The phosphorus retention also depends highly on the areal phosphorus load, being maximum for reservoirs with very high phosphorus loads of more than about 15 g $m^{-2}$ yr. The high retention capacity is reached at relatively short retention times (Fig. 6.5). The phosphorus load of the river flowing out of a reservoir will be up to one-tenth of the load of the load in the inflowing river, unless there are also major phosphorus sources along the reservoir shoreline. The decreased phosphorus levels are considered a positive improvement of the trophic degree of the outflowing river. However, there also are instances in which the decrease was considered negative, due to its assumed effects on the decreased fish populations in the lower-lying lake (Kootenay Lake, British Columbia—according to Friedl and Wüest, in print).

In addition to phosphorus, reservoirs also can retain a fairly large quantity of organic matter and other components. Nitrogen is retained to a considerably smaller degree, much more in shallow than in deep reservoirs, and significantly only in tropical and subtropical regions. However, our knowledge of the optimal conditions for intensive nitrogen retention in different reservoirs is still limited. Silica also is retained in reservoirs, which is important for reservoirs dominated by diatoms. Armengol et al. (1999) have observed an exponential decline of many variables in Sau Reservoir in Spain, expressed as the rate of decrease of individual components along the length of this polluted reservoir in the following order: ammonia, soluble reactive phosphorus, turbidity, total phosphorus, total nitrogen, silica, alkalinity, chlorides, conductivity.

*Degradation of organic substances by reservoirs.* The retention of water in a reservoir causes a higher degradation of easily-decomposable organic substances, expressed as bio-

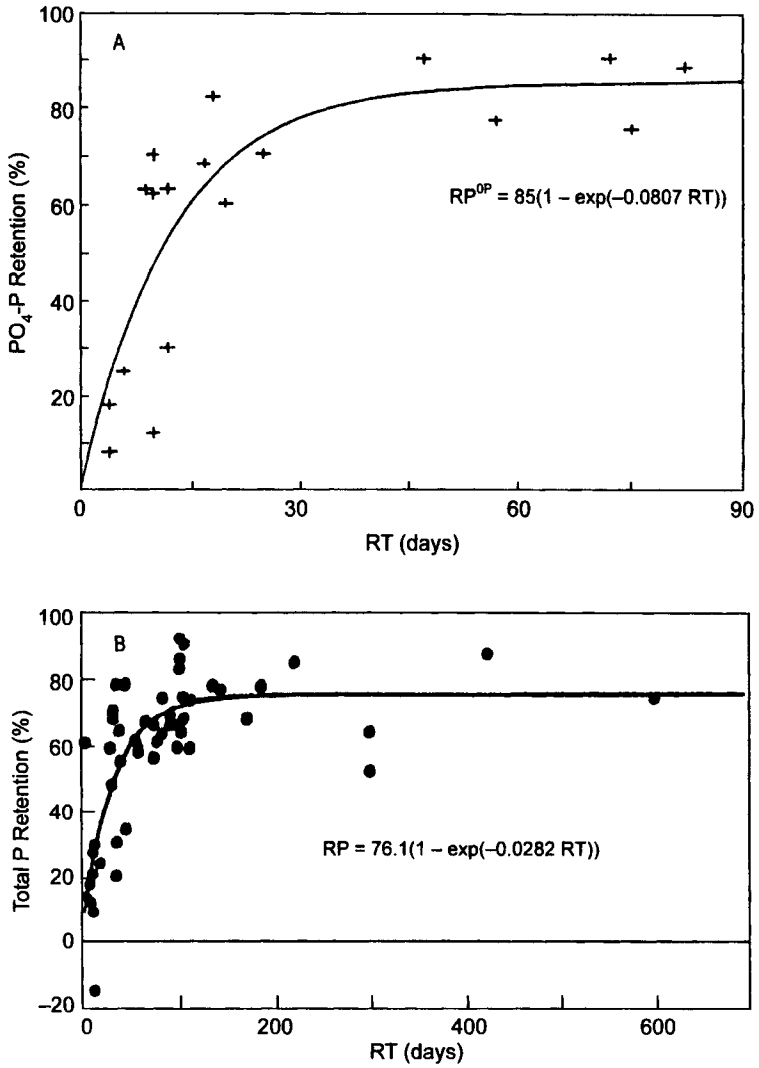



Fig. 6.5. Phosphorus retention by deep reservoirs. A—Phosphate phosphorus retention of German reservoirs; from data by Wilhelmus et al. (1978); B—Total phosphorus retention in the reservoirs of the Northern Hemisphere temperate region and dependence on the theoretical water retention time, *RT* (days). The reservoirs have mostly high areal loads.

chemical oxygen demand, BOD (self-purification), than in a comparable stretch of the original river. Quantitative relations of the water quality improvement from this reservoir characteristic in this direction were estimated by Straškrabová et al. (1993).

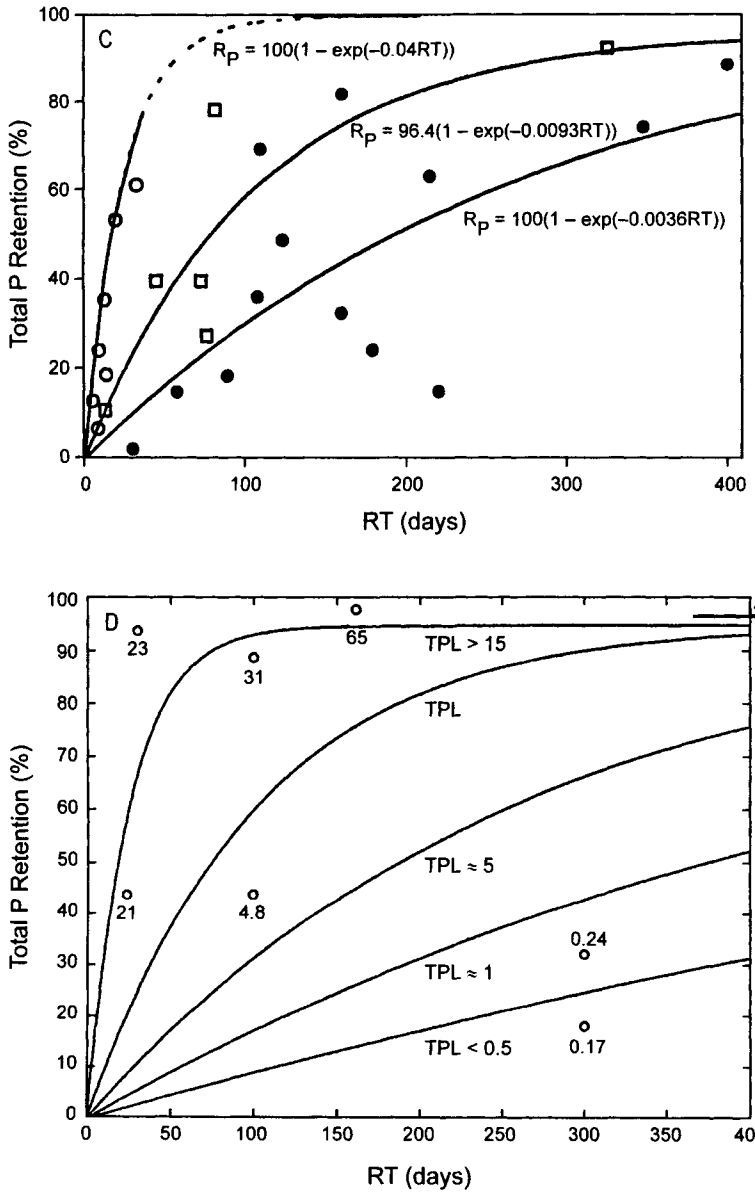



Fig. 6.5 (*continued*). C—Idealization of the phosphorus retention and dependence on *RT* and areal loads, based on data for reservoirs in the United States. The three curves are, from the highest, for areal load of 15, 10 and 5 g $m^{-2}$ yr, respectively (from Kennedy, 1999); D—Idealization for a range of areal loads and the position of data points for a few tropical Brazilian reservoirs. The tropical data suggest that, with respect to phosphorus retention, the tropical reservoirs behave in the same way as temperate ones (Straškraba, 1999).

Table 6.2. The effect of increasing water retention time on deep, stratified valley reservoirs

| Variable | Reaction |
|---|---|
| Loading | Decreases |
| Stratification | Increases, then levels out |
| Surface temperature | Increases, then levels out |
| Bottom temperature | Decreases |
| Longitudinal differentiation | Increases, then decreases |
| Amount of sediments | Decreases |
| Nutrient retention | Increases, then levels out |
| Phytoplankton biomass | Increases, then decreases |
| Bottom fauna | Decreases |
| Trophic status | Increases, then decreases |
| Cyanobacteria | Increases |
| Hypolimnetic anoxia | Increases, then decreases |
| Zooplankton | Increases, then decreases |
| Fish biomass | Increases |
| Period of aging | Increases |

The effects of water retention time on deep valley reservoirs, some of which were treated in more detail above, are summarized in Table 6.2. All the changes, except the decrease in load and the associated decrease in phytoplankton, are only valid below a critical *RT* of about one year. The phosphorus load decreases after this point, although at a slower rate, thereby also decreasing the phytoplankton levels in the reservoir.

## 6.2 DIFFERENT TYPES OF RESERVOIR WATER QUALITY

Management of reservoir water quality must take into account the specific characteristics of each waterbody, including its geographic location, water retention time, morphometry, geology and trophic and pollution status. This section points out management requirements and possibilities for reservoirs, based on these specifics. A classification of reservoirs is made, with the goal of facilitating appropriate management efforts. It must be kept in mind that, with respect to the all the variables discussed herein, there is a continuum of reservoirs, rather than a clear grouping of reservoirs into specific categories. The categorization is more or less arbitrary and transitions are common. Further, the same reservoir may belong to different categories in different years, depending on its condition.

### *6.2.1 Geographic Differences in Reservoir Water Quality*

The geographic background for differences in reservoir water quality in different regions was described in Section 1.3.2. This section focuses on management possibilities and limitations for reservoirs in particular latitudinal and altitudinal positions. Most reservoir management methods have been developed in temperate regions. Their success in other

conditions, however, is not guaranteed, and it is necessary that we gain experience with their application in other regions.

*Humid tropics*

Reservoir construction is underway in many countries, and a pre-construction evaluation of the positive and negative environmental consequences with the use of environmental impact assessments (EIA, see Section 6.5.2) is very important for making correct management decisions. The water quality of tropical reservoirs (both in humid and more arid regions) is often much worse than that of temperate regions, with eutrophication far from being the only problem. Rapid economic growth can cause reservoir pollution from organic chemicals, eutrophication and toxicity to overlap, with their contribution to the total pollution being difficult to determine. Waterbodies in tropical regions are particularly prone to low oxygen conditions, with anoxic hypolimnia present in tropical regions at much lower nutrient loads than seen in temperate regions. One reason is the higher decomposition rates of organic matter in the elevated temperatures in the bottom waters. Another is the higher load of organic matter. The latter is related to the higher abiotic turbidity of many tropical rivers which, moreover, is composed of fine particles with high organic content. The character of tropical rains causes higher erosion rates for comparable precipitation levels. The consequence is that artificial mixing and oxygenation are often a more necessary and suitable management alternative, in spite of temperature differences of a degree or even less within the thermocline region. Such small temperature differences can be accompanied in tropical regions with layers of poor water quality (deoxygenation, hydrogen sulfide, methane, high ammonia, highly increased phosphorus concentrations, high quantities of iron and manganese, etc.). Thus, more elaborate mixing techniques (e.g., hypolimnetic mixing, oxygenation) are preferable.

Ecotechnological methods have a greater potential for reservoirs in tropical regions, not only because of the fewer resources available for water quality management and much lower labor costs, but also because of more active biological processes throughout the year. Suitable ecotechnological methods for reservoirs include the use of wetlands (Section 4.2.4), river shore vegetation preservation and management (Section 4.2.1), and pre-reservoirs and hydraulic regulation in reservoirs (Section 6.4.2). The application of biomanipulation (Section 4.3.1), very successful for temperate region, is still only in an experimental phase in the tropical regions, being more complicated because of more complex and local composition of fish fauna and the presence of omnivores.

One method for improving water quality in tropical reservoirs is the application of copper sulphate to kill phytoplankton blooms. Algae grow rapidly under the higher temperatures in the tropical regions, with a consequence being that the applications of copper sulphate have to be repeated several times during the year. The danger of an accumulation of toxic copper concentrations in the sediments of copper-treated waterbodies also increases significantly. Further, the rapid decay of the killed algal mass creates short-time, intensive difficulties by causing low oxygen concentrations and accompanying negative water quality changes. Finally, algae can adapt to increased copper levels, becoming progressively more insensitive to treatment. A more suitable method for preventing algal

blooms is phosphorus inactivation with in-lake alum or iron treatment. This method has not yet been widely used in tropical regions, however, so that verification of the rules for its successful application in temperate regions (Section 4.3.5) is not yet available.

*Semi-arid and dry regions*

Due to the character of soils and their weathering, and the character of water flow variability, a water quality problem more often encountered in reservoirs in semi-arid regions is turbidity (see Section 2.2.2—*Low Water Volume*).

The scarcity of natural lakes in semi-arid and dry regions, with evaporation exceeding precipitation, is addressed with the construction of numerous reservoirs. A negative aspect of this situation, however, is that water evaporation is increased by the formation of large areas of open water. The reservoirs in these regions are constructed so as to have large drainage basins and capacities to provide water reserves for dry periods. Large water level fluctuations create water quality difficulties, due to water quality deterioration during periods of decreased water levels. The combination of hot weather and concentrated nutrient levels provokes development of Cyanophycean water blooms with the production of toxins (Armengol et al., 1999—Spain; Naselli-Flores, 1999—Sicily).

*Temperate regions*

Construction of new reservoirs is becoming rare in temperate regions. Thus, problems of reservoir aging are not pressing. Wherever point source pollution is highly reduced, the dominant reservoir water quality problem is diffuse (nonpoint source) pollution. The extent of diffuse pollution can be very high, and also can be from such sources as air pollution, which is difficult to control. The overall environmental pollution is still high, and identification of its sources is difficult in many cases. Innovative, combined and integrated methods are sought and becoming successful (e.g., see the example of the Grosse Dhünn Reservoir). Wetland restoration and constructed wetlands (Section 4.2.4), pre-reservoirs, both with fixed construction or with the use of plastic curtains (Section 6.4.2—*Phosphorus Retention*), selective withdrawal, both with fixed structures and curtains (Section 6.4.2—*Selective Withdrawal*) and biomanipulation (Section 4.3.1) belong to the category of ecotechnological methods with least cost for application to the global environment. Properly-managed reservoir cascades also are capable of significantly improving water quality.

*Subarctic and high mountain regions*

Reservoirs are not very common in this region, although some large reservoir complexes exist in the subarctic region of northern Canada. There also are high-mountain reservoirs in the Alps and other mountain ranges. Because of the slow decomposition of organic matter and the low temperatures characteristic of these waterbodies, their oxygen levels are high. Some water quality problems may be related to elevated humic matter concentrations in tundra and forest regions. Further, faecal pollution can persist for prolonged periods in such waterbodies, even in pristine cold water.

### 6.2.2 Morphometric Differences

The differentiation of reservoirs into shallow and deep categories is based objectively on the effects of wind fetch (Section 3.2.1).

*Hydrologically-shallow reservoirs.* Reservoirs with maximum depths that do not exceed the water mixing depth are hydrologically shallow. Hydrologically-shallow reservoirs are different from deep, stratified reservoirs, being much more similar to hydrologically-shallow lakes than to deep reservoirs. Longitudinal differences are less pronounced, with wind-conditioned circulation and flows dominating if the theoretical water retention time, *RT*, exceeds 200 days. However, these differences are important for all shallow reservoirs. Macrophytic nearshore vegetation also is important, affected mainly by water level fluctuations. The same water level fluctuations in shallow reservoirs cause large areas of bottom sediment to be exposed. The same areal nutrient load results in higher trophic levels, the primary production being realized dominantly either as phytoplankton or macrophytes. Switching between phytoplankton and macrophyte stages depends on the water transparency and fish populations (see Section 3.2.1).

This categorization of *lake sizes* can only be arbitrary. The only objective differentiation might be waterbodies small enough that the geostrophic Coriolis force does not play a role, whereas it is important in large waterbodies. However, such delineation is difficult. Thus, a subjective categorization developed by Straškraba and Tundisi (1999) is used here (Table 6.3).

Small reservoirs are not favorable as a water supply because they are sensitive to local pollution. Successful biomanipulation is more easily achieved in small, shallow waterbodies, the main reason being that fish populations are more easily manipulated and maintained at certain levels in these types of waterbodies.

Similar to natural lakes, important water quality differences and management possibilities exist for hydrologically-shallow (unstratified) and hydrologically-deep (stratified) reservoirs (Section 3.2.1). Drinking water of good quality is more easily obtained from stratified reservoirs.

### 6.2.3 Classification of Deep Reservoirs on the Basis of Water Retention Time

The importance of the theoretical water retention time, *RT*, was delineated in Section 6.1.3. In accordance with the outlined relations, the categories of reservoirs given in Table 6.4 can be distinguished. It is stressed that the reservoirs of the first two categories are so different in their water quality conditions that the use of standard empirical relations for estimating their water quality conditions cannot be used, or have to be highly modified.

Reservoirs of the first type are often called *weirs*, and are used for navigation purposes and to enable water intake for industry. *Re-regulation reservoirs* used for nivelizing the river flow below energetic peaking operation reservoirs also belong to this type. The second group represents *main stem* reservoirs, and the last group *tributary* reservoirs (e.g., Søballe et al., 1992). It is to be remembered that, due to high flow variability, the same reservoir may belong to a different type under extreme flow or drought conditions.

Table 6.3. Size categories of lakes and reservoirs (Straškraba and Tundisi, 1999)

| Category | Area ($km^2$) | Volume ($m^3$) |
|---|---|---|
| Large | $10^4$–$10^6$ | $10^{10}$–$10^{11}$ |
| Medium | $10^2$–$10^4$ | $10^8$–$10^{10}$ |
| Small | 1–$10^2$ | $10^6$–$10^8$ |
| Very small | < 1 | < $10^6$ |

Table 6.4. Categories of deep reservoirs, based on their long-term, average theoretical water retention time (*RT*)

| Category | Retention time | Stratification |
|---|---|---|
| With very short retention time | *RT* < 14 days | Nonstratified |
| With short retention time | 15 < *RT* < 1 yr | Strongly increasing with *RT* |
| With long retention time | *RT* > 1 yr | Fully developed, lake type, geographically conditioned |

### *6.2.4 Classification into Hard-Water and Soft-Water Reservoirs*

The distinction of hard-water and soft-water lakes and their management consequences are the same for reservoirs as for lakes (Section 3.2.2). Treatment of reservoir water containing humic materials for drinking water is difficult because bad tastes can be produced from the organic matter contained in them.

### *6.2.5 Oligotrophic and Eutrophic Reservoirs*

Differences among reservoirs of different trophic states are the same as for natural lakes (see Section 2.2.2—*Eutrophication*, and Section 3.3.3).

Water throughflow also plays a significant role in reservoirs. When the water flow is very large (i.e., a short *RT*), full development of phytoplankton does not occur. Lakes and reservoirs with very short detention times also have lower phytoplankton biomass, as the biomass is continuously released from the reservoir in the outflowing water. The water quality of the reservoir is largely determined by the inflow, with the reservoir itself playing a dominant role in defining the quality of the outflowing water. Basic differences between the inflows, in-lake conditions and outflows of an oligotrophic and a eutrophic reservoir are summarized in Figure 6.6, along with corresponding management efforts necessary in the drainage basin, in the lake and at the outflow.

### *6.2.6 Degree of Pollution*

The types of reservoir water pollution do not differ from that of lakes, as outlined in Section 2.2.2. The degree of pollution in reservoirs is often greater because of two reasons:

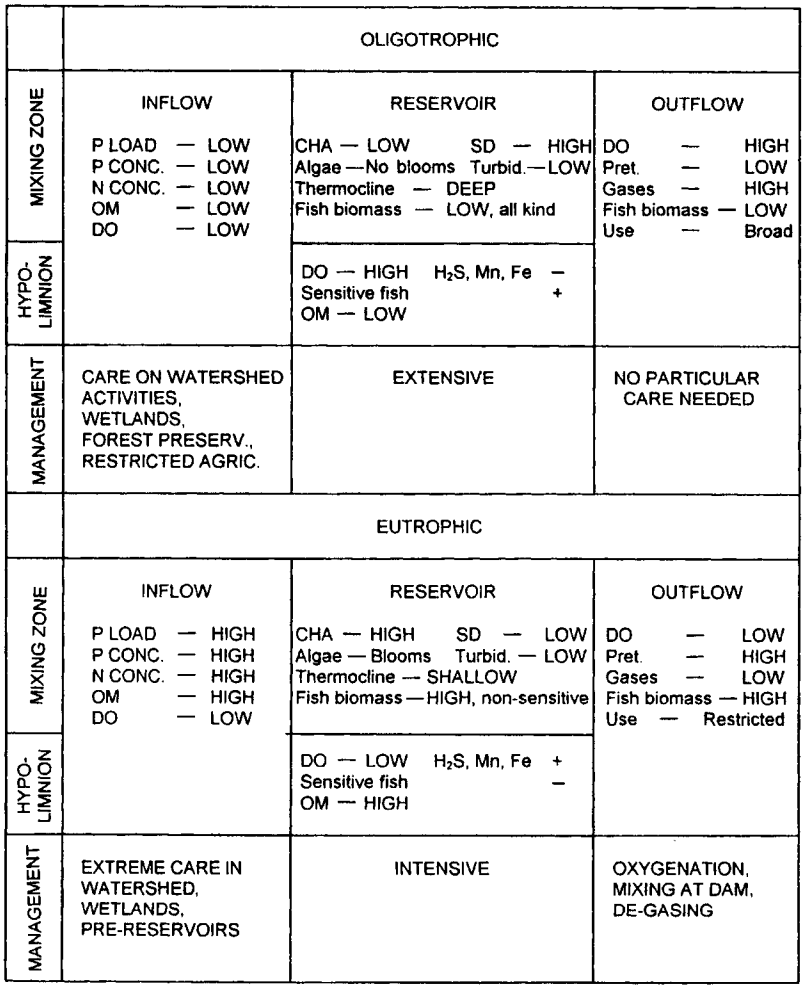



Fig. 6.6. Schematic water quality differences between a deep oligotrophic and deep eutrophic reservoir.

- For the same pollutant concentration in the inflowing water, the pollutant load of a reservoir is usually higher than for a lake, and is related to their relative retention times,
- The pollution concentration in the inflow to a reservoir is usually higher because of the typically larger pollutant-generating drainage basin area in relation to the area of the reservoir.

In rapidly developing tropical countries, some reservoirs simultaneously receive heavy organic pollutant and nutrient loads. Moreover, the pollution is often not only of domestic origin, but also of industrial origin, with toxic materials being produced in the latter. In this

case, the allochthonous organic matter is rapidly decomposed, resulting in poor water quality. Autochthonous organic production may be limited by toxic effects. Such waterbodies are simultaneously polluted with toxic materials and eutrophic. It is not easy, but is of great management important, to distinguish between the two forms of pollution and their relative contributions to the water quality status of a waterbody. A sign of toxic pollution is the discrepancy between the high production potential corresponding to nutrient levels and the actual production and/or algal crops and significantly lower amounts of chlorophyll-a than expected on the basis of the potential trophic level (see Section 3.4.2).

## 6.3 THE EFFECTS OF RESERVOIRS ON RIVERS

A reservoir manager has a duty to take into consideration not only the reservoir itself, but also the reservoir outflows. People living downstream of a reservoir, fish populations in the downstream river and other biota should not be highly disturbed by the construction or operation of a reservoir. Two interconnected management aspects, one quantitative and one qualitative, must be considered.

In terms of river hydrology, different effects of a reservoir on the outflowing river can be distinguished, depending upon specific reservoir uses (see Table 6.5, upper part). In water supply and irrigation reservoirs, the withdrawn water is entirely lost to the downstream river reach. This results in severe water quality consequences and impacts on river biota, especially if the reduction in the river flows is great. As an example, a series of reservoirs on the Colorado River in the western United States reduced the flows in the river to such an extent that flows into Mexico are minimal and the quality of the water is poor. Flow regulation reservoirs ameliorate extreme flow regimes. Nevertheless, negative effects can occur, an example being the disastrous changes in the Nile Delta caused by the Aswan High Dam by the interruption in the continuity of sediments flows. The relation between water quantity and water quality are due to the fact that low outflows not only limit river water use, but also have negative consequences for water quality. Another important aspect is the depth from which water flows out of the reservoir. Water releases from the bottom layer of stratified reservoirs usually have the poorest water quality, compared to other water layers in the reservoir. In geographic regions with pronounced stratification, accompanied by low water temperatures, there are interferes with recreational activities along the river. The quality of the released water and its various uses also are affected by the extent of water level fluctuations. In an extreme case (e.g., the hydroelectric reservoir system with remote operation in the Snowy Mountains of the Australian Alps), it is dangerous to visit the outflowing stream when it only contains scattered puddles of water because people can be swept away without warning in a wave of released water. The physical impacts of a reservoir on the river depend on the flow reductions or increases (Table 6.5, lower part).

The limnological characteristics of reservoirs have strong influences on the degree of reservoir effects in the downstream river into which its water is released. Hydrology plays a major role, particularly in semi-arid and arid regions. The location of the reservoir on

Table 6.5. Impacts of four general uses of reservoirs on the discharges into the downstream river and details of the impacts of discharge modifications (modified from Crisp, 1994)

| Reservoir type | Environmental impact | | | | |
|---|---|---|---|---|---|
| | Modified amplitude of discharge regime | Modified seasonal pattern of discharge | Modified diel pattern of discharge | Decreased average flow | Increased average flow |
| Regulating | + | + | + | | |
| Water supply | + | + | | + | |
| Hydroelectric | + | + | + | | |
| Water transfer | + | ++ | + (donor) | + (recipient) | |
| **Discharge modification** | **Physical impact** | | | | |
| | Bed scour | Sediment deposition | Water velocity | Stream depth | Wetted area |
| Reduced flow | − | + | − | − | − |
| Increased flow | + | − | + | + | + |
| Modified flow pattern | + or − | + or − | + or − | + or − | + or − |

the river along its length determines the degree of the effects. Reservoirs on small headwater rivers have the most profound effects. Shallow, unstratified, wind-mixed reservoirs have a much less profound effect than deep, stratified reservoirs. Further, in hydrologically-shallow reservoirs, the outlet depth has little effect on downstream water quality, while that of stratified reservoirs has a significant effect. Reservoirs with water retention time of only a few days do not have much effect on the outflowing river, whereas the effect increases with increasing retention time up to some critical value, remaining constant above these critical values. Oligotrophic reservoirs have only a small effect, while eutrophic reservoirs have major effects. This multivariate character of reservoir ecosystems should be borne in mind. The listed effects following a change of one variable are only valid in comparable conditions. A deeper reservoir, for example, will have a greater effect on the outflowing river than a shallow reservoir only if it has the same water retention time. Physical, chemical and biological variables in the downstream river are affected by the reservoir in various ways. Factors listed in Table 6.6 represent gross generalizations that do not account for multivariate effects. Further details and many examples are provided by Petts (1984, 1989). The situation within a reservoir at the depth of its outflow is decisive for the quality of the water flowing into the downstream river. Mathematical models of reservoir water quality discussed in Chapter 6 can be used to attempt to predict the consequences of a reservoir on the outflowing river. Some mathematical models also enable reservoir operations that result in optimal effects on downstream aquatic resources. Additional changes can be caused by gas saturation in the dam spill gates (i.e., oxygen saturation and gaseous nitrogen supersaturation), and hydrological effects caused by instantaneous increases in the outflow volumes, such as those occurring below hydropower reservoirs.

Significant differences in the quantities of most variables mentioned in Table 6.6 exist between surface and bottom water discharges. Discharges from either level are either

Table 6.6. Downstream water quality effects (modified from Straškraba et al., 1993)

| Variable | Changes in downstream water quality |
|---|---|
| *Physical variables* | |
| River channel structure | River channels located below dams can be substantially damaged by decreased, increased, and particularly erratic water flow rates |
| Hydrology | Decreased average water flow rates result when the usage of reservoir water is high (e.g., irrigation reservoirs) or when water evaporation rates are high. Periodic short-time flow variability increases and unnatural pulses are produced, with increased variability and disruption of natural hydrological cycles occurring, particularly below hydropower reservoirs. The timing of annual maxima is shifted |
| Thermics | A decrease in average temperature occurs. The degree of temperature decrease escalates with greater water retention time and outlet depth. Geographical distinctions: in temperate regions, temperatures below reservoirs are higher than in unaltered river systems in winter, but are lower in summer; in tropical regions, the temperatures are increased in both winter and summer. The annual temperature range is increased in surface outflow reservoirs, but decreased in deep-water outflow reservoirs. Vernal temperature rises are delayed in surface outlet reservoirs, and even further delayed in deep-water outflow reservoirs |
| Silt content (= turbidity) | Decreased silt loading occurs, which may cause a decrease in floodplain soil fertility, with corresponding consequences on agricultural, wetland and forest productivity |
| Detritus | The composition of particles changes from abiotic to biotic, and particle size decreases |
| Light | Light penetration is increased |
| *Chemical variables* | |
| Oxygen | If the reservoir is eutrophic, and the outlet depth is below the thermocline, the dissolved oxygen concentrations in the outflow water may drop to near-zero values |
| Hydrogen sulfide ($H_2S$) and carbon dioxide ($CO_2$) | Concentrations increase, especially in eutrophic, stratified reservoirs with long water retention times |
| pH | Values decrease, except when the inflow pH is very low (e.g., in blackwater reservoirs in the Amazon drainage basin) |
| Nitrogen | The gaseous nitrogen content increases in aerated reservoirs to supersaturation levels that cause fish kills. This is not due to high gaseous nitrogen in the outflow depth of the reservoir, but rather related to processes that occur during aeration at the spill gates |
| Organic matter | Organic matter concentrations decrease downstream when there are no sources of in-lake organic production. Phytoplankton production can result in overwhelming in-lake organic matter production in highly eutrophic reservoirs receiving low organic matter input |

Table 6.6 (*continued*)

| Variable | Changes in downstream water quality |
|---|---|
| *Chemical variables (continued)* | |
| Phosphorus | The phosphorus concentrations decrease, and the degree of the decrease is higher when the water retention time and trophic degree are greater. An exception occurs when bottom waters are released from eutrophic reservoirs with anoxic hypolimnia. Reduced phosphorus levels result in decreased biological productivity in the outflow river |
| Nitrates | Nitrate concentrations are usually nearly unchanged, but sometimes may increase slightly. When strong reducing conditions exist in the reservoir, the concentration of nitrates downstream is decreased |
| Nitrites | Concentrations of nitrites are usually increased, particularly during deep-water releases from reservoirs of higher trophic degree |
| Total solids | Total solid concentrations remain nearly unchanged |
| Seston | The energetic value of seston is higher in reservoir outflows |
| *Biological variables* | |
| Plankton | Plankton abundance generally increases downstream |
| Phytoplankton composition | The phytoplankton composition changes downstream. In small rivers, there is a transition from riverine (periphytic) species to pelagic species. In large rivers, the number of lacustrine species increases below a reservoir |
| Phytoplankton production | Specific phytoplankton production (per unit mass of phytoplankton) can be extremely high if hypolimnetic phytoplankton enriched with chlorophyll-a are released to the river and reach high light conditions |
| Phytoplankton biomass and chlorophyll-a | The quantities depend on the location of the outflow: Surface outflows deliver more phytoplankton, whereas hypolimnetic releases reduce the transport of phytoplankton biomass. Phytoplankton from the hypolimnion have higher chlorophyll-a content |
| Zooplankton | A small river below a reservoir is highly enriched in zooplankton. In a large river below a reservoir, a transition from potamoplanktonic composition to pelagic composition takes place. Zooplankton biomass usually increases in the outflow, compared to the inflow due to flushing from the lake |
| Benthos | Increases below slightly eutrophic reservoirs, and usually decreases below highly eutrophic reservoirs in relation to anoxia. Composition is usually highly changed. Detrimental effects of short-term water level fluctuations on benthos are high |
| Fish | Reservoirs represent a barrier to fish migration, and the spawning waters often cannot be reached. Feeding habits are often reduced. Fish abundance below reservoirs vary greatly, based on the specific conditions. Changes can include fish kills due to gaseous nitrogen supersaturation, declines in fisheries, and improved oxygen and feeding conditions in some situations |

constant for each reservoir, or variable according to the hydrologic conditions and reservoir operation. Examples of differences between surface and bottom outlets are given in Table 6.7. As can be seen, on the average and for similar sets of variables, the epilimnetic releases result mostly in reduced concentrations, compared to the hypolimnetic releases from the same reservoir. The hypolimnetic releases lead to an increase of concentrations, compared to epilimnetic releases, particularly in eutrophic reservoirs. Such increases are greater in reservoirs with longer retention times. An exception is represented by algae, with their outflow to the downstream river being higher during epilimnetic releases.

Chemical changes in the downstream river below a reservoir are highly affected by the process of reservoir aging. Phosphorus can leave a reservoir for a number of years after its filling, instead of exhibiting decreased concentrations (Section 6.1.3—*Phosphorus Retention*). As an example, Grimard and Jones (1982) estimated an increase of phosphorus concentration from 5 to 22 $\mu g\ l^{-1}$ in LaGrande-2 Reservoir. Inorganic nitrogen also was reported to increase by more than 50% during the aging period of a few shallow large Volga reservoirs (Pirozhnikov, 1984).

The biological consequences of large reservoirs in affecting the nature of downstream rivers are great. If the reservoir causes a reduction in the maximal flows and scouring effects, increased macrophyte growth can occur downstream. Below the reservoir, complete disruption of the river ecosystem is often observed, and can result in fish kills and impoverishment of fish populations. Deterioration of the drinking water supply and loss of high value recreational sites also can occur. However, there also can be positive effects, such as when the outflow from deep reservoirs improves water quality in a polluted river to such an extent that trout or similar sensitive fish species are able to survive. Because reservoirs

Table 6.7. Chemical changes below an epilimnetic release reservoir (Cow Green Reservoir on River Tees, United Kingdom) and from two bottom release reservoirs with different average theoretical water retention times (Vir Reservoir, Czech Republic, $RT = 100$ days; Slapy Reservoir, $RT = 38$ days). The inflow value in concentration in $mg\ l^{-1}$ and the outflow value in percent of the inflow. Data from Petts (1986) according to Holmes and Whiton (Cow Green), according to Penáz et al. (1968) (Vir), and data from Hrbáček and Straškraba (1973) (Slapy Reservoir)

| | Epilimnetic release | | Hypolimnetic release | | | |
|---|---|---|---|---|---|---|
| | River Tees | | Slapy Reservoir | | Vir Reservoir | |
| | Inflow | Outflow | Inflow | Outflow | Inflow | Outflow |
| Sodium | 3.6 | 72 | | | 6.4 | 178 |
| Potassium | 1.01 | 51 | | | 3.7 | 168 |
| Magnesium | 2.03 | 37 | 4.4 | 100 | 1.8 | 233 |
| Calcium | 36.3 | 24 | 16.4 | 95 | 12.3 | 162 |
| Sulfate—S | 5.2 | 78 | | | 19.4 | 160 |
| Nitrate—N | 0.13 | 154 | 0.83 | 111 | 5.5 | 127 |
| Ammonia—N | 0.02 | 200 | 136 | 163 | 0.3 | 100 |
| Phosphate—P | 0.025 | 121 | 0.052 | 81 | 0.18 | 78 |
| *Average* (except for phosphorus) | | 98 | | 137 | | 171 |

function as settling and biological oxidation basins and phosphorus traps (Section 6.1.3), a significant improvement in outflow water quality typically occurs.

The extent of the river length affected by the reservoir, and the point at which conditions in the river return to normal for an undamped river, is defined by the *reset distance* (Ward et al., 1984). This distance can vary considerably, and is determined by the reservoir position in the drainage basin, the geography of the area and other variables. Inflows that enter the river below a reservoir also are important. For example, below Balbina Reservoir (Brazil), deoxygenated waters persist in the downstream river for as far as 20 km from the reservoir outflow. The Eildon Dam outflow modifies conditions in its downstream river for 138 km, and the temperature effect below Hume Reservoir (both in Australia) ceases only after 200 km in the downstream river. The effect of large reservoirs can extend as far as river deltas, and result in the loss of estuary ecosystems, reduction of fishery habitats, and saltwater intrusion into delta farmlands.

Fish kills take place below reservoirs when anoxic hypolimnetic water is released. Reservoirs also often hinder natural fish migration, leading to serious impoverishment of fish fauna. The construction of fish passes in dams is a delicate issue, and many fish ladders constructed for this purpose do not function properly. Further, in many cases, fish passes were not constructed at all.

## 6.4 MANAGEMENT OF RESERVOIR WATER QUALITY

The dual goal of reservoir management is to ensure sufficient quantities of water of acceptable quality. Good water quality is of prime importance for drinking water supply reservoirs, but human life requires good quality water in most instances. In fact, water from most reservoirs is used for many purposes, both directly and indirectly.

The following four basic options for managing water quality in a reservoir and/or its outflowing or withdrawn water are currently in use, with some being common for both natural and man-made lakes:

- Management of the water quality in the reservoir drainage basin,
- Management options applied in the reservoir itself,
- Use of selective offtakes to withdraw water of the best possible quality,
- Use of additional means to improve the gas conditions of outflow water.

The decision regarding which option to use in a given situation requires consideration of the feasibility of the options, estimation of the probable effects, and estimates of the costs and needed manpower. Some recent mathematical models belonging to the category of specific reservoir models (e.g., Davis et al., 1987a, or Fontane et al., 1981, for selecting best layers, or Grimard and Jones, 1982, for estimating the reservoir aging process), multiparameter optimization models (e.g., Kalceva et al., 1982) or to the category of decision support systems, *DSS* (e.g., Davis et al., 1987b; Davis et al., 1991) can be helpful. Further details are provided in Chapter 5.

### 6.4.1 Management of Water Quality within the Reservoir and Its Drainage Basin

Simultaneous management of the water quality of both an entire reservoir drainage basin and the reservoir itself is the best option for achieving high quality in a reservoir and its outflowing water. General methods for achieving this goal were provided in Chapter 4.

### 6.4.2 Management Approaches Specific to Reservoirs

In addition to the common methods of water quality management, previously discussed in Chapter 4, there also are a number of methods specific to reservoirs. These methods include the possibility of manipulating the hydrological conditions of a reservoir by selective water withdrawal, the Wahnbach method of phosphorus precipitation at the reservoir entrance, the use of pre-impoundments, methods of managing reservoir outflows, and methods utilizing reservoir systems.

*Selective withdrawal (hydraulic regulation)*

*Use of selective offtakes.* Selective water offtakes use an outflow depth or depths that allow utilization of the water of the best quality in a reservoir. This is particularly important in offtakes for drinking water treatment plants, but also important for river outflow considerations. The use of selective offtakes in reservoir water quality management has important consequences for reservoir outflows. However, it must be noted that any change of the outflow depth has consequences for the quality of the water at that depth. The outflowing water is replaced by water from surrounding strata, and usually results in differing water quality. Thus, the dynamic nature of water quality must be acknowledged and respected. The use of reservoir water quality models that address hydrodynamic conditions (Chapter 5) offers considerable help in attaining this goal.

The use of selective offtakes is a fairly recent practice since water quality and environmental problems were not considered during the construction of most existing reservoirs. McMahon and Findlayson (1995) pointed out the high costs incurred when selective offtakes must be constructed in existing reservoirs for environmental reasons. For example, they estimated that the cost of one recently-constructed new outlet in the Upper Yarra Dam in Australia was US $10 million. However, the method outlined below, using plastic curtains, offers a less expensive substitute. Reports of successful uses of selective offtakes for water quality management (Christenson et al., 1996), and for problems with downstream water quality (Yogo et al., 1997) suggest that the use of mathematical models for optimal performance is useful (Fontane et al., 1981; Edelshtein and Puklakov, 1996).

Hydraulic regulation that selects the best time and depth of the water outflow is the cheapest option for improving water quality. This is a good method, provided the limnological situation of the reservoir is known beforehand. A simple, but rarely adopted, precaution is to ensure at the planning stage that the reservoir possesses a multiple outlet structure with the outlets spaced vertically at about 5 meter intervals. A minimum provision would be a surface and a bottom possibility of water discharge. A better control is afforded by reservoirs with multiple offtake horizons, so that water of good quality can always be

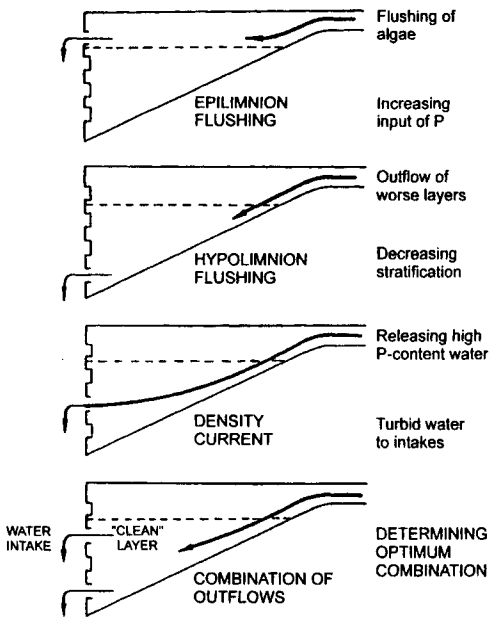



Fig. 6.7. The use of selective offtakes for managing reservoir water quality by hydraulic regulation. In each case, some positive (upper row) and some negative effects take place. The management must be directed to domination of the positive effects. A hydrodynamic model may help to decide the best conditions for improving the situation.

withdrawn by selecting the appropriate outlet depth in relation to thermal stratification (Fig. 6.7).

The following options are possible with the use of multiple offtakes:

- Removal of excess algae by using near-surface outlets,
- Removal of near-bottom water layers with deteriorated quality by using deep outlets,
- Using short-cut currents (undercurrents) for rapid passage of the inflow water peaks with the worst water quality,
- Selecting the layer of best quality water for intakes to the drinking water treatment plant.

Any of these options has both positive and negative effects on water quality. Using the *near-surface outlets* (the first bullet above), the excess algae can be flushed out, as demonstrated for Slapy Reservoir (Javornický and Komárková, 1973). However, the phosphorus input is increased by directing all inflow water to the surface layers, thereby increasing the algal production rate. The retention time of the surface layers must be less that 10 days to allow significant flushing of algae. The flushing rate has to more than compensate for the increased algal growth rate.

When attempting to dilute or remove the worse hypolimnetic water layers by *a near-bottom outlet*, a negative impact is produced by decreasing the stratification. Higher flow rates of deep water layers can create shear stress, causing more water mixing with the

upper layers. In addition to releasing anoxic waters, another positive effect is to increase the oxygen content, which will decrease sulfide accumulation (and toxicity effects), raise the redox potential and enhance microbial processes.

In creating short-cut currents, the rapid passage of the inflowing water through the reservoir can be used to get rid of pollution spikes. The spring floods often represent the dominant phosphorus input, and the probability of high summer algal crops and hypolimnetic anoxia decreases if these floodwaters pass rapidly through the reservoir without being extensively mixed with other water strata. In a similar manner, an accidental spill of toxic, radioactive or other water degrading substance can pass rapidly through a reservoir without much interference or in-reservoir mixing. However, this also does not necessarily occur without a negative side effect. Turbid water reaching the layer used for drinking water offtake during the phosphorus flushing, for example, increases the need for chemicals at the water treatment plant. Thus, it is necessary to determine in each particular situation whether the positive or negative consequences will prevail. The best means available to help in this decision is a good hydrodynamic model (Chapter 5). Hypolimnetic releases in reservoirs must be tested before they are done, by calculating the nutrient budget prior to and after possible releases. The numerical model SELECT (Davis et al., 1987a) is capable of simulating the withdrawal limits of the various proposed applications of the withdrawal approach. It is possible to determine the maximum possible flow rate under a given stratification, in order to minimize the release of epilimnetic water.

For the selection of the best water layer, automation of monitoring water quality parameters (depth profiles of temperature, dissolved oxygen and water transparency) is provided by the "clear layer" equipment of Pařízek (1984). The procedure is based on the observation that the most transparent water layer typically contains the best quality water, with the lowest concentrations of organic substances. Automatic determination of the most transparent layer drives the selection of the best layer for a drinking water treatment plant. Nevertheless, it is best to determine whether or not the assumption that the most transparent water layer contains the lower content of organic substances holds true. A considerable saving on chemicals used for drinking water treatment can be achieved in this way.

The *advantage* of the method is its *cost*, which is nearly zero. There are some *limitations*, including the need for good knowledge of the inflow water quality and its depth distribution within the reservoir. Moreover, the knowledge of changes in water quality under the effects of changing the outlet or offtake depth is helpful. Possible strong *negative impacts* on the downstream users also must be considered.

*Curtains.* When multiple outlets structures are not available, it is still possible in some instances to use plastic curtains. The use of plastic curtains was recently shown to be a successful method for modifying the outflow depth (e.g., Asaeda et al., 1996). However, the depth selection is more limited than with multiple outlets. Creation of a near-surface and near-bottom outlet can be realized by a curtain of a corresponding height anchored at the bottom and lifted by floats. Another way is to anchor a curtain at a certain height above the bottom and float it up to the surface (Fig. 6.8). Outlets from intermediate depths need more elaborate arrangement. Considerable leaking around the edges and bottom must be taken

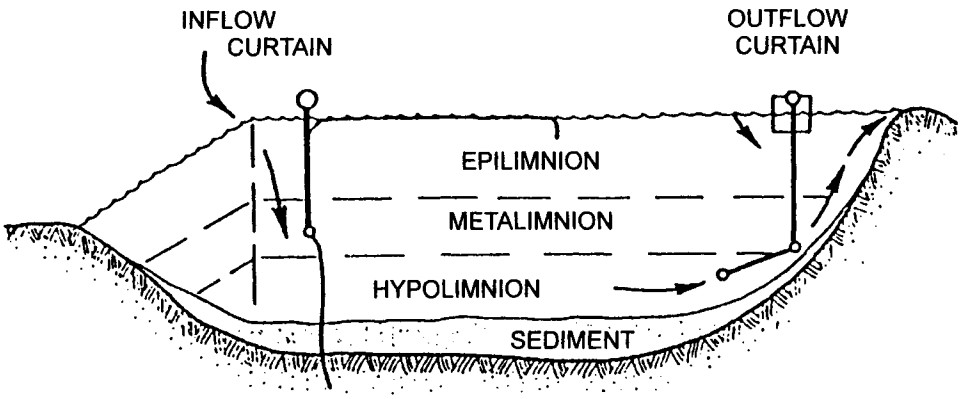



Fig. 6.8. The use of plastic curtains as a partial substitute for water quality management by hydraulic regulation (from Aseada et al., 1996).

into account. Because the pressure exerted on the curtains from internal water movements is enormous, the curtain material must be of high quality and strong enough to endure it.

Curtains also have been used in a few instances at the inflow to direct the water flow to the hypolimnion or epilimnion, respectively. Considerable mixing with the reservoir water is created as the water passing through the curtain submerges or rises according to the density differences.

*Phosphorus retention by pre-impoundments*

As shown above, reservoirs can trap both total and orthophosphate phosphorus efficiently through particle sedimentation, and algal uptake of the dissolved nutrient followed by algal sedimentation. This phosphorus retention capacity can play an important role when using this feature of reservoirs to decrease the phosphorus concentrations with the use of small reservoirs (called pre-impoundments) located at the inflows of the main reservoir. The design of a pre-impoundment is illustrated in Figure 6.9. The reservoir must be deep enough to enable intensive sedimentation of the inflow particles as well as the algae produced in it. In the optimal design, a sill enables the sedimentation of inflow particles to be concentrated in the narrow inflow part, with a baffle helping to the steady water flow through it.

This feature was recognized by a group of limnologists from Dresden, Germany (Uhlmann et al., 1971; Uhlmann and Benndorf, 1980; Benndorf, 1973; Benndorf et al., 1975; Pütz, 1995; Pütz and Benndorf, 1998), who developed a calculation method for sizing such pre-reservoirs for the maximum retention capability. The calculation method is outlined in Figure 6.10. The phosphorus removal rate increases with the average water retention time, as seen in Figure 6.11 for reactive phosphorus (OP) and total phosphorus (TP). The figure also indicates a good agreement between phosphorus removal predictions and observations for 11 pre-impoundments in Germany. In temperate regions, there is a *seasonal* trend in the elimination rate due to changing phytoplankton development, and this

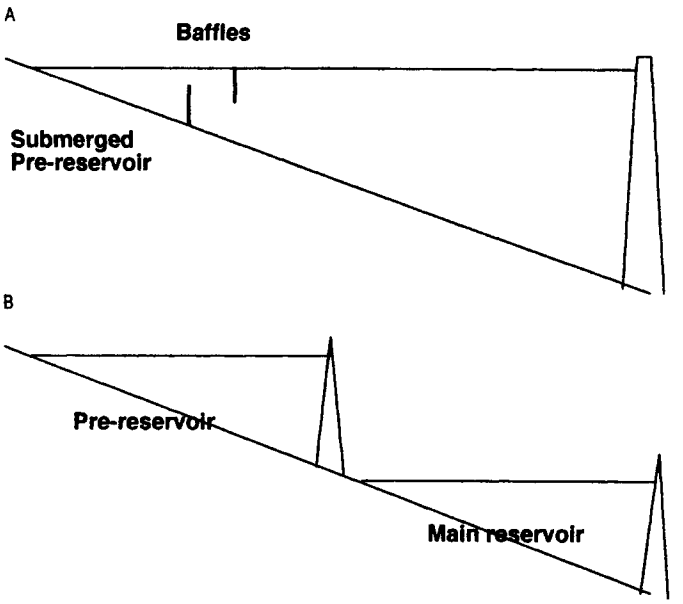



Fig. 6.9. Design of pre-impoundments (= pre-reservoirs). A—Submerged pre-impoundment with baffle. During the low water stage in the reservoir, the pre-reservoir functions as in B; B—Standard pre-impoundment build on inflowing streams.

trend is also well described by the calculation model. Long-term observations of phosphorus removal efficiency for another German reservoir are presented in Figure 6.12.

Paul et al. (1998) increased the phosphorus retention of the Saidenbach Reservoir in Germany with a submerged, overflown flexible curtain across the mouth of the inflow. The curtain was a cost-effective substitute for conventional pre-impoundments, causing a 30–40% increase in the soluble phosphorus retention.

*Water level operation*

Due to reservoir outflows that can be manipulated, water level operations, with positive effects on reducing macrophyte vegetation and decreasing organic matter in the drying shore sediments (Section 4.3.3), are more easily performed in reservoirs than in lakes. Further, the manipulation of fish reproduction by a decline in the water level at the time of laying eggs among shore vegetation is possible.

However, negative effects also can be created by the water level drawdown. As the newly-exposed sediments release water, air bubbles can escape from the supersaturated pore water and the sediment may become unstable and cause slumps. The larger the water drawdown, and the steeper the reservoir sides, the more frequent slumps may occur. Slumps have comparable effects in regard to extreme flood events, in that they can cause relocation of sediments into deeper water strata, cause turbid water layers unsuitable for treatment, and cover bottom-living organisms, destroying some of them.

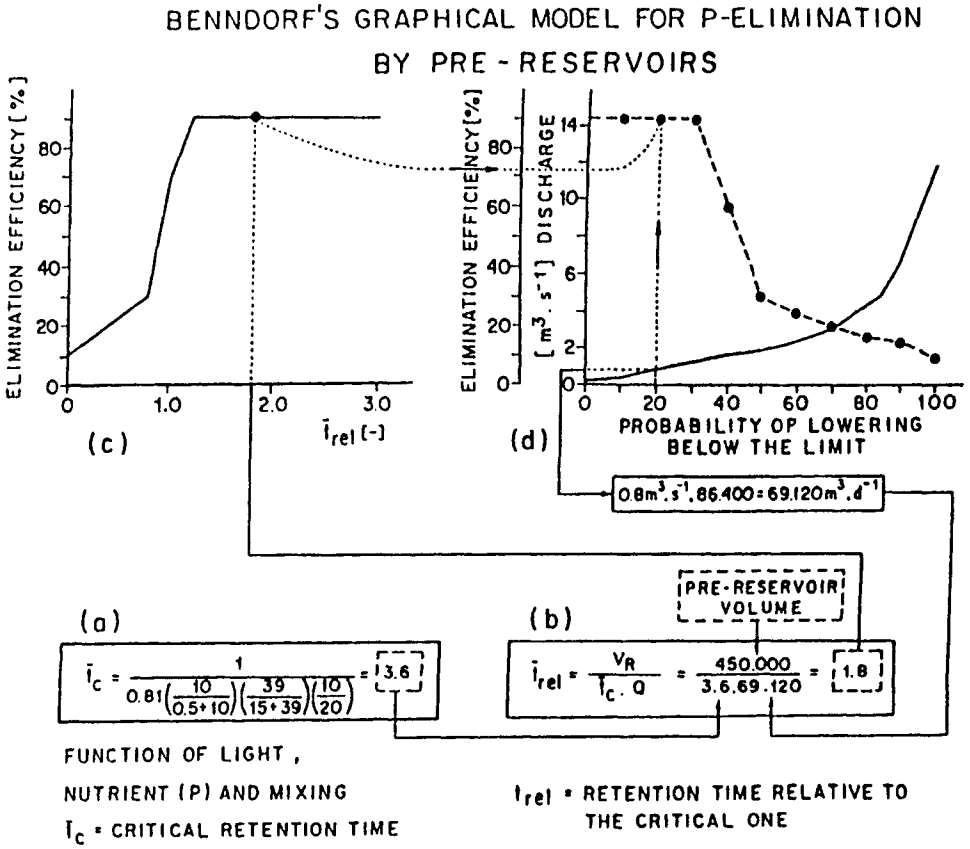



Fig. 6.10. Graphical model for the calculation of pre-reservoir efficiency. The model can be used for sizing the pre-reservoir at a stream with a given flow, for the given size of the pre-reservoir, or for estimating pre-reservoir functioning during different flow, phosphorus concentration and solar radiation conditions. For the calculation of the critical retention time (at which maximum efficiency is retained), the first bracket is for the effect of the phosphorus concentration, the second for the effect of radiation, and the last for the effect of the depth relative to the optimum depth ($z_{max}$) of 10 meters. In the expression for phosphorus, the two higher values represent the measured phosphorus concentrations, and the low value the half saturation constant of phytoplankton growth for phosphorus. The middle term gives average values of solar radiation during the period of calculation and the corresponding half saturation constant for phytoplankton growth (from Benndorf, 1973).

### 6.4.3 *Management of Reservoir Outflows*

The goal of management of reservoir outflows is to maintain sufficient water quantity and quality conditions for downstream use and aquatic life.

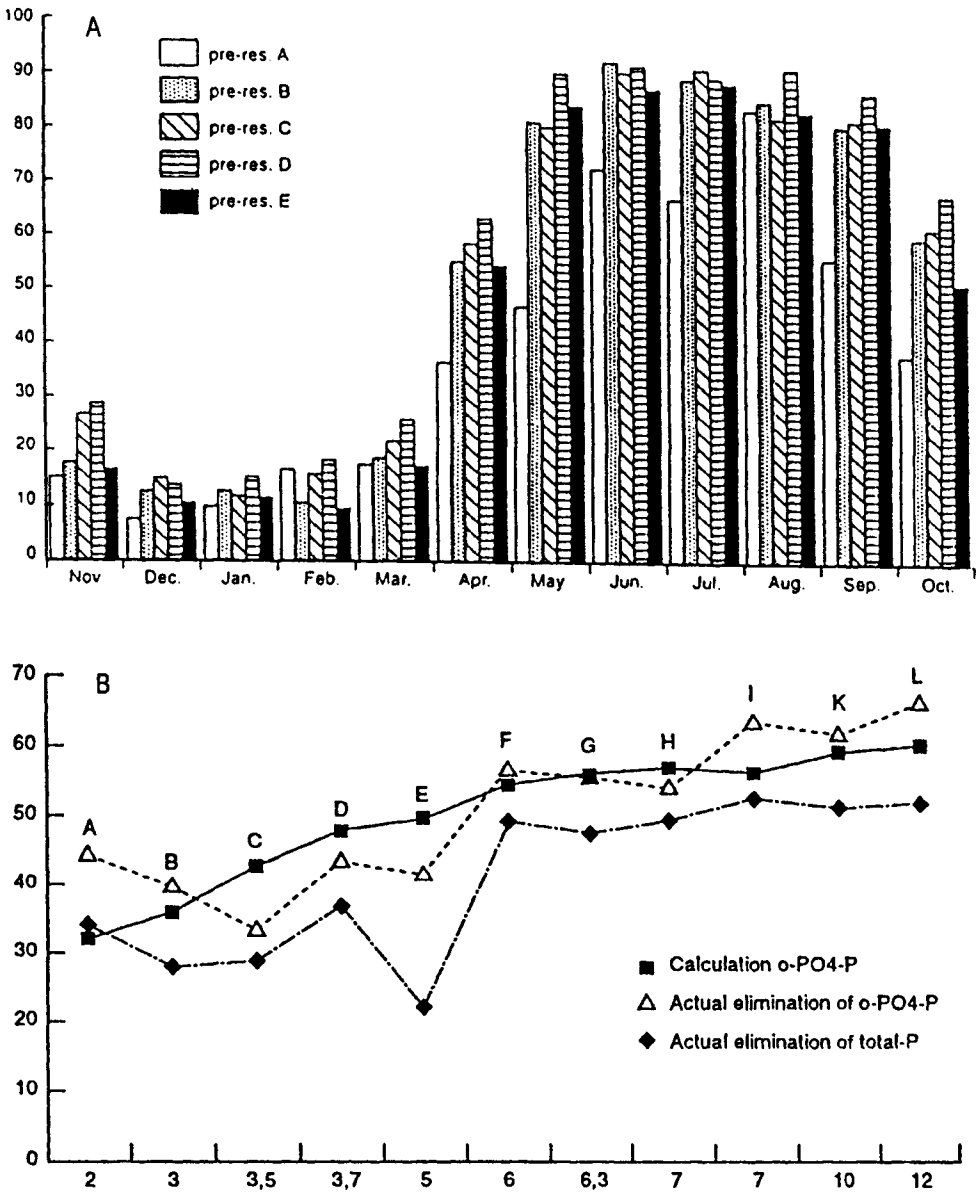



Fig. 6.11. Comparison of the calculated and observed pre-reservoir efficiencies for a number of pre-reservoirs in Germany. A—Seasonal changes in 5 different pre-reservoirs; B—Comparison for 11 pre-reservoirs with different retention times, showing a very good agreement and lower efficiency achieved for total-P than for orthophosphate (labile inorganic) phosphorus (from Pütz, 1995).

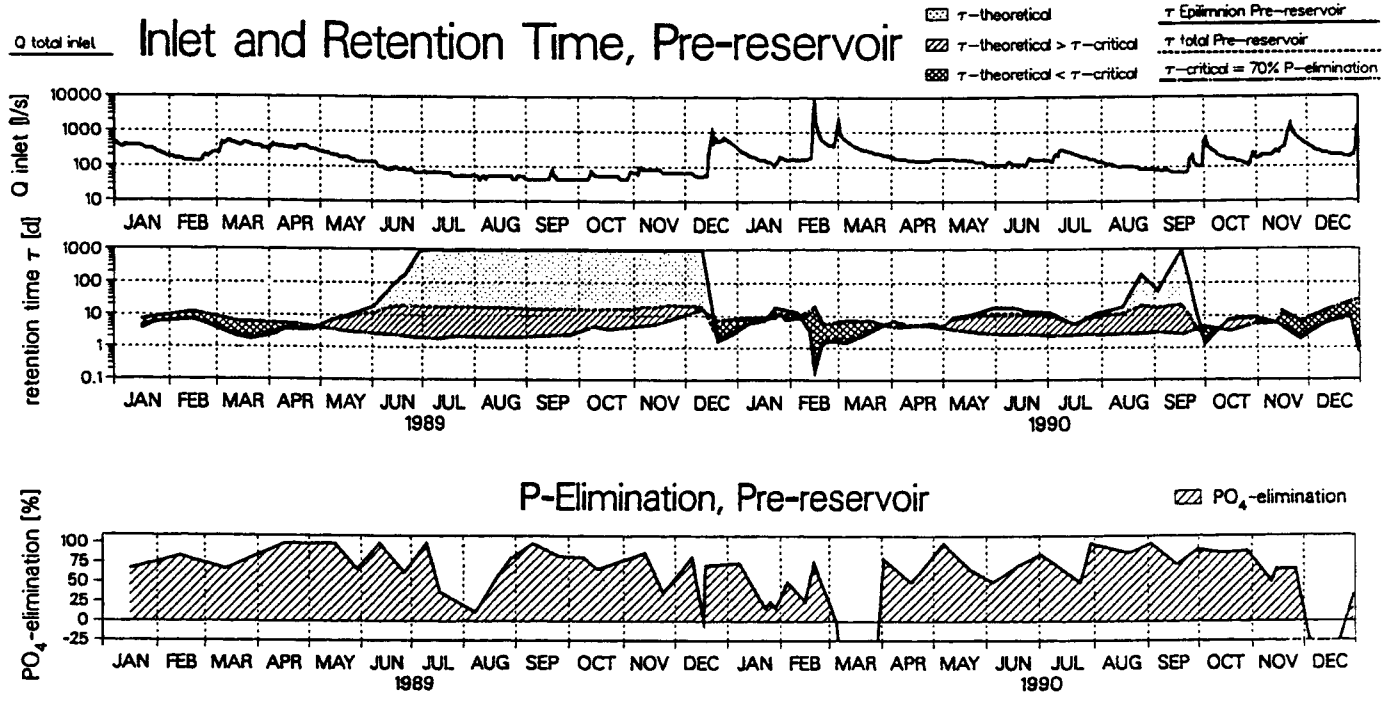



Fig. 6.12. Regular observations of the retention efficiency of a pre-reservoir in Germany and comparison with values calculated according to Benndorf's model. The inflow rate, theoretical water retention time of the total pre-reservoir, its epilimnion, and comparison with the critical retention time for achieving 70% phosphate—phosphorus elimination according to the model of Benndorf.

The quantitative aspect, maintenance of adequate water flow, is usually an obligation for the manager prescribed in the form of minimum permissible flows. Additional requirements may be defined in the form of maximum flows and rates of change in flow (Morehardt, 1986). Quantitative methods for determining minimum flows are not yet well developed, and local conditions of downstream users must be taken into account. Some methods used in the United States and the United Kingdom are based on conditions for downstream fish life (Binns and Eiserman, 1979; Orth and Leonard, 1990). The input variables for calculating these flows are relatively simple ones, including depth, discharge or velocity, basin area or distance from one stream shore to the other along the bottom. A hydrologic model is used to estimate the consequences of flow variations. The determination of habitat suitability for locally-important stream fish species is necessary.

The task of determining minimum permissible flows simultaneously from the quantitative and qualitative aspects can be difficult. Although higher water discharges are generally considered more favorable than low flows, a simple augmentation of water quantity does not solve water quality problems below reservoirs polluted by urban and industrial effluents or by irrigation return flows. Methodologies along these lines have been formulated (Gore and Judy, 1981; Richter et al., 1997). Moog (1993) formulated management strategies to minimize the impacts of peaking operations on stream fauna, and Cada (1998) reviewed the efforts to reduce the impacts of power supply reservoirs on reservoir fisheries. Mathematical models used to help minimize the environmental impacts of hydroelectric reservoirs are discussed by Dionne and Thérien (1997). The operation of a temperature control device for management of the in-reservoir and outflow temperatures was modelled by Hanna (1998).

Possibilities for improving water quality at the location of the dam consist mainly of modifications of the gas regime of the outflow. Table 6.8 lists alternative techniques used to manage outflows. Rather than discussing the details of each procedure, a significant reference is provided for each method.

Several approaches regarding aeration/oxygenation that were previously discussed in Section 4.3.2 can be included in managing hydropower outlets. The use of destratification in the vicinity of the dam is shown in Figure 6.13A. Figure 6.13B illustrates the use of a combination of oxygenation from the bottom, and mixing by a propeller or a water

Table 6.8. Techniques for management of reservoir outflows. The references are either to the inventor, to a thorough summary of the usage, or both (modified from Sraškraba et al., 1993)

| Technique | Reference |
|---|---|
| Selective withdrawal | Gaillard, 1984; Parízek, 1984; Filho et al , 1990 |
| Aeration/oxygenation at hydropower outlet works | Cassidy, 1989 |
| Vacuum-breaker venting system | Price and Meyer, 1992 |
| Spill-water re-aeration | Cassidy, 1989 |
| Czech method of oxygenation | Haindl, 1973 |
| Epilimnetic pumps | Quintero and Garton, 1973; Mobley and Harshbarger, 1987 |

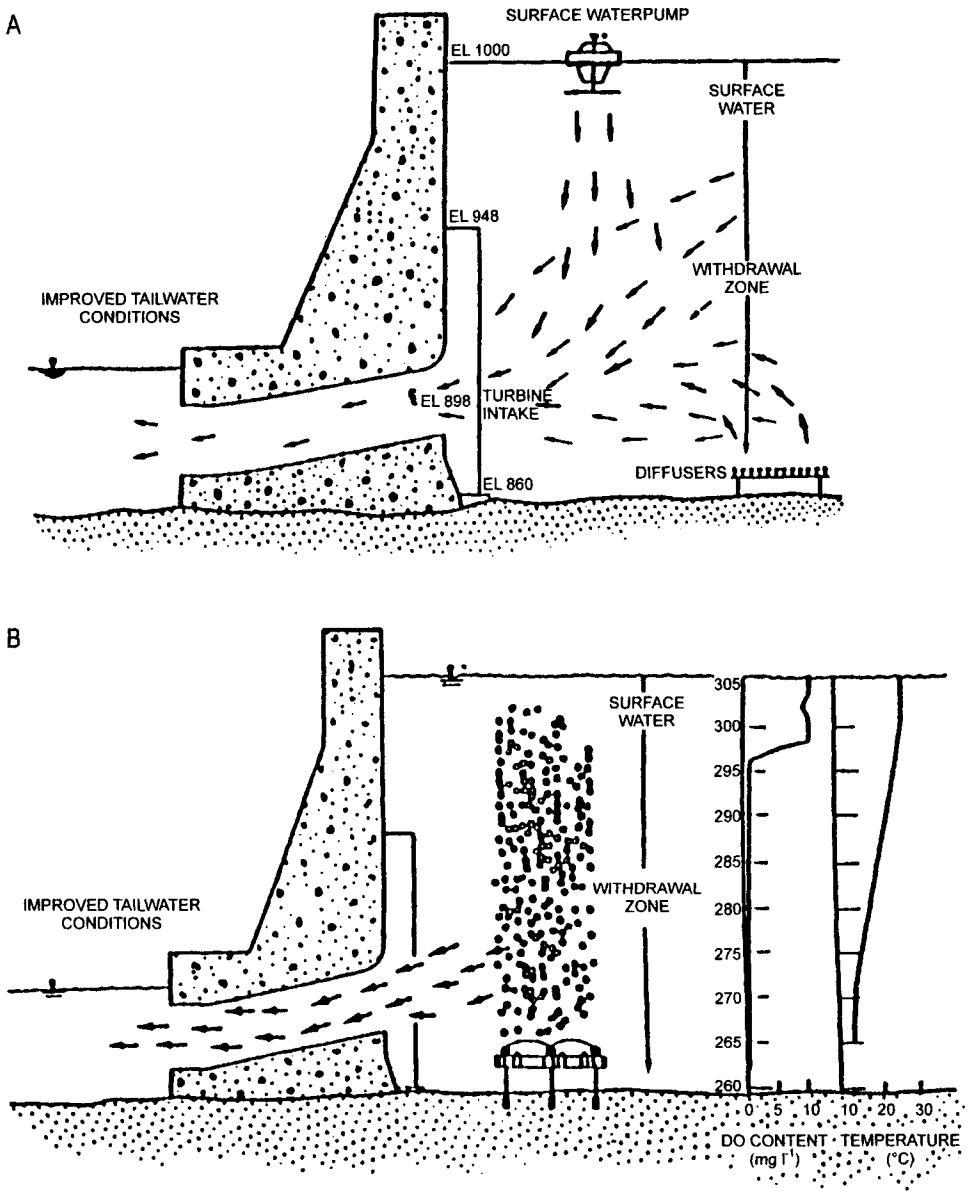



Fig. 6.13. Increasing oxygen concentrations in the reservoir outflow by mixing and aeration. A—Using propeller mixing and oxygenation by a diffuser in the close vicinity of the dam. B—Destratification close to the dam. The profiles of temperature and oxygen before the mixing started is shown (from Mobley, undated).

pump from the surface (Mobley, undated). An improvement is achieved in cases of deoxygenated water at the turbine intake levels. The outflow oxygen concentration can be increased to levels suitable for fish and other aquatic life, and devoid of smells, by using such approaches. Further, outflow temperatures can be increased, in comparison to unmanaged situations, and structures also are better protected against corrosion.

Problems may arise when the water comes from the hypolimnion of an eutrophic reservoir. This is because the water is deoxygenated, contains large concentrations of organic compounds, iron and manganese, and has a high phosphorus content. The consequences may include large-scale fish kills and eutrophication of the river, accompanied by taste and odor problems that make its treatment difficult for a drinking water supply.

For reservoirs, there also is a possibility to improve water quality downstream during the release of water from the reservoir into the outflowing river. Many technical components were invented to increase the oxygenation of the water being released from a reservoir via turbines or overflow, examples being spill water aeration (Cassidy, 1989), vacuum breaker venting systems (Price and Meyer, 1992), infuser weirs and high-performance weirs (Hauser and Brock, 1996; Hauser and Morris, 1995), and auto-venting and re-aerating turbines (Hopping et al., 1997, 1999; March et al., 1992).

The quality of water flowing out of a reservoir is directly related to the horizontal and vertical distribution of the water within the reservoir (and the quality of the water), as well as the flexibility of the operation of the multiple outlets (Cassidy and Dunn, 1987; Ashby et al., 1995). There is a strong relationship between water quality and the quantity of water flowing out of a reservoir. When the outflow quantity is low (in relation to the reservoir volume), only a thin layer is taken to the outlet. With large water releases, the water quality can significantly change as broad layers of water are taken to the outlet, and as more inflowing water enters different layers to replace the outflowing water.

Selective withdrawal was previously discussed as a means for managing water quality within a reservoir. Several management options for re-oxygenation are available, including spill water oxygenation and the use of hydraulic skis.

### *6.4.4 Management of Reservoir Systems*

Reservoir systems are managed in respect to both water quantity and quality. The quantitative management is typically solved with detailed mathematical optimization methods. The problem is how to best manipulate the water outflow, uptake and pumping from individual reservoirs to achieve the maximum water use, while keeping the constraints of slow and low water level declines, minimum releases and others.

When water quality considerations are included, the task becomes much more complicated because the number of variables and constraints to be considered increases. Nonlinear interrelations between water quantity and quantity must also be considered.

The simplest consideration is the quantity operation of reservoir cascades. The structure of the system is simple: Water flows unidirectionally from the uppermost to the lowermost reservoir. Each reservoir improves water quality by natural processes of sedimentation, production, etc., occurring within the reservoir. Suspended solids, organic matter and phos-

phorus, as well as other polluting substances, are decreased by the passage of the river through the reservoirs. Because the decrease is in percentage (but also strongly depends on retention time), the uppermost reservoir usually decreases the pollution most intensively if its retention time is not very short. The cheapest and very efficient water quality management approach is hydraulic regulation. By selecting the proper depth levels for the water uptake and outflow, considerable water quality improvement can be achieved.

Operation of pumping schemes is usually not combined with water quality problems, either because the quantity of water being pumped is low, or because the reservoirs of the pumping scheme are used solely for hydropower generation.

A specific method using reservoir systems is *storage sequencing* and *reservoir blenching*. If reservoirs in a system have different water quality, it is possible to optimize water quality for the supply by properly scheduling their use (Kortmann, 1989). The rule is to use the most eutrophic reservoir early in the growing season, whereas less eutrophic reservoirs are used later in the growing season and during fall overturn. Moreover, if one reservoir in the system is devoted to water distribution to users, it is possible to use selected offtakes to obtain the best possible quality.

## 6.5 CONSIDERATIONS FOR NEW RESERVOIRS

Although the period of intensive reservoir construction around the world appears to be over, reservoir construction nevertheless continues in many developing countries. The problems associated with various water quality and nontechnical aspects during the construction of reservoirs fall into the following categories:

- Consideration of positive and negative aspects of a given reservoir and its site selection,
- Evaluation of environmental consequences of reservoir construction and its alternatives,
- Determination of optimal reservoir features, from a water quality perspective.

### *6.5.1 Issues in New Reservoir Construction*

*Reservoir aging.* Poor water quality is a feature of newly-constructed reservoirs during the period ranging from a few to up to 10 years after they begin to fill. This process is termed reservoir aging or "trophic upsurge". The length of this period depends on several variables. It is shorter when the average water retention time of the reservoir is short, with the reverse being true when reservoir filling takes several years. It also is longer when the nutrient and organic matter input from soil and drowned vegetation is high, whereas increased temperatures make it shorter under otherwise comparable conditions. Reservoir aging is marked by deteriorated oxygen conditions in deeper reservoir water strata, increased manganese and iron concentrations in the reservoir, and more complicated and expensive drinking water treatment. The name "trophic upsurge" is based on the increased productivity levels during the first few years of the reservoir's existence. It is also observed in the form of high fish yields several years after filling is complete. Fishery biologists in

the former Soviet Union predicted high fish yields from the huge reservoirs in the country. These high yields, however, were not observed after the period of trophic upsurge passed (for more details, see Section 6.5.4).

Before the planning for any reservoir construction begins, it is both wise and prudent to consider is potential positive and negative consequences from a broad perspective (Goldsmith and Hildyard, 1984; Cernea, 1988; UNEP/UNESCO, 1990; Straškraba and Tundisi, 1999). Alternatives to reservoir construction (energy- and water-saving programs, utilization of other water sources, location of the use for which the reservoir was built being directed elsewhere, etc.) should be evaluated. The reason for this approach is that many reservoirs constructed in the past appeared not to fulfill the expectations and/or have a number of negative impacts. Table 6.9 lists possible positive and negative consequences of reservoir construction to be considered. Some might be irrelevant for a specific reservoir, but it is nevertheless useful to consider all possible aspects. The fact that the list of negative effects is longer than the positive effects should not create the impression that negative effects dominate. It only expresses the need to consider more items from a water quality perspective, as discussed below in more detail.

An example of the extent of positive and negative effects is provided by the Aswan High Dam in Egypt and Sudan. The reservoir produces 7000 million kWh of electricity annually, enables cultivation of 3655 $km^2$ of land, and has increased the national income of Egypt by US $500 million annually. The negative consequences also are rather extensive, but are not yet expressed well in economic terms. The infection rate of bilharzia increased from 0 to 80% because of the dam and irrigation ditches, the catch of sardines in the eastern Mediterranean Sea decreased 95%, and the recreation industry in the Nile Delta, particularly around Alexandria, is threatened because the dam prevented beach replenishment. Moreover, the annual silt enrichment of the Nile floodplain has been eliminated by the dam.

*Social problems.* The primary socioeconomic consequence of reservoir construction is the need to relocate people. In the recently-realized gigantic Chinese project of the Three Gorges Reservoir, it is expected that more than one million people will be relocated. To the present time, the greatest relocation of people was related to the construction of the Aswan High Dam (i.e., Lake Nasser), with 120,000 people being relocated. This is followed by Lake Volta with 70,000 people, and Lake Kariba and Lake Kanji with 50,000 people relocated. The Caborra Bassa Reservoir in Mozambique required the resettlement of 25,000 people to new villages, which was done with due respect for family and tribal groupings. The relocation of people away from their traditional scattered communities nevertheless led to many social problems.

Population displacements are often accompanied by the loss of jobs and other adverse socioeconomic consequences. The World Bank (1993) stated: "Involuntary resettlement gives rise to a great extent different from, and usually more severe than those encountered in voluntary resettlement. A feeling of powerlessness and alienation is often engendered in those who are relocated, especially when entire communities are uprooted from familiar surroundings. Therefore, whenever possible involuntary resettlement must be avoided or minimized, and alternative development solutions must be explored".

Table 6.9. Possible positive and negative effects of reservoirs (from Straškraba and Tundisi, 1999)

Positive effects:
- Production of hydropower energy
- Creation of low-energy water purifiers
- Retention of water in the area
- Creation of drinking water and water supply sources
- Representative reserves of biological diversity
- Increased economic possibilities for portions of the local populations
- Creation of recreation possibilities
- Protection of downstream areas from flooding
- Increased fishery possibilities
- Water storage for low-flow periods
- Enhanced navigation
- Increased irrigation potential

Negative effects:
- Displaced local populations
- Deteriorated conditions for original population
- Health problems created by spread of waterborne diseases
- Loss of edible native riverine fish species
- Loss of agricultural land and valuable timber resources
- Loss of wetlands and land/water ecotones—useful natural structures; loss of the natural floodplain and conversion of wildlife habitat
- Loss of biodiversity (unique species); displaced wildlife populations
- Loss of mature agricultural land, well-cared for generations (e.g., rice paddy)
- Excessive human immigration to the reservoir region, and increases in many associated social, economic and health problems
- Need for adequate compensation for loss of agriculture, fisheries grounds and housing, as well as compensation for loss of fishing, recreational and subsistence activities
- Degradation of local water quality
- Decreased flow rates below the reservoir and increased flow variability
- Decreased downstream temperature and silt/nutrient transport
- Decreased dissolved oxygen at the bottom and the outflow of the reservoir
- Decreased pH and organic matter content in the outflow water
- Increased hydrogen sulphide ($H_2S$) and carbon dioxide ($CO_2$) at the bottom and outflow of the reservoir
- Barrier to fish migration
- Loss of valuable historic or cultural resources (e.g., loss of countless Native American burial grounds and other sacred sites in Oregon, resulting in the loss of cultural identity for some tribes)
- Decreased aesthetic values

Secondary social impacts may include:
- Internal anthropogenic pressures on labor resources, such as the end of the nomad era and the necessary re-training of nomadic shepherds into settled farmers, urbanization and industrialization of the territory, economic development connected with new access roads and electricity, increased recreation pressure, etc.,

- Effects of changed environment on the health and productivity of humans and domestic animals, namely with regard to new waterborne diseases,
- Potential danger of disasters resulting from dam breaking.

*Health problems.* Large reservoirs in warmer climates are associated with the spreading of infectious diseases (hepatitis, poliomyelitis, typhoid, dysentery, cholera, etc.) and with mass reproduction of parasites. The greatest threat is associated with irrigation projects, which creates good conditions for the proliferation of parasites. Mosquitos, blackflies and other animals are vectors of malaria, virus infections and intermediates for bilharzia and onchocerciasis. In the Amazon River drainage basin, people living around Tucurui Reservoir suffered from a large attack by blackflies, which caused several hundred cases of encephalitis. The health effects of some major reservoir projects are summarized in Table 6.10.

*Weed problems.* After Lake Kariba was flooded, a large infestation of the floating fern *Salvinia molesta* appeared and covered large areas of the lake surface (Magadza, 1988, 1997). A very inefficient strategy was used; many specialists were engaged to solve the problem, which finally disappeared by itself. The reason for the disappearance was the exhaustion of nutrients entering the lake as a result of the floods. If a nutrient budget calculation had been made before the reservoir construction, and the aging process (Section 6.5.4) taken into account, these unnecessary negative impacts might have been avoided. (See Chapter 9.)

*Rapid filling with sediments.* The length of the operational life of a reservoir is much shorter than that of a natural lake, due primarily to its more rapid filling with sediments. The primary reason is the larger drainage basin : waterbody ratio of a reservoir, as compared to a natural lake. The consequences are major water quality changes in the inflow part of the reservoir, due to the influence of the accumulated sediments, which usually contain significant quantities of easily-decomposable organic matter (soil particles). As a result, bacterial decomposition of the organic matter consumes the oxygen in the waterbody. Anoxic conditions are caused by the remobilization of phosphorus and other polluting substances

Table 6.10. Major health problems connected with reservoir construction (from data by WHO, with supplement from Gleick, 1999)

| Project | Year of completion | Health effect | Degree of impact |
|---|---|---|---|
| Vo Ha Dam (Ghana) | 1966 | Schistosomiasis | 70% |
| Sugar estate irrigation (Tanzania) | 1968 | Schistosomiasis | 85% |
| Kainji Dam (Nigeria) | 1969 | Schistosomiasis | 30–70% |
| Aswan High Dam (Egypt, Sudan) | 1969 | Schistosomiasis | 100% |
| Ubolratana Dam (Thailand) | 1970 | Helminths | 52–90% |
| Malumfashi irrigation (Nigeria) | 1978 | Schistosomiasis | 65% |
| Srinagarind Dam (India) | 1978 | Malaria | 25% |
| Gezira irrigation project (Sudan) | 1979 | Schistosomiasis | 70% |
| Diama Dam (Senegal) | | Schistosomiasis | 100% |

(manganese, iron, heavy metals, etc.) from the sediments into the water column. The water layers deprived of oxygen, and enriched by pollutants in the inflow zone, may extend along the reservoir, reaching the offtakes or outflow. The degree of organic enrichment of sediments might be sufficiently high to lead to the evolution of gases like methane and/or hydrogen sulphide, whose smell may be obnoxious. During periods of decreased water levels, the accumulated sediments may produce smells and/or may be rapidly overgrown by plants, which can then decay after being submerged.

*Downstream problems.* During the reservoir aging process, the quality of the outflow water will typically be worse than in following years. The degree of this water quality deterioration depends on the state of the reservoir, and on the hydrodynamic conditions that are decisive for which water layers reach the outflow. Multiple outlets can help improve the downstream water quality during this period, particularly when severe anoxia occur in deeper strata.

### 6.5.2 Environmental Impact Assessment

The method recommended for proper evaluation of the environmental consequences of reservoir construction is the use of Environmental Impact Assessments (EIA) (Canter, 1977). This procedure is now obligatory for all projects supported by the World Bank (World Bank, 1993), and is even subject to law in some countries. The goal of the assessment is to specify all possible negative consequences of the reservoir construction, with respect to the social, physical and biological environment. Different alternatives must be specified and evaluated in order to determine the alternative with the minimum negative impacts. Public involvement based on the availability of relevant information also is typically requested. The immediate effects connected with different aspects of construction, as well as long-term effects, must be analyzed. These effects are not limited only to the immediate reservoir basin, but also its downstream effects and socioeconomic impacts. The list of items to be covered (Table 6.11) is based on experience gained during the construction of huge reservoirs in Africa and Brazil. An example of EIA evaluation for a reservoir, in the form of a questionnaire matrix, is shown in Figure 6.14.

After reservoir completion, post-construction monitoring should continue at least several years. The goal is to ascertain whether or not the results expected on the basis of predictions have actually been achieved.

### 6.5.3 Guidelines for Construction of Reservoirs

The water quality aspects of reservoir construction depend on the planned primary use of the reservoir, which is related to the restrictions for its size, theoretical water retention time, depth and other features, as well as the local possibilities. However, it usually happens that the primary purpose is exceeded beyond the initially-designated primary use and, therefore, the water quality of any constructed reservoir should be optimized. The hydrological and technical issues of reservoir construction and its costs are not discussed here.

Table 6.11. Items to be considered in environmental impact assessments (EIA) associated with reservoir construction (from Straškraba et al., 1993)

- A relocation program for the local displaced population
- Health problems arising out of the reservoir construction (e.g., dispersal of waterborne diseases) and excessively dense population at the construction site
- Loss of edible native species of fish, some of which may be indigenous and unique
- Loss of native terrestrial communities of fauna and flora, especially threatened species
- Loss of local aquatic and terrestrial biodiversity
- Loss of valuable timber resources
- Loss of mature agricultural land, well-cared for generations (e.g., rice paddies)
- Loss of natural wetlands, wetland species and unique land/water ecotones
- Loss of recreational potential
- Loss of cultural heritage
- Loss of water to evaporation
- Excessive human immigration to the reservoir region, with many attendant social, economic and health problems
- Provision of an adequate compensation for loss of agricultural lands and fisheries grounds, as well as compensation for loss of fishing, recreational and subsistence agricultural activities
- The effect of the presence of the reservoir itself upon the local ecology, economy and society
- Degradation of local water quality
- Spread of waterborne diseases and pests
- River bed and coastal erosion
- Damage to the downstream river by the loss of water, loss of fertilizing silt downstream, fish kills, changed temperature regimes and general water quality deterioration (also see Section 6.3)

Rather, only the features of future reservoirs that are optimal from the perspective of drinking water supply are discussed. The strictest criteria are used here, which evidently cannot be applied everywhere. Further, there are uses which cannot easily be combined, and optimization from one perspective cannot simultaneously represent optimal conditions with respect to another usage. Compromise solutions will certainly be necessary. Nevertheless, the overriding goal should be to approach as much as possible the features listed below for any reservoir being constructed.

The reservoir features optimal from a drinking water supply perspective can be summarized as follows:

- Geographical latitudes between 20–35 degrees are mostly characterized by evaporation exceeding precipitation and, therefore, by large water losses. These are typically regions with few natural lakes and a greater demand for reservoir construction. Nevertheless, alternative solutions should be sought.
- In addition to its present state, take into consideration probable future development of the drainage basin. Drainage basin aspects are as important as the site for the reservoir construction itself. Present and future land use is of vital importance.
- Select the least inhabited and industrialized drainage basin possible. The pollution important from a water quality perspective does not include only domestic, organic and toxic pollution, but also the nutrient concentrations, particularly phosphorus, in the river.

| | RESERVOIR | | | | | | HEADWORKS | | | | | | PRIMARY CONDUIT | | | | | | | | | | RIVER DOWNSTREAM | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | -Sedimentation | -Shoreline protection | -Slope stability | -Preparation/cleaning | -Water quality | -Secondary reservoir uses | -Dam or diversion weir & intake | -River diversion for const'n | -Fill/borrow cores | -Spillway or flood protection works | -Fishpass facilities | -River traffic facilities | -Canal | -Pipeline | -Utilization facilities | power station | irrigation | pumped storage | intermediate storage | -Process stations | pump/treatment | power station | -Flow regime | -Channel regime | -Water quality | -Flow passing/energy | -Dissipation facilities |
| **SOCIAL ENVIRONMENT** | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Public participation | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Employment | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Settlement | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Land value | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Existing land uses | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Risks and anxieties | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Personal and social values | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Historical/cultural | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Landscape/visual | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Recreation | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| **PHYSICAL ENVIRONMENT** | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| River regime | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Erosion/land stabiliy | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Sedimentation | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Surface water | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Ground water | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Agricultural soil | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Foundation materials | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Climate/atmosphere | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Nuisance (noise, dust, smell) | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Landform | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| **BIOLOGICAL ENVIRONMENT** | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Forest | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Shrubland | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Grassland | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Herbfield (alpine) | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Send/Shingle/Rock | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Cropland | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Urban land | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Lakes | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Rivers | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Estuaries | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Later-tidal | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Marine | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Wetlands | | | | | | | | | | | | | | | | | | | | | | | | | | | |

Fig. 6.14. Example of a matrix of variables to be taken into consideration for an EIA concerning construction of a new reservoir.

Thus, minimization of agricultural nonpoint source pollution, and efficient purification, including treatment for phosphorus reduction, must be considered. Future development in the drainage basin should be controlled with respect to its water quality consequences.

- In the planning phase, consider restrictions on further increases in pollution and the decrease of existing pollution. The processes of "clean production" (Section 4.2.2) should be considered where appropriate.
- Choose localities at higher elevations if possible. Higher elevations are often less inhabited, less intensively industrial, and utilized less for agriculture. The colder climate is also favorable for enhancing good water quality, since autochthonous-related eutrophi-

cation is less developed. In terms of water quality, several smaller reservoirs located at high elevations are preferable to one large lowland reservoir.

- Forested areas and meadows are preferred over agricultural areas. This is because the fertilization of fields is usually more intensive than the fertilization of forests and meadows, so that the probability of increased eutrophication is diminished (Section 4.2.1).
- If site selection can be made between sites with point and diffuse pollution sources, the former are preferred. Construction of purification facilities for point source pollution purification can be made part of the reservoir construction plan, while reducing pollution from diffuse sources is typically more difficult.
- Select the maximum depth available, avoiding shallow sites. The depth of a reservoir decides on the ratio between the volume of the layers where autochthonous organic matter is produced (trophogenic zone) and where it is reduced (decomposed, mineralized-tropholytic zone). A higher relative volume of the tropholytic zone results in less negative consequences from in-lake decomposition.
- Localities more open to wind effects are preferred. Higher winds cause deeper mixing of the surface water strata. Thus, phytoplankton are more or less homogeneously distributed over the whole water mixing depth. Phytoplankton get less light (i.e., decreased energy) if mixed to deeper depths, and autochthonous organic production is minimized. In tropical regions, a fully-mixed reservoir might be advantageous in preventing the formation of hypolimnetic anoxia.
- Larger lake surface areas are preferable to smaller ones. The wind speed increases over the water surface and larger distances of wind travel on the surface, creating deeper water mixing. Particularly important are the distances in the direction of the most common winds.
- Multiple outlets enable flexible operation and selective hydraulic regulation.
- A very short theoretical water retention time is not favorable. An optimal average theoretical retention time is about 100 days. Reservoirs with short theoretical retention time are subject to large variations in water quality, due to flow-dependent changes in the inflowing river. Reservoirs with long retention times are more susceptible to hypolimnetic anoxia. The higher water supply of reservoirs with average retention times facilitates the efficient utilization of hydraulic regulation methods.
- Locating the multiple offtakes in the lake-like region of a reservoir, but not directly in the dam, allows a more flexible selection of the depths from which raw water is taken, which is useful during periods of reservoir stratification.
- Fisheries management is preferable, but intensive fish cage cultivation will degrade water quality. The relationships between fish composition and water quality (Section 4.3.1) show that balanced fish populations, with predators controlling zooplankton-feeding fish, can significantly decrease the quantities of algae in a reservoir. The possible consequences of sport fisheries that concentrate on predatory fish for the mass reproduction of zooplankton-feeding fish have to be checked.
- Construction of turbine outlets and spillways in a way that maximizes oxygenation of water, which is important for downstream water quality, should be planned.

- The possibility of combining several reservoirs into one system, and selecting the most suitable water quality among them, should be considered. Because a reservoir has a high self-purification capacity (Section 6.4.2), the second or third reservoir in a cascade will likely have better water quality (if not subject to high pollution load from its shores and secondary tributaries). However, a water resource system only provides advantages if the water quality is the dominant, or at least subdominant, aspect.
- Agricultural activities located close to reservoir shores may have more negative effects than when they are located more distant from a reservoir.
- The preservation of forests and wetlands is of primary importance.
- The use of mathematical models enables estimation of future water quality, evaluation of different management options possible and evaluation of construction and site alternatives from a water quality perspective (Chapter 5).

### 6.5.4 *Reservoir Aging*

The water quality of every new reservoir is worse during the filling phase, and in the first years of its existence, than during the following period. A particular characteristic is the anoxia of deeper layers. In instances of heavy organic and nutrient load from the submerged territory and inflow, the anoxia conditions may even reach nearly up to the water surface. This is due to the process called reservoir aging, which describes the rapid changes and deteriorated water quality that occurs during this period. The period of aging is also called "trophic upsurge", because higher biological production takes place during this period. Much slower limnological changes following the period of aging, which may last for decades or centuries, are referred to as reservoir evolution.

A reservoir manager should not be surprised that the predicted water quality for a reservoir is not realized in the first years. These predictions typically are based on stable conditions, in contrast to the unstable conditions that prevail in the first years of a reservoirs' existence. Aging takes place in all reservoirs, but to a different degree and with different speed, depending on the quantity of organic matter and nutrients liberated from the inundated soils and/or brought into the reservoir in its inflows. The water quality consequences of aging will be particularly severe when the organic and nutrient load from the flooded territory is high. For a reservoir with such a heavy external pollutant load that the water quality can be expected to be low even after the aging period, the manager must already expect water quality difficulties. The greatest water quality differences between the aging and post-aging periods are encountered in relatively pristine areas, where water quality predictions for stable (post-aging) conditions are very favorable, but water quality during the aging period can be very low.

Reservoir aging has two primary causes:

- External, physical–chemical,
- Internal, biological.

The *external, physical–chemical causes* are due to the high input of organic matter and nutrients from soil disturbed during construction, and from the decay of flooded terrestrial

vegetation, including trees, shrubs and grasses. The felling of trees, destroying of constructions, building of roads in the river valley to be submerged, construction of roads and bridges along and across the reservoir, rock and soil extraction for the dam construction in the vicinity of the site, and the construction activities of the dam and associated buildings and other structures, are environmental disturbances that lead to the enrichment of the water accumulating in the reservoir. The particular water quality difficulties connected with these activities raise the question of whether or not to clear out all the vegetation on the land surface to be inundated, and to what a degree (summarized in Ploskey, 1983, 1986). The felling of trees in the inundated territory is a common practice for small- and medium-size reservoirs constructed in populated territories. However, it is impossible in the case of large reservoirs in less-inhabited countries. The importance of trees will depend on the reservoir morphology. Trees prevent fisheries in the productive nearshore areas, and endanger the use of nets reaching to greater depths. Thus, at least partial clearing of selected shoreline stretches is recommended when fisheries are of importance. On the other hand, submerged trees represent nursery, feeding, and protection grounds for many species of fish, so that fish density is generally higher among such vegetation. The food supply in such overgrown regions is high, because the associated bacteria and algae with invertebrates feeding on them represent favorable food supplies for certain fish species. The decomposition of wood in the reservoir is very slow, and it takes decades before trees disappear. Moreover, trunks floating on the surface endanger boat traffic. The felling of trees may have negative water quality consequences when done by heavy machinery with the consequent destruction of the soil. During the initial period, more organic materials and nutrients are released into the water from the disturbed soil than from decaying needles or leaves from trees. Shrub clearing is done less frequently, but may be more costly if done carefully without destruction of the soil layers, while drastic grass eradication will only lead to soil disturbance.

The *internal, biological causes* are related to the reconstruction of the biological associations from those present in the rivers to those more suited to the newly-created lake-like conditions (Straškraba et al., 1987). The management consequence is that, in the initial period phytoplankton growth is more intensive not only because of higher nutrient releases, but also because of the absence of proper control of the phytoplankton by invertebrates that feed on it. The reason is the differing rates of reproduction of the small organisms (algae) and larger organisms (zooplankton). It takes a longer time before the more slowly-developing zooplankton reach population densities sufficient to be able to control phytoplankton to levels normal for mature waterbodies. Fish have the slowest development rates, requiring several years to reach maturity. Development of fish populations in a reservoir, starting from a few individuals living in the river, takes more time than the period when the first newborn generation reaches maturity. The first generation is still diluted, and more generations must develop before the new lake is saturated with fish. Further, in temperate regions, fish reproduce only once a year, while the reproduction of smaller zooplankton and phytoplankton is more or less continuous. Strong interrelations between fish, zooplankton, and phytoplankton, and also among different fish species, exist in the aquatic-free water environment. This is due primarily to the trophic feeding relations: Predatory fish feed on

nonpredatory fish species, nonpredatory fish species feed on zooplankton (a few species also on detritus macrophytes and phytoplankton), zooplankton consume phytoplankton. Feeding by all organism groups is selective, with the selection based on the size and taste of the prey organisms. The effect is a top–down control; that is, control by predatory fish, which is propagated by zooplankton-feeding fish and filtering zooplankton down to the phytoplankton. The composition of the pelagic associations of organisms, including the species composition of phytoplankton, depends on the composition of the fish populations. A balanced fish population, consisting of predators controlling the density of nonpredatory fish, is preferred for good water quality, since it leads to the persistence of zooplankton species capable of keeping phytoplankton populations low. When predatory fish are either absent, low or consist only of littoral or benthic species, the nonpredatory fish populations become dominant, eliminating the larger zooplankton species. Larger zooplankton are then substituted for smaller species, being less efficient filterers, and the phytoplankton control does not work. Up to twice much phytoplankton is present when the small zooplankton are numerous and the large zooplankton filterers are rare. The absolute concentrations of phytoplankton depend on the concentrations of available nutrients, light and temperature.

Water quality problems that commonly occur during reservoir aging, and their respective causes, are given in Table 6.12.

Table 6.12. Water quality events and problems that can occur during reservoir aging, and their underlying causes

| Problem | Causes |
|---|---|
| • Increased concentrations of organic matter | Leaching of organic matter from soil; decomposition of drowned vegetation |
| • Increased color | Color indicates resistant organic matter; color changes occur very slowly, and increased values are lasting signs of aging |
| • Low oxygen concentrations (particularly in hypolimnion) | Oxygen is consumed during the decomposition of dissolved and particulate organic matter that enters through the inflows and is released from disturbed soil and decaying vegetation |
| • High nutrient concentrations | Nutrients are leached from disturbed soil |
| • Excessive growth of vegetation (particularly floating aquatics) | New tropical reservoirs are particularly susceptible to this problem |
| • Increased phytoplankton production | Algae grow rapidly as the result of increased nutrient concentrations |
| • Increased fish production | Some species of fish are able to reproduce rapidly in response to the high food supply; nevertheless, some years must pass before the fish population is well established |
| • Increased manganese and iron, remobilization of mercury, emissions of carbon dioxide and methane | While the increase of manganese and iron is common in anoxic hypolimnia, a significant increase of mercury by remineralization from soil is observed rarely; carbon dioxide and methane emissions have mainly been observed in tropical reservoirs |

The duration of the reservoir aging period differs between reservoirs. The average span, however, is between four to more than ten years. In Amazonian reservoirs, stabilization takes a minimum of 10 years, depending upon the rate at which the submerged tropical humic forests decompose. The length and shape of the physical–chemical process of reservoir aging is controlled by the latitude, reservoir volume, water retention time, quantity of organic matter accumulated during filling, activities in the drainage basin, and the quantity of input of suspended materials. With respect to biological processes, the most important element is the rate and degree of the development of fish populations, and the control these organisms exert over the rest of the reservoir ecosystem. The aging process is shorter in reservoirs with rapid water throughflow than in those with slower throughflow. The geographical location (more rapid aging in tropics) and time required for filling (slow and intermittent filling prolongs the aging period) are decisive factors in the aging process. The sequence of events that normally take place during reservoir aging is shown in Figure 6.15.

Most reservoirs stabilize after a few years. Following the aging process, the limnological evolution of the reservoir occurs, and is largely driven by the impact of human activities (e.g., intensification of land use activities and industrial activities).

*Management during aging*

Reservoir aging will usually not create many difficulties in reservoirs for which water quality conditions are not crucial for their functioning (e.g., hydropower generation, flood protection). However, the water quality conditions in the outflowing river may become critical for downstream users and aquatic life, particularly for fishes dependent on reservoir use. The manager of a drinking water reservoir is in a particularly difficult situation, and must cope with this feature to the maximum extent. The manager has essentially three possibilities for reacting to the expected period of water quality deterioration, as follows:

- Start using the reservoir for a drinking water supply only after at least one or two years (depending on the expected deterioration).
- Make provisions for treating water of worse quality during this period than that expected after this period.
- Use appropriate in-lake management options (e.g., mixing of different options).

Knowledge of the aging process, and consideration of its causes as outlined above, will enable a reservoir manager to apply at least some of the following countermeasures, which also will have positive effects on water quality at a later period:

- Any measures decreasing pollution of the major inflows must be done before the filling of the reservoir begins.
- Minimizing soil cover disturbance in the future submerged bottom areas. The disturbance associated with the reservoir construction, including building roads, should as much as possible be located on the downstream side of the dam.
- The vegetation cover of the future reservoir bottom is best left intact, except for at least partial felling of trees and shrubs. The use of heavy machinery that disturbs the soil will significantly increase nutrient leaching from soils.

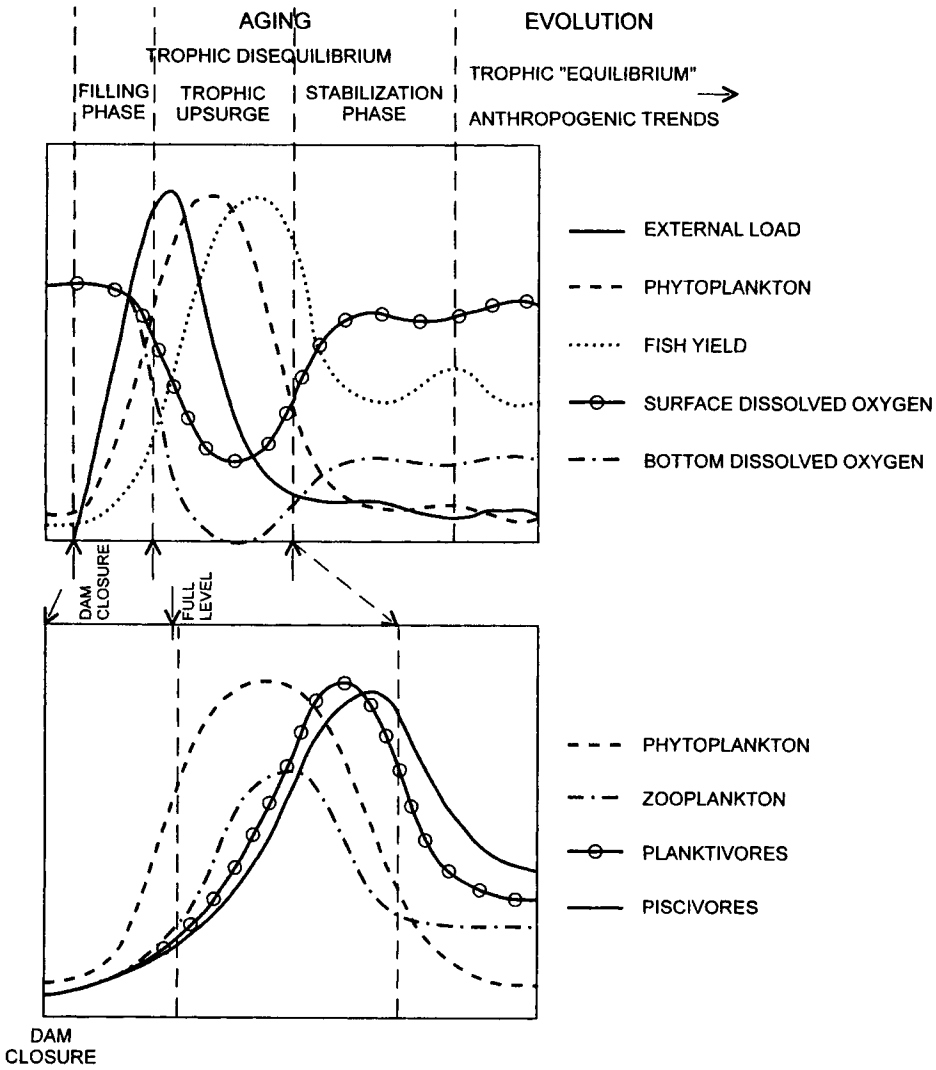



Fig. 6.15. Schematic representation of several observations on reservoir aging ($x$-axis). The succession of peaks of different organisms is shown in a more extended form in the lower part.

- Agricultural activities on the future reservoir bed, and around the future lake perimeter, should be avoided, or the fertilizer and pesticide use minimized, particularly those in the form of powder. No plowing should be done before these areas become submerged.
- Preservation of wetlands and/or simple support of vegetation, particularly marshy ones, along the future perimeter of the reservoir enhances pollution retention.

- Pre-reservoirs, both those to be submerged in the future and those higher on the inflows, constructed before filling begins will help decrease water pollution, particularly the future phosphorus load to the reservoir.
- The introduction of suitable predatory and prey fish during the filling period accelerates the development of balanced fish populations, and shortens the trophic upsurge.

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*Chapter 7*

# INTEGRATED LAKE AND RESERVOIR MANAGEMENT

This chapter is intended for more senior-level reservoir managers and researchers dealing with the integration of water quality management concerns into overall environmental management planning. Some less scientific and technical aspects of these topics, along with some other topics, are discussed in Chapter 8. This chapter focuses on both the near future and the prospective development of reservoir management beyond the present dominant management style based on making independent decisions for each problem and the different aspects of each problem.

A working definition of integrated management is: *The management of a system of functions under single general control in a way that seeks to simultaneously maximize the combined benefits from the individual functions.* From this definition, it follows that the ultimate goal of integrated management is more effective resource management, which considers the simultaneous consideration of formerly separate resource development problems. Integrated management can be regarded as the coordinated control, direction or influence of all human activities in a defined environmental system to achieve and balance the broadest possible range of short- and long-term objectives (Mitchell, 1990; Cairns, 1991; Harper and Fergusson, 1995; Nienhuis et al., 1998; International Environment Technology Center, 1999).

There are many aspects of integrated management. These are divided in this chapter into several topics to enable a structured representation. The goals of integrated management are presented first, followed by suggestions for their practical realization. Several structural topics are then presented, followed by associated political and technical problems.

The importance of clearly setting the goals for management activities is stressed in Section 7.1. Different aspects of the practical realization of the goals stated cannot be underestimated (Section 7.2). Section 7.3 gives an overview of different levels and subjects of integration to be considered, some of which are detailed in the following sections. The subjects internal to water management in the stricter sense are the integration of water quality and water quantity, because water quality is strongly related to water quantity and conflicts can be expected between these two components (Section 7.4). Water uses and management from the perspective of specific uses can conflict with the available natural resources, and integration has to respect both in seeking a compromise solution (Section 7.5).

The subjects more external to the water environment are environment and technology, and partnerships between societal components. Environment technology needs to respect the limitations of the natural environment. Thus, both must find integrative, mutually-friendly ways to interact (Section 7.6). It is important that all community groups, not just managers, are aware of the associated problems, and cooperate on developing joint goals in

partnership (Section 7.7). The particular instruments for achieving integrated management are of both a political and legal character (Section 7.8) and a technical nature (Section 7.9).

## 7.1 GOALS OF INTEGRATED MANAGEMENT

The goals of integrated water quality management are summarized in Table 7.1.

Decision makers and land managers must base their actions to protect aquatic resources on scientifically-sound policy to the maximum extent. It is expected that ecological research and management programs based on partnerships between science, policy and management will contribute to the solution of critical problems.

## 7.2 PRACTICAL REALIZATION OF INTEGRATED MANAGEMENT

As suggested by one experienced manager, the following integrated management guidelines should be followed (Tyson, 1994):

- Development of methodologies for integrated river basin management that include land-use management and water planning,
- Improvement of the volume and accuracy of national and global assessment of water resources,
- Development, promulgation and implementation of new, innovative approaches to water supply and sewage treatment,
- Development, promulgation and implementation of waste minimization and recovery techniques,
- Development of low-tech, low-cost treatment options,
- Increased application of use-related receiving water standards,

Table 7.1. Summary of the goals of integrated water quality management (modified from Lindstrom and Rebescu, 1994)

| |
|---|
| • Making better provision for the conservation, allocation, use and quality of water |
| • Protecting human health |
| • Promoting soil conservation and preventing damage by flood and erosion |
| • Providing and controlling multiple uses of water and drainage of land and integrating them with conservation programs |
| • Ensuring adequate account was taken of the needs of citizens and industry |
| • Enabling water-related recreation |
| • Protecting all aquatic life, fisheries and wildlife |
| • Preserving aesthetic and scenic values |
| • Protecting and recovering of vegetation |
| • Providing adequate monitoring and use of databases, particularly for early warning of difficulties |
| • Ensuring long-term protection of aquatic resources |
| • Achieving cost effectiveness |

- Development and application of economic evaluation tools to both environmental costs and benefits,
- Informing, educating and training professionals and the public.

Howard et al. (1995; also see Rast and Holland, 1988; Rast et al., 1989) have provided direction on how to achieve these general guidelines, as follows:

- Identification of drainage basin wide issues, and short- and long-term development of suitable strategies for the identified issues.
- Prioritization of sub-catchments for action, and identification of areas where maximum benefit can be derived, using criteria such as fitness for use, development status and maintenance of integrity of potable water supply.
- Identification of specific areas and variables of concern, and giving priority to those areas containing the most significant sources of pollution and those that threaten the health of people and communities.
- Prioritizing pollution sources according to the magnitude of their contribution and the possibility of managing the problem, with specific site inspections and meetings to prioritize these sources (this process may be steered by a sub-catchment advisory committee).
- Screening the types of control options and proposing strategies, including control at the sources, control in the delivery process (e.g., wetlands, pollution traps), and control of the water users by provision of alternative water sources.
- Screening practices, and ranking and developing the action plan.
- Implementing actions, under the assumption that the organizations responsible for implementing the various components of the plan are already supportive of the proposals.
- Monitoring the results and revising the strategy as necessary; Water Quality Indices and Biotic Indices are useful management tools and useful tools for conveying this information in a "user-friendly" manner (the experience gained also may be used to revise the action plans and assist with formulation of new strategies).
- Auditing the action plans, since it is essential that implementation of actions is audited to ensure that they take place at the correct place and correct time within the agreed budgets.

## 7.3 LEVELS AND SUBJECTS OF INTEGRATION

As the name suggests, integration is the basic premise of the management approach discussed in this chapter. It is necessary to consider several elements of integration:

- Water systems have space scales. They range from global environment concerns, the country that decides about the economic conditions, laws and general rules of water quality management, through the drainage basin affecting the water quality of lakes and reservoirs, up to our main objects. Integration of these scales is necessary.
- Time scales of management actions span from the immediate hourly and daily regulation actions, through weekly and monthly decisions, up to decisions of an investment character and long-term sustainability planning. Because these decisions are interrelated, integration of time scales is the goal.

- Water quality is highly dependent on water quantity, and management of these two aspects must be integrated.
- Most lakes and reservoirs are now used simultaneously for multiple purposes (navigation, recreation, drinking water supply, hydroelectricity, etc.; see Chapter 2, Section 2.1). Integration of conflicting uses is not a simple task.
- Water represents only one natural resource, and water management presents only one branch of the economy. As an example, industry is dependent on water, and water is used and polluted by industry. Modern agriculture is the largest consumer of water, and agricultural pollution represents a major pollution source. Thus, decisions made in an integrated way help optimize these different aspects of the economy.
- Ecological aspects are increasingly becoming a focus of interest to citizens in developed countries. Thus, merging ecology, economy and technology is an urgent goal of integration.
- Partnerships with the local population, its political representatives, industry and other enterprises, and scientific institutions.
- Finally, from the more theoretical perspective, three basic levels of integration can be distinguished: Normative, strategic and operational. One level cannot be successfully solved without integration with the others.

A typical feature of management, but also of other activities and branches of economy, is their hierarchical character. Hierarchy distinguishes various levels, some overlying and some underlying. In decision-making, hierarchy means not only that the overlying levels coordinate several units at a lower level (called top-down effects). It also means that the lower-lying levels insert an effect (feedback) up the hierarchy (bottom-up effects). In a reservoir system, for example, the decreased water quality in one reservoir must be understood at the level of a reservoir system, as a signal to decrease the water supply from this reservoir and instead supply the population from another reservoir. Due to these two-sided effects, it is not important if we consider the lower or at the upper levels as the basic units. It will depend on our position in the hierarchy: If we are members of a water quality laboratory, the most important will be this level. Our activities will be driven by decisions made at a higher level, but our interests will mainly be in the feedback to the nearest, higher level: Only we can give hints for the need for the laboratory, for new methods to be used, for actions recognized as important from our analyzes. Any feedback can be positive or negative. Positive feedback between levels or elements of a hierarchy means that the development or increase of the element from which the feedback originates supports its further development or increase. In controlled systems, the negative feedback is very important, an example being a drinking water treatment plant, when the filling of a tank leads automatically to a decrease of its inflow.

Interactions in any hierarchy are mutual, and in democratically-organized systems the bottom-up effects are quite strong. The consequence of the interrelations is also a feature called multiple feedback, with the consequence that a change at one decision level may have some unexpected consequences at remote hierarchy levels. The borders of the levels are not impermeable, and the detailed definition of the levels depends on local physical,

economic, social and political conditions. This is related to one characteristic of large systems; namely, their openness to energy, matter and information.

## 7.4 INTEGRATION OF WATER QUALITY AND WATER QUANTITY VARIABLES

The water quality of streams is highly dependent on flow rates, in both a positive and negative way. For diffuse (nonpoint) sources of water pollutants, water quality deteriorates with increasing water flows; for point sources, the opposite is true. In reality, however, there typically is a mixture of different types of pollutant sources, and the measured values of water quality parameters can exhibit a wide range, preventing the recognition of clear trends from either pollutant source.

The picture also is obscured by a hysteresis effect, which is caused by different types of dependency between water quantity (flow rate) and water quality, when the water level is rising at the beginning of a flood, and when the water level is falling after the onset of a flood (Fig. 7.1).

The existing data suggest there is a recent trend to an increasing occurrence of floods. A direct historical observation from Germany indicates a general linear rise in flood peak values, associated with increased imperviousness of the drainage basin area. An increasing flood trend also is evident on the basis of records of damage due to floods in the United States (Naiman et al., 1995). This trend has two underlying reasons:

- The extent of flooding is increasing,
- An increasing number of, and more costly, constructions are situated near rivers, so that the same intensity of floods can produce more damage.

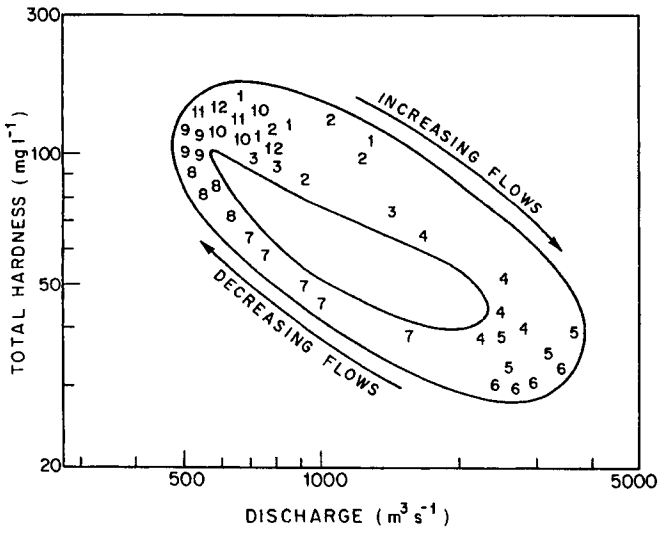



Fig. 7.1. Hysteresis effect of water quality variables during floods.

In regard to the increased density of population and more costly buildings, an increasing number of legislation and other measures are being taken in the United States to prevent construction in areas subject to flooding. Although no direct figures are available, American experts are suggesting that the increasing costs are at least partly compensated for by these measures. The extent of floods seems actually to have risen since 1990. The dominant cause is an improper management approach taken toward water budgets and water retention capacities of the involved areas. The water retention capacities are continuously decreasing due to many factors, including increasing desertification due to deforestation, increasing erosion leading to filling of water retention areas, increased evapotranspiration due to intensive agriculture, drying of wetlands, stream and river channelization, and excessive amelioration. The predominant flood protection measure is to build higher levees often constructed close to the riverbed, which further decreases the area's retention capacity, increasing the extent of downstream flooding. Floodplains capable of retaining large volumes of water, and which reduce water flow velocity, are being lost. Greater extents of impermeable surfaces are being created, preventing the percolation of water into the soil and increasing the surface runoff.

Because of the feedback between increasing channelization and imperviousness and increasing floods, an upward spiral is being created: More channelization produces greater flooding which, in turn, results in increased channelization.

To correct this self-perpetuating cycle, a change in human attitudes regarding flood protection is needed, directed toward maximizing the water retention capacity and perviousness of the surface areas, rather than simply increasing the degree of channelization and the height of levees.

One solution to this problem for cities is the construction of large underground reservoirs. This is being done, for example, in Brazil, whereby the water runoff from some cities is being controlled and the pollution input to rivers is being reduced.

In the above examples, the interests of water quality and water quantity are coherent—the decrease in floods is positive from both aspects. In other instances, however, the interests might be contrary, particularly for reservoir management. Flushing water of poor quality from reservoirs might be in the interest of those responsible for water quality management, while the retention of the water might be the preferred option from the perspective of water quantity. To reduce the reproduction of coarse fish in a reservoir or regulated lake, with the goal of improving water quality (see Section 4.3.1—*Biomanipulation*), a decrease in the water level might be required, whereas those responsible for water quantity might be very interested in retaining water entering the lake during the spring runoff period for later periods of summer drought. Selective water releases from the bottom of a reservoir can be used to rid a waterbody of anoxic hypolimnetic waters, but this also may cause a downstream water quality problem in the form of releasing cold water unsuitable for downstream recreation, as well as losing water storage capacity.

Erosion is an important variable to consider in regard to certain kinds of pollution, particularly phosphorus. Water pollution related to phosphorus is greatly increased as a result of erosion of soil particles onto which phosphorus is bound. This also causes the loss of productive agricultural soil.

Deforestation on a massive scale in certain regions not only creates major changes in water flows, but degrades water quality. It acts on water flows in two ways: It decreases low flows from a given region due to desiccation of the region, thereby decreasing the water retention capacity. It simultaneously increases the high flows, in that most of water falling on the land surface as precipitation flows immediately out of the region, not being retained by the vegetation. Floods increase and water quality deteriorates. Such changes are most dramatically observed in dry regions and less developed countries, where both water shortages and damages due to floods are greatly increasing. However, changes also can be severe even in hydrologically-balanced and well-developed regions, an example being the 1997 and 2002 floods in Central Europe. The water quality decreases during low flows because of a decreased dilution of effluents, and also during floods because of an increased load of suspended materials, the washing out of pollutants, a decay of inflowing organic materials, flooding drinking water sources and deteriorating groundwater.

## 7.5 INTEGRATION OF NATURAL RESOURCES AND VARIOUS WATER USES

Figure 7.2 suggests that water is only one basic resource for human activities and that, simultaneously, all resources comprise various aspects of management. The figure also illustrates that, in this partial matrix of the whole integration problem, two directions of integration can be considered: Integration of different functions or management aspects for each resource separately (Fig. 7.2A), and integration of individual functions across various resources (Fig. 7.2B).

The first approach has dominated until now; there are usually "barriers" of responsibilities between individual resource management sectors. Forestry and agriculture, which

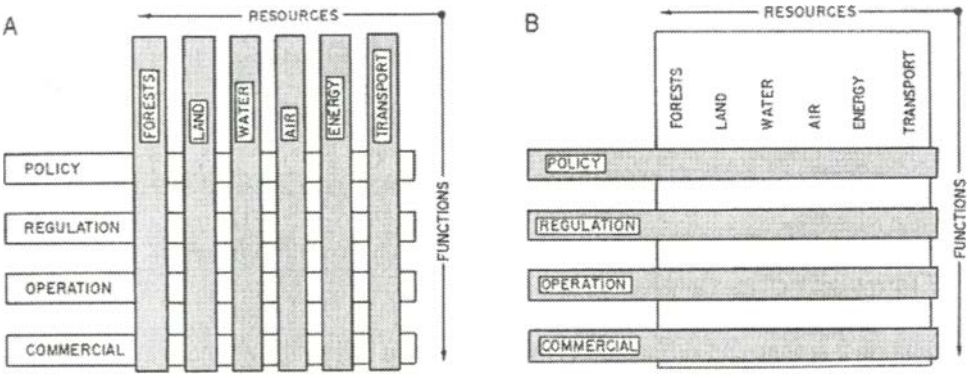



Fig. 7.2. Integrated management of environmental problems. A—Integration of functions for each resource separately. B—Integration of functions across different resources. The approach sub B seems more efficient, but is rarely realized (from Mitchell, 1990).

Table 7.2. Conflicting water uses

| |
|---|
| • Hydroelectricity Production $\Longleftrightarrow$ Drinking Water Supply |
| • Drinking Water Supply $\Longleftrightarrow$ Recreation |
| • Agricultural Irrigation $\Longleftrightarrow$ Industrial Supply |
| • Nuclear Power (Cooling) $\Longleftrightarrow$ Recreation |

have activities tightly coupled to water quality and water quantity concerns, are the responsibility of separate ministries in many countries.

An example of the advantages of the across-resource approach was provided by the recent multimedia approach to radon pollution in the United States. During comparative evaluation of the alternatives of decreasing the exposure of the population to radon, it was recognized that if the U.S. Environmental Protection Agency substituted indoor air radon reductions for reductions of radon concentrations in drinking water, the agency could save 80% of the costs, or nearly five times as many lives (Greenwald, 1977).

There are many conflicting water uses (Table 7.2). Energy generation and industry have the highest water demand. Energy policy, however, often consider water availability only as a constraint, not being concerned about the consequences of the quantities of water evaporated for water quality (increased concentration of pollution due to less dilution, thermal pollution, use of polluting substances in the power plant, etc.).

The water quality consequence of decreased water usage and decreased hydraulic loads of wastewater treatment plants is that less capacity for purification is needed. If wastewater treatment capacity is constructed on the basis of predictions that do not consider savings and decreased water needs, the water released will be of better quality. However, money can be wasted which could otherwise be used for more appropriate water quality improvement measures.

Agriculture is considered one of the most conservative enterprises. Its effects on water resources are enormous, and can take many forms:

- Changing the water budget of an area via the different crops being cultivated,
- Creating conditions for extensive erosion,
- Constructing excessive under-draining systems that dry the area and accelerate the outflow of water,
- Applying unnecessary fertilizers subsequently washed out into streams,
- Converting wetlands into fields,
- Applying toxic chemicals for crop protection,
- Polluting ground and surface water by concentrated cultivation of domestic animals.

And water is not the only part of the environment that is affected. Air can be saturated with some greenhouse gases, and soils can become exhausted, compacted and polluted. Water quality managers must be patient in explaining the consequences of such agricultural practices, which will ultimately damage the farmers themselves.

## 7.6 MERGING ENVIRONMENT AND TECHNOLOGY

As the population density increases, our environment is rapidly deteriorating, with increasing Gross National Products leading to increased production of goods and increasing consumption. The relationships between the degree of population density and the Gross National Product are well documented for a number of compounds. They are clearly evident if one visits some of the most populated regions, for example, and sees not only the heavily-deteriorated environment, but also its consequences for the state of the health and the psychological state of the population.

The science that deals with the interrelations of the nonliving and living (biotic) parts of the environment is ecology. Nowadays we often use the term ecology in a much broader sense than for a scientific discipline alone—we speak of ecology as meaning any degree of care about the environment, sometimes even in a pejorative sense (e.g., care regarding some unimportant, or even dangerous organisms). The human environment, however, is the subject of our interest. Those "unimportant" animals are only a part of this environment in regard to its general state. They are not only important for their beauty and function in nature (mostly not well known), but particularly because they reflect the state of the human environment. If everything around us gets eradicated, and we are the part of this environment, when will the time come that we also start to become eradicated?

In most cases, technology is the main way we produce food, clothes and the many goods that we need. And humans seem to need more and more, meaning more clothing, more cars, more traveling, etc. Unfortunately, the way we use technology is the main cause for global environmental deterioration. During every phase of the technological process, byproducts with negative consequences are created. Pollutants are emitted to the air and water, reaching in this way also to the soil. Wastes are accumulating either immediately during the technological production process, when products are transported and sold to consumers, after they have been unpacked, and after the products lose their functionality.

Merging of these two seemingly-conflicting perspectives is not easy. Although we seek measures for a clean environment, we also perform activities that inevitably lead to pollution (e.g., we need industry and agriculture to feed and clothe ourselves). Requests for recreation are often conflicting: On the one hand, we desire to visit pleasant, quiet environments and, on the other hand, recreation often represents a destructive force to the environment.

Is there a way to cope with the opposing demands of the technology and the environment? In fact, merging of the two can be facilitated by means of ecotechnology (see Chapter 4). Ecotechnology provides us with an approach to decrease the damage due to the environment. We cannot stop creating pollution. It is as unavoidable because dissipation is one of the basic laws in nature; energy and other resource losses accompany every process. We respire and burn part of our food for our life processes, releasing the waste products into the environment in the form of heat, urine and feces. With the consumption of energy in the form of coal, gas or electricity, we convert part of it from high, useful quality to heat as a waste product. Highly-organized organic materials (e.g., the numerous plastic materials) are decomposing into often-toxic rubbish and wastes. These processes are inevitable.

What we can do is to minimize these losses by saving, by using less, by reusing, and by finding ways to produce less wastes. Nature is capable of digesting or otherwise neutralizing certain quantities of waste products, so we can try to maintain and even increase this capacity. Indeed, we can learn from nature in many ways: There are many natural products from the plant and animal kingdom we do know and use, and knowledge of the laws of natural ecosystems enables us to imitate the efficient ways that nature can work. Nature is the only system from which we can learn how to cope with the complexity of interrelations, how to deal with very complex natural systems, and how to optimize the existence of the human race. We can learn intensively from nature. Many new branches of science and technology are created (e.g., cybernetics and its many derivatives, increasing the speed of ships and airplanes, building resistant constructions, producing new medicines). This is *ecotechnology* in the broader sense, directed to these aims; that is, to a merging of the environment with progressive technology.

## 7.7 PARTNERSHIP

What is meant by partnership? The overriding goal here is to join the relevant stakeholders, and harmonize as much as possible the various interests in a drainage basin with respect to its water resources. Based primarily on the program implemented in the Lobo/Broa Reservoir drainage basin in Brazil, the following main roles must be developed to facilitate a long-term program in river basin management (Table 7.3). Building an appropriate integration among users of river basin water resources is fundamental for water quality management. A first component of this partnership is *university and/or local research institutes*. The research system provides the basic information, organizes and improves the data bank, proposes specific monitoring actions, and represents the continuity factor allowing for identification of problems and long-term planning and management.

*Private sector.* The private sector also has a key role in this partnership. It can develop joint ventures and consortia with universities regarding the implementation of new and innovative environmental technologies. It can contribute with funding of specific projects for which incentives by the public sector can be promoted. A recent World Bank study on Environmentally Sustainable Development showed that clean water and sanitation facilities for approximately two billion people are urgently needed (World Bank, 1993). Sustainability criteria for water resource systems have been discussed in detail by a working group of the International Hydrological Program of UNESCO, with contributions from the Task Committee of the Division of Water Resource Planning and Management of the American Society of Civil Engineers, with the results being presented by Loucks and Gladwell.

*Public* and private sectors should be actively engaged in providing "environmental jobs" for the local community (e.g., reforestation, lake shore cleaning, waste recycling). The "water industry" can proportionate several employment opportunities for the local community.

Table 7.3. Main partnership roles for river basin management (from Tundisi and Straškraba, 1995)

| |
|---|
| University: |
| • Theoretical background and data bank |
| • Economic evaluation of ecosystem and sub-system functions and processes |
| • Monitoring perspectives, technological and scientific development |
| • Alternatives for management and ecotechnological approaches |
| • Evaluation of costs of environmental disruption and costs to repair damage |
| Public sector: |
| • Application and decisions on new ideas and methods for environmental planning and management |
| • Legal actions (ombudsman for environment) |
| • Consortia of municipalities and integrated management |
| • Integration with the private sector |
| Private sector: |
| • Technological applications and joint ventures with universities and the public sector |
| • Financing of environmental projects |
| • Participation in wastewater treatment projects, recovery of ecosystems, watershed protection |
| • Consulting (environmental engineering, applications of ecotechnology) |
| • The "water industry" |
| Community participation: |
| • Re-direction of development with a realistic view of the problem and the use of resources |
| • Decodifier of scientific information (e.g., school teachers training programs) |
| • Participating in the use and application of ecotechnology |
| • Participating in monitoring and environmental projects |
| • Initiating legal actions |

*Community participation.* The confrontation between development and conservation is due partly to the excessive centralization of authority, and the attitude of governments toward the use of resources. The local community may have other designs and aspirations for the use of the water resources in a drainage basin. One of the greatest benefits of decentralization is the possibility to include the views of the community in the development of the river basin management strategies and in environmental planning. Thus, community participation has the important objective of critiquing, re-directing and offering alternatives to development plans. Without this participation, any regional development plan with an environmental background will fail, and the term "sustainable development" will only be rhetoric (Biswas, 1988). The effectiveness of the conservation, recovery and planning programs can be significantly increased with public participation. This includes not only the representatives in the public sector, but also direct participation by non-governmental organizations (NGOs) and conservation societies (Tundisi, 1986; Tundisi and Straškraba, 1995).

Development of partnerships in drainage basin management is a long term process that must progress on the basis of two basic assumptions:

- A systemic and articulated approach to the problem, with the drainage basin as the basic management unit (i.e., an ecosystem approach).
- A better quality of life, with sustainable development as the main goal.

The scientific community can take the lead in the process of integrated drainage basin water resource management. To do so, however, the scientific community will have to think and act strategically, which involves six key elements, as follows:

- Any strategy must be guided by a clear vision. In this case, the vision is one of people working together to achieve something that would be impossible to achieve by those same people working alone or in isolation. Intense communication must be part of this vision.
- It must be recognized that strategy describes activities that involve an extended time horizon, including the time it takes to carry out the activities and the time it takes for their impacts to become apparent.
- Although the consequences of pursuing a given strategy may not become apparent for a long time, their eventual impact can be significant.
- An effective strategy usually requires concentrating activity, efforts or attention on a narrow range of pursuits. Focusing on these chosen activities implicitly reduces the resources available for other activities. One of the key areas upon which to concentrate is that of overcoming barriers to communication. This is an area in which chief executives should be most active. These barriers manifest themselves in many forms (e.g., psychological, organizational, technical, disciplinary).
- Most strategies require that a series of certain types of decisions be made over time. These decisions must follow a consistent pattern and be supportive of one another.
- A strategy embraces a wide spectrum of activities, ranging from resource allocation processes to day-to-day operations. The need for consistency over time in these activities also requires that all levels of an organization act to reinforce the strategy. The key is adequate communication (e.g., see World Lake Vision Committe, 2003).

Integration of scientific reasoning into management decisions is made easier by the application of mathematical models, particularly Decision Support Systems (DSS) (see Chapter 5). DDS represent tools that facilitate the making of decisions on the basis of the summarized scientific knowledge. The name stresses that DSS are devoted to *support* decisions, rather than providing them; they provide the manager with the analysis of the problem, using the expert and scientific knowledge condensed into the DSS. For decisions that involve simultaneously considering multiple criteria, the use of multi-criterion optimization is necessary (e.g., Bogardi and Bardossy, 1987).

## 7.8 POLITICAL AND LEGAL ISSUES

Lake and reservoirs are complex ecosystems, and their effective management implies the involvement of many organizations, institutions, NGOs and individual citizens. Thus, an effective integrated lake and reservoir management strategy requires that these entities have a wide understanding of the problems and their implications for society. It does not

imply that the various groups involved in the preservation of lakes and reservoirs agree on all details in the management plans—on the contrary, it is always very fruitful that a wide spectrum of viewpoints are represented in the discussion phase, before a final strategy is selected. However, it is of great importance that all parties accept the dimensions of the problem in space and time, which implies that the entire drainage basin area should be considered, and both the short- and long-term effects discussed.

The complex nature of lake and reservoir management problems often requires a complex set of solutions. There is usually not a simple solution to these problems, but rather a spectrum of cooperative solutions, which can only be found when the spatial dimensions of the problems are accepted. Thus, a wide spectrum of the techniques selected from those presented in Chapter 4, and applied to the entire drainage basin area of a lake or reservoir, will in practice encompass the optimum solution.

Proper management plans that focus on long-term preservation and protection are the most cost-efficient solutions to lake and reservoir environmental problems, since pollution abatement is expensive and many pollutants can accumulate in these aquatic ecosystems. Unfortunately, politicians often have an effective time horizon of only four to six years, because they are typically elected only for terms of this length. In contrast, the perspective of effective lake and reservoir management must be viewed within time intervals that can extend to several decades or more. What is required, therefore, is a profound understanding by politicians and communities of the need to accept a long-term preservation strategy as an alternative to a short term strategy (i.e., just solve the most pressing problems now). The latter strategy is often cheaper on a short-term basis, but ultimately expensive over the long term, and may result in additional obstacles to implementing the best solution.

A common understanding of the core problems, and their significance for society among the politicians and all citizens in a community, is the appropriate basis for making proper environmental decisions. An approach used in Scandinavia, whereby an ombudsman for the environment is nominated, may be a proper idea to adopt in many cases, since it would work to ensure that politicians make decisions on behalf of citizens, rather than on behalf of economic special interests.

Cooperation and understanding are catalysts for the right decisions. However, they also need to work within the framework of clear, detailed and widely-based environmental legislation. Although detailed recommendations on how future environmental legislation should look for developing countries is beyond the scope of this document, a few considerations are discussed in this section, as follows:

- Environmental legislation must consider the specific environment toward which it is aimed. This implies that legislators in developing countries should not imitate the environmental legislation from industrialized countries, because they have a very different environment in the broadest sense of the word. Every country has its own local conditions, which should be considered in the environmental legislation. A minimum standard may be acceptable for all countries. In addition to the minimum requirements, however, such factors as the economic situation of the country, and the geographic, ecological and morphological conditions, should be considered in efforts to intensify the requirements.

- The technological and economic possibilities of the country should be considered in the legislation. Expensive environmental solutions are not necessarily the right answer, for example, for a country for which lower labor costs and more available space suggest the use of soft technology. Many of the techniques mentioned in Chapter 4 offer feasible soft-technological solutions, which could and should be reflected in the environmental legislation.
- Ecological principles should be considered in the legislation, particularly if a wide use of soft technology is foreseen. The legislation that focuses on planning and utilization of natural resources urgently needs ecological considerations to be incorporated, in order to ensure preservation of the environment. Legislation about a buffer zone along rivers and at the shorelines of lakes is an illustrative example of the use of ecological principles in environmental legislation. Legislation on the utilization of recycling principles, and on landscape planning, are other examples in which ecological principles should play an important role.
- The application of green taxes and green auditing also merit mention, since they also should be reflected in environmental legislation. The applied practice of assessing fines for violation of environmental rules/legislation may also play a major role.

Environmental problems are, of course, also complex for politicians. Thus, they should not seek partial solutions, since they are not feasible in such conditions, but rather should utilize the entire spectrum of political instruments to achieve a proper solution to any environmental problem. It implies they should draw properly on the environmental legislation, using it on behalf of the environment, applying green taxes where appropriate, encouraging citizen involvement, and give open information on present and future environmental problems and on the plans for their solution. This is not an easy task. Thus, citizens, the environmental ombudsman and the NGOs of the region must prey relentlessly on the conscience of politicians. The "ping–pong" exchange between politicians and environmentalists is an important democratic process to ensure a proper long-term environmental management, but also requires an open discussion from both sides and a strong involvement on the part of citizens.

The political aspects may be considered an attempt to find an optimal solution or trade-off on the basis of political means. It is an enormous challenge to politicians, particularly when the long-term aspects, as they should be, are included in the considerations. The concept of sustainable development is an attempt to define political goals with respect to all the aspects, and with respect to long-term effects. It is, however, not an easy task to find sustainable solutions to our current problems in concrete situations. With the present population growth in many developing countries, it often is impossible to find solutions for the sustainable development of a region without inclusion of family planning, for example, as a possible tool in environmental management. The need for water supply and safe drinking water, and the impacts on the environment, increases with population density. If the available water resources are not sufficient for the growing population, the only feasible solution is to control the population growth. This is often a very hot political issue in developing countries, but there appears to be no alternative in many of the most threatened regions of the world.

## 7.9 TECHNICAL ISSUES

It has been emphasized many times throughout this book that it is necessary to utilize the entire spectrum of techniques and other solution possibilities, which may be grouped in five classes:

- Environmental technology or "end-of-pipe" technology,
- Ecotechnology,
- Cleaner technology,
- Environmental legislation,
- Proper long-term environmental planning.

The selection of the right combination of methods and approaches also was previously touched upon in this book. A wide spectrum of environmental models (see Chapter 5) are presently available as powerful instruments in environmental planning. The models answer such questions as: Which reduction of the environmental impact on a particular ecosystem is needed to achieve a given environmental quality within a given period of time? The answer is found by application of models to assess the environmental consequences of various environmental strategies, or else the model is used to examine a certain spectrum of scenarios. With the present state-of-the-art of ecological modelling, the inclusion of political, economic and social aspects into models is in its infancy. Thus, it is also necessary to make a political, economic and a social evaluation of the various solutions found with the models.

The technical costs of a given environmental solution are usually not difficult to estimate. For all the techniques mentioned in Chapter 4, the general basis for assessment of costs has been discussed. It is significantly more difficult to assess what society gains with a cleaner lake, higher biodiversity, more bird life, or better water quality. This problem also is discussed further in the next section. It should be mentioned here, however, that ecological–economic models have been developed. In this context, it is relatively easy to assess the direct costs of various solutions and economic consequences, for example, of the quality of water used for drinking water, or the impacts of the water quality on the fishery. It is considerably more difficult to assess the value of a beautiful natural environment. The recreational value of lake water with a high transparency, for example, can be estimated on the basis of different costs of lake shoreline properties on a clean versus pollution lake. It is understandable that very few models have been able to consider the value of the environment. If the very rapid development of models which we have experienced during the last two decades is taken into account, and the emergence of what is called ecological economy, it will likely be possible within the next decade to more widely use ecological–economic models to assess the best environmental strategy, which would include considerations of the value of a beautiful environment.

The inclusion of political and social aspects in modelling exercises is presently almost impossible. Social models are generally much less developed than ecological and economic models (compare with Chapter 5). It does not imply that the social aspects of an environmental strategy cannot be considered, but only that the social consequences have to be considered by other methods in efforts to find a trade-off between the two aspects in

an environmental plan. Usually it is not too difficult, since the environmental and social benefits often work hand in hand, although the optimal conditions for both aspects considered simultaneously is not necessary the optimum solution for the two aspects considered separately. More often, there are conflicts between the economic and social benefits.

Economic development without considerations of the environmental and social implications often has a negative effect on the environmental and social conditions in a given region. Elimination of the negative effects at a later stage may cost more than the economic advantages gained in the first place. Thus, it is crucial to attempt to optimize the economic, social and environmental aspects by the use of the recommendations given in Chapter 10 and throughout this book. Lacking a proper system analytical tool to find the optimal solution or trade-off between the various aspects and problems, proper planning and a good management strategy presently seem to offer the only feasible solution thus far. In this context, the inclusion of social aspects is expected to play a more important role in the development of management strategies in the future.

Attempts have been made to formulate landmarks to be used in a blueprint directed toward sustainable development (e.g., see László et al., 1988; Rast, 2003). The concepts of ecosystem health and ecological indicators represent such attempts, although much wider experience in the use of the concepts in practical environmental management is needed before final recommendations can be given. For example, the extent to which these concepts consider the long term effects and the social-environmental aspects is still unclear.

Environmental managers want to include more ecological and social considerations in their management strategies. Thus, they have asked the following question to ecologists: How can we measure and express that an ecosystem is ecologically sound, and that it has a positive effect on the health of the population? The doctor of medicine attempts to express the health condition of his patient by the use of such indicators as blood pressure, temperature, kidney function, etc. In the same manner, the environmental manager is searching for ecological indicators that can assess the ecosystem health and its impact on the health of the population. Because an ecosystem is a very complex system, it is not surprising that it is not easy to find good ecological indicators to give the appropriate information on ecosystem health, although many ecologists and system ecologists have worked, and are still working, on this problem.

Costanza (1992) and Costanza et al. (1992) summarize the concept of ecosystem health by

- Homeostasis,
- Absence of disease,
- Diversity or complexity,
- Stability or resilience,
- Vigor or scope for growth,
- Balance between system components.

He emphasizes that it is necessary to consider all, or at least most, of the definitions simultaneously. Consequently, he proposes an overall system health index (HI) as:

$$HI = V * O * R,$$

where: $V$ = system vigor,
$O$ = system organization index, and
$R$ = resilience index.

With this proposal, Costanza probably touches on the most crucial ecosystem properties for assessing ecosystem health.

The concepts of exergy and ecological buffer capacities have been proposed as ecological indicators (Jørgensen, 1995a, 1995b). It can be shown that these two concepts combine the above given definitions. However, several other indicators also have recently been proposed in the environmental literature.

The current collection of lake data by the International Lake Environment Committee (ILEC) and the Global Environmental Monitoring System (GEMS) is very important in this context, since an accurate assessment of ecosystem health can only be made on basis of real data. Moreover, a global overview of ecosystem health requires a global monitoring system. The regional efforts in this direction are also important. The main objectives of the European Environmental Agency in Copenhagen, for example, is to provide information on the environmental status in various parts of the European Union.

More experience is urgently needed before the use of ecosystem health and ecological indicators can be generally recommended. Thus, it would be inappropriate to include them as defining elements in the development of management strategies. The results obtained with the use of these concepts, however, look very promising. It may be important to follow the development of these approaches carefully, and incorporate them in development of management strategies in the future when the concepts have more widely demonstrated their applicability. The relationships between ecosystem health in the sense presented above, and the health and the social conditions of the population, have not yet been subjects of research on a sufficient scale. Prevention, rather than cure, however, is widely discussed also in the pathological literature. Environmental management will be able to play a major role, particularly if more research support will be allocated to study the interrelationships between environment, human health, and social problems.

Based on the above discussions, and an optimistic view (which may or may not be realistic), the following trends in environmental management are expected in the future:

- Consideration of the social problems and aspects related to the environmental management of lakes and reservoirs.
- Wider use of integrated models (i.e., models that consider more than one aspect of a problem). Economic–ecological models probably should be in wide use within a decade, while social–economic–ecological models may not be ready to be applied on a large scale within a time horizon of less than 20 years.
- More application of landmark concepts, such as ecosystem health and integrity, when these concepts have shown their applicability on a broader scale.
- A generally more integrated management that simultaneously considers ecosystem health, human health and the social conditions determined by the environment.
- A more open debate between politicians and citizens in their efforts to find proper solutions to environmental issues.

For developing countries, it is particularly recommended that these trends be followed, because they will provide additional opportunities for assessment of sound environmental management. To this end, developing countries also should be prepared to slaughter, if necessary, some "holy cows", a primary example being the introduction of family planning.

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*Chapter 8*

# DEVELOPMENT OF WATER QUALITY MANAGEMENT STRATEGY

The goal of this chapter is to bring together the scientific and technical information previously provided in Chapters 1–7 into a form useful for decision-makers and policy-makers. Although these individuals may be generally familiar with water issues, they do not necessarily possess the scientific background needed to fully address the complex mix of chemical, biological and physical factors to be considered in developing water quality management strategies. Accordingly, this chapter attempts to integrate the major scientific, socioeconomic and political concerns that should be addressed in management strategies for the sustainable use of lakes and reservoirs, for those lacking a scientific background, but who nevertheless have major responsibility for managing lake or reservoir water quality.

## 8.1 INTRODUCTION

That freshwater has a fundamental role in supporting both human survival and facilitating socioeconomic conditions is without question. It is easily observed that people living in areas with scarce freshwater supplies typically exhibit poorer health conditions, more limited economic development, and fewer aquatic ecosystems than those in areas with abundant freshwater supplies. The United Nations reported that, at the present time, approximately one-third of the world's population lives under "water stress" conditions (WMO, 1997). This situation is expected to worsen in future years, with the prediction that two of every three individuals on this planet will live under water stress conditions by the year 2025 *if present water use trends continue*. The dominant role of water pollution in defining the human condition is illustrated by the fact that contaminated water is still the single greatest cause of human illness and death on a global scale. Inadequate treatment of human wastes, and their subsequent discharge to receiving freshwater systems, is the primary culprit in this regard.

Previous chapters of this book deal largely with the scientific and technical aspects of lakes and reservoirs, including the identification of relevant water quality parameters, sampling strategies, the role of models, remedial and control measures and options, etc. As such, they provide information on the biological, chemical and physical characteristics and dynamics of these important waterbodies. However, this type of information does not translate directly or easily into knowledge that can readily be used to manage the water quality of lakes and reservoirs. This is due partly to the fact that fundamental authority for natural resource management (including water resources) typically falls under the responsibility

of decision-makers and policy-makers. Such individuals often lack the scientific and technical background needed to understand and apply this information in a water management context. Thus, the "translation" of scientific and technical knowledge into a form that can be readily used by decision-makers and managers is a fundamental goal for the development and implementation of appropriate water quality management strategies for lakes and reservoirs.

To address this gap, this chapter focuses less on the strictly scientific and technical issues related to lake and reservoir water quality, instead moving more to the management aspects. It attempts to consider both the positive and negative vagaries of the human condition as well. This chapter discusses relevant issues and provides a general framework for developing cost-effective lake and reservoir water quality management strategies. It also attempts to identify specific factors to be considered in tailoring action plans that incorporate the physical, social, institutional, regulatory, economic and political realities for a given lake or reservoir drainage basin.

In striving to achieve this goal, there are several caveats regarding the effective development and implementation of water quality management strategies. First, it is difficult to present a definition of a decision-maker that is accurate in all cases. Depending on the individual situation, they can comprise legislators, executives, directors, senior administrators, etc. Further, water resource concerns and priorities often differ significantly within a given country, as well as on a regional and/or global scale. In many cases, water uses also can be conflicting in nature. The specific circumstances in a given case, therefore, may require a decision-maker to consider or interject issues and concerns other than those discussed in this chapter, to identify and formulate solutions to a given lake or reservoir water quality problem.

The goals of water quality management are similar for virtually all freshwater systems, whether they are on the land surface or under it. As discussed below, surface waters must be managed within the context of a drainage basin, including all the water-related components contained therein. These comprise rivers, streams, lakes, wetlands, ponds, etc. These freshwater systems are generally affected by the same water pollutants and their causative factors, and typically suffer the same general impacts, although the magnitude and timing of the impacts may vary. The goals of this chapter, therefore, overlap with the general goals of water quality management, whether directed to lakes, reservoirs, rivers, wetlands or groundwater aquifers.

There is no universal management program for all lakes and reservoirs, and there is no water quality management strategy or program that will address every situation. As discussed below, it is necessary to relate water management programs to human water needs, as well as the water needs of aquatic ecosystems, which can vary widely within and between countries. Further, effective management strategies must consider not only the strictly scientific, technical and engineering issues related to lakes and reservoirs, but also the relevant nontechnical issues. The latter include such elements as legislation, institutions, economics, social concerns, regulations, politics, etc. Experience around the world has repeatedly demonstrated that these latter elements often are the most important components to consider in developing and implementing effective water resource management

programs of any kind, because they fundamentally influence *how* humans use their available water resources. An adequate water quality management strategy, therefore, must not only contain elements directed to the physical, chemical and biological characteristics of lakes and reservoirs, but also the socioeconomic aspects that control human uses of these waterbodies as well. To facilitate this goal, this chapter also provides some general principles that can be used as guidance for developing lake and reservoir management programs for their sustainable use (see Section 8.3.6).

## 8.2 CONSIDERATIONS IN SELECTING MANAGEMENT STRATEGY

There are several factors to consider in identifying and implementing lake or reservoir water quality management strategies, as described in the following sections.

### *8.2.1 Water and Sustainable Development*

As noted earlier in this book, adequate supplies of freshwater are an essential requirement for human survival and socioeconomic development. The notion of sustainable use of natural resources in relation to socioeconomic development focuses on economic development that is sensitive to the size and condition of the available resource base, including freshwater resources (United Nations, 1992; WMO, 1992; Smith and Rast, 1998; Rast, 2003a). To be sustainable, the type and extent of development must be done within the limits and ability of the natural environment to provide the needed resources. For freshwater resources, sustainable economic development would be based on the ability of the natural environment to renew or sustain the water resources for beneficial human uses over the long term. It would not be possible, for example, to grow water-intensive crops over the long term in arid or semi-arid regions, without extraordinary measures being undertaken to supply the water (building reservoirs, interbasin water transfer, etc.). The notion of sustainability, therefore, should be a cornerstone of effective lake and reservoir management as well (WMO, 1992; Rast, 2003a).

The ultimate goal of water resource management in a given lake or reservoir drainage basin should be to maximize the benefits of the water resources to the basin inhabitants. This is in contrast to dividing the available water resources equally among the basin inhabitants. For example, irrigated agriculture uses the largest volumes of water on a global scale. From a hydrological perspective, therefore, it would be more logical for a nation or region characterized by water scarcity to use its existing resources to produce goods of high economic value, and subsequently trade these goods for food supplies, rather than using their scarce water resources to attempt to grow all their food needs (WMO, 1997).

### *8.2.2 Integrated Water Resource Management*

Sustainable use of lakes and reservoirs requires an integrated management approach. Integration refers to "making a whole from the various parts". The notion of integrated

management within a technical context was introduced in the previous chapter. For the purposes of this chapter, and consistent with the previous discussion on this topic, the integrated management of lake or reservoir drainage basins means consideration and accommodation of not only the scientific and technical aspects (e.g., water supply and demands, geology, physiography, flora, fauna, land use, pollutant sources), but also the socioeconomic aspects (e.g., legislative and regulatory frameworks, institutional capacity, economic conditions, human health and demography, political structure and activities) that fundamentally control how humans use these resources. Further, these elements must be considered within the context of the long-term development and use of the water resources (see Chapter 7 for more details).

In considering management actions, the focus should be on maintaining or achieving the "average" conditions likely to be encountered over the long-term, rather than attempting to provide for the maximum or worst case condition. Human knowledge and understanding of natural cycles and processes is not sufficient for us to be able adequately to consider all the relevant elements necessary for effective management of natural resources. Addressing worst-case conditions in a given situation, for example, would require a more extensive (and more expensive) management program that one developed to address the long-term, average conditions. Further, there is no guarantee that a given management program, no matter how extensive, could adequately address worst case conditions.

### *8.2.3 Watershed Approach to Pollution Control*

The simplest description of a lake is that it is a natural depression in the land surface that became filled with inflowing water over time. In contrast, a reservoir is an artificial lake created by damming a flowing river. Considerable volumes of freshwater, and their biological communities, characterize both natural and artificial lakes. Further, the water quality of these waterbodies is a direct function of the quantity and types of materials entering them from their surrounding drainage basins. The drainage basin is the fundamental freshwater management unit, therefore, for addressing both water quantity and water quality issues. The management of lake and reservoir water quality, as well as their sustainable use, will necessarily focus on the inflowing water and sources of polluting materials entering them from the drainage basin. This means that control measures or programs should be directed to (i) the sources of the polluting materials in the drainage basin, (ii) their transport to lakes and reservoirs and/or (iii) their changes within the waterbody (degradation, transformation, etc.).

A complicating factor with this approach is international basins (i.e., a drainage basin in which the water is used or shared by two or more countries). Nevertheless, a drainage basin approach is equally appropriate in this latter situation. National water needs, priorities and perceptions, however, can differ significantly between the different countries comprising a drainage basin. Further, water use in upstream countries can impact both the quantity and quality of the water reaching the downstream countries in the basin. The water quality of a lake or reservoir that serves as a boundary between multiple countries also can be impacted by all the countries, affecting the quality of the water available for their use.

### 8.2.4 Linkage between Water Quantity and Quality

Water quality and water quantity are fundamentally related. The pollutant load to a lake or reservoir is a function both of the quantities of the polluting materials being carried by the inflowing waters (i.e., the concentration of the polluting materials in the water) and the volume of the inflowing waters carrying these materials (see Section 7.4).

A main impact of water pollution is to limit or otherwise reduce the possible range of human uses of the water without requiring some type or degree of pre-treatment. Water of the highest quality has virtually no restrictions in regard to its uses. Water of drinking water quality, for example, also can be used for virtually any other purpose. As water becomes more polluted, however, it can only be used for fewer purposes without necessitating treatment of some type. Inadequately-treated municipal wastewater, for example, cannot safely be used as a drinking water supply without pre-treatment to remove disease-causing microbes. Under specific conditions, this partially-treated water also could be used to irrigate food crops. Water used for bathing or washing purposes also could be used for this purpose. However, one would not want to use the latter as a drinking water supply unless it was first treated. Similar considerations exist in regard to using storm-generated water runoff from agricultural and urban areas, which can pick up and transport pollutants from the land surface.

Thus, water pollution does not reduce the absolute quantity of water available for human use. Existing technology is adequate to take care of most water pollution problems. However, depending on the types and quantities of the pollutants of concern, such treatment can be both expensive and time-consuming.

### 8.2.5 Water Quality and Competing Water Uses

Lakes and reservoirs are used for a wide range of human water uses (see details in Section 2.1), including:

- Drinking water supply (for humans and livestock),
- Cleaning and bathing,
- Growing and cooking food,
- Fisheries and aquaculture,
- Industrial development (both as a basic commodity of goods and for transporting wastes and by-products from the production arena),
- Bathing and water sports,
- Production of hydropower,
- Transportation of goods, and
- Aesthetics.

Each possible water use has its own water quality requirements. In developing a water quality management strategy, it must be recognized that some water uses often compete with other ones. In developing national water quality standards in the United States, for example, the Environmental Protection Agency was charged with the mandate of achieving "swimmable" and "fishable" waters throughout the country. The water quality needed to

address these two goals, however, is fundamentally different. One would desire water of high quality for swimming purposes. In contrast, the fisheries of a lake or reservoir would be enhanced by increased nutrient loads, which would increase the possibilities of lake or reservoir eutrophication, the latter being contrary to the goal of "swimmable" waters. Thus, the goals of a lake or reservoir water quality management strategy usually should be based on meeting the primary or most sensitive water use(s).

### 8.2.6 *Differing Perspectives of Desirable Water Quality*

There is no universally-accepted perspective for determining "desirable" water quality for lakes and reservoirs. Such factors as the environmental and human conditions in a drainage basin, the existing water sources and the current and anticipated water uses, the current and future economic development plans, the available economic, legal, institutional and political framework, etc., can result in different, and sometimes conflicting, water quality goals for a given waterbody.

This is particularly evident when considering the contrasting water quality situation in developed and developing countries. The inhabitants of lake and reservoir drainage basins in developing countries often face crucial survival issues (Laszlo et al., 1988; Rast, 1999, 2003a). They also suffer from a high level of poverty and other factors limiting their economic development potential. Thus, the daily spectre of simple survival is of high priority. A consequence of extreme poverty is to limit the possibilities of individuals living in a drainage basin. People living in poverty often have few (or no) options to environmental degradation; they do what they must do to survive, regardless of the environmental consequences. This results in continuing, and often increasing, water pollution and associated water-related disease and death. The ability of national governments to adequately address water quality problems in such situations typically is limited by their economic and technological capabilities.

In contrast, inhabitants of the developed world typically do not worry about fundamental survival issues. In fact, even though it typically results in some degree of environmental degradation, a basic level of economic development appears to be necessary to facilitate human consideration of environmental issues (including the quantity and quality of water resources). Thus, many water resources in developed countries are of higher quality than those in developing countries. The latter reflects the increased economic and technological capabilities of developed countries which, in turn, allow for a greater ability to treat and otherwise ensure a higher level of water quality.

The relation of "desirable" water quality, as a function of human water uses, is discussed further in a following section.

## 8.3 DEVELOPING A WATER QUALITY MANAGEMENT STRATEGY

Considering the above-noted issues and concerns, a general approach for developing a realistic water quality management strategy for lakes and reservoirs would comprise the following:

- Identifying the major water quality problem(s), including current water uses and water pollutants of concern, and establishing water management goals (Section 8.3.1).
- Assessing the extent of data and information available about a lake or reservoir and its drainage basin, including sources, locations and characteristics of water quality problems (Section 8.3.2).
- Identifying the available options for addressing water quality problem(s) (Section 8.3.3).
- Analyzing all the relevant costs and expected benefits of alternative management or control options (Section 8.3.4).
- Analyzing the adequacy of existing institutional, legal, regulatory and political framework for implementing alternative management strategies (Section 8.3.5).
- Developing a comprehensive action plan to address the identified water quality problems, and distributing a summary of the plan to all interested parties prior to implementation (Section 8.3.6).
- Ensuring post-treatment monitoring, and providing periodic progress reports on the water quality management control program to citizens and other interested parties (Section 8.3.9).

### *8.3.1 Identifying Water Quality Problem and Establishing Management Goals*

The need for an integrated view of land, atmosphere and water interactions in a drainage basin is essential in both developing and developed countries (Rast, 1999; Falkenmark et al., 1999). This is necessary because of increasingly serious water quality problems resulting from both point and nonpoint source pollutant loads to surface and groundwater. In arid and semi-arid regions, the problems are even more serious, because the absolute lack of water is ultimately both a basic survival and socioeconomic issue (WMO, 1997).

*Identifying water quality management goals*

The designation of good (acceptable) versus bad (unacceptable) water quality should be based on the intended water use(s). Humans can use water of varying quality for different purposes. However, as a lake or reservoir becomes more and more polluted, its possible water uses are reduced, without expensive and time-consuming treatment measures prior to its use. Thus, the water quality management goals for a lake or reservoir should be based on achieving the needed quality for the water uses deemed most important.

The desired water quality relative to the eutrophication status of a lake and reservoir provides an example. Ideally, a lake or reservoir used as a drinking water supply should have water quality as close to an oligotrophic state as possible, thereby ensuring that no or minimal treatment would be necessary to produce water suitable for human consumption. Some water uses may require no treatment at all, regardless of the quality of the water, an example being the transport of commercial goods by ship. In some areas with limited water resources, virtually all available water may be used, regardless of its quality. Thus, although humans can use water exhibiting a range of water quality, there usually is a desired or optimal water quality for virtually every use. A qualitative example of the identification of acceptable and unacceptable water quality applied to eutrophication is given

in Table 3.3 (also see Section 3.2.2). Because information on acceptable versus unacceptable concentrations of other water quality pollutants also is generally available, a similar approach appears feasible for other pollutants as well. The situation will be complicated by the fact that less is known about the ultimate water quality impacts of some other pollutants (e.g., synthetic organic chemicals).

A prudent approach to setting water quality management goals, therefore, is to determine the minimum acceptable water quality for the primary or most sensitive use(s) of the water, and attempt to manage the waterbody to achieve these desired conditions. The situation is more complicated if multiple pollutants must be simultaneously considered. If the existing water quality is not adequate for the intended uses, it will be necessary to develop and implement a remedial program to achieve the desired water quality.

*Who should be involved in addressing the problem?*

The nations of the world comprise a range of governments and socioeconomic conditions. It is difficult, therefore, to provide unequivocal guidelines regarding the role of governments in environmental protection programs that would cover all social, economic and political conditions likely to be encountered. Nevertheless, virtually all governments have some type of civil service infrastructure that could be used to attempt to facilitate governmental activities related to sustainable aquatic resources. Further, many water quality management programs are typically developed and implemented by a governmental body at the local or national level. Thus, all affected government agencies should be consulted in developing lake and reservoir water quality management programs. Relevant agencies include those involved in issues related to environmental quality, water quality, water supply, natural resource management, fisheries, energy production, agriculture, industry, and public health. This inter-governmental consultation also is a good proactive planning strategy, and should facilitate cooperation, rather than confrontation, between governmental units. It also will focus more energy and resources on developing and implementing environmental problems. Ryding and Rast (1989), Smith and Rast (1998) and Thornton et al. (1999) provide examples of specific expertise to consult regarding the management of lakes and reservoirs undergoing varying degrees of eutrophication.

*Citizen involvement*

It is usually always helpful to seek the view of the citizens in lake and reservoir drainage basins regarding water quality problems and their possible solutions. A readily-accessible and "user-friendly" forum to obtain this input should be established to facilitate citizen input (e.g., formation of a citizen's advisory group or committee). Further, the decision-maker typically must balance the interests of advocates of long-term benefits against those wishing more politically-expedient solutions. Citizen inputs can sometimes provide decision-makers with a buffer against the latter solutions.

Knowledge gained from citizen fora also can be disseminated among the general population, facilitating more informed future judgments and actions on its part. Effective public participation, however, requires that government officials be honest in their dissemination of information, as well as responsive to the public views presented to them. Whether true

or perceived, the belief that government officials are not paying appropriate attention to those providing input and advice can seriously damage citizen confidence in a government-sponsored initiative, even if the latter has positive results to offer.

As an illustration of the potential value of citizen participation, many developing countries may be under such financial constraints that the implementation of large structural solutions to water quality problems (e.g., building municipal wastewater treatment plants) is beyond their means. In such situations, governments can strive to make maximum use of community-based public awareness and information programs on water pollution control measures, especially if citizens can directly participate in developing and/or implementing community-based control measures.

### 8.3.2 *Assessing Available Information and Data*

Accurate data and information on the full extent of the problems are essential if one is attempting to address a lake or reservoir water quality monitoring or management program with confidence. Previous studies on similar problems in other lakes and reservoirs, as well as relevant case histories, should be identified and reviewed prior to developing management programs. Relevant information sources include government ministries and agencies, universities, research centers, professional organizations, and scientific and technical literature. The increasing availability of computer-based information sources in many places around the world also should facilitate the identification of such data and information.

Identifying such data and information is necessary for several reasons, including:

- Establishing past and present baseline water quality conditions,
- Identifying significant data and information gaps,
- Facilitating the development of cost-effective monitoring programs.

If the existing data are not adequate to provide the needed information for assessment or management purposes, it may be necessary to establish a drainage basin and/or in-lake monitoring program to obtain the data. However, with careful scrutiny of the data and information needs, and the requirements for obtaining them, an initial monitoring program can be modest in nature, while at the same time allowing for progressive expansion and revision as necessary. Cale and McKown (1986) previously developed a simple methodology for estimating the costs of such monitoring programs, based on the specific data needs, knowledge of the water quality variability in a lake or reservoir, and the costs of individual chemical analyses, relevant personnel and sampling trips.

### 8.3.3 *Identifying and Evaluating Available Options for Addressing Problems*

*Identification of pollutants*

The remedial program or action plan ultimately implemented will be a function of the water quality pollutant or range of pollutants to be addressed, as well as the number and locations of the pollutant sources in the lake or reservoir drainage basin.

Many United Nations organizations (WMO, 1997; UNEP, 1997, 1999) identified inadequately-treated human wastewater as the most serious water-related health problem

faced by humans. The most pervasive water quality problem affecting lakes and reservoirs on a global scale is cultural eutrophication (Rast and Thornton, 1996). Water quality problems resulting from related to excessive concentrations of heavy metals are found in both developed and developing countries, particularly in heavily-industrialized areas. Synthetic organic materials have become an especially serious problem in recent years throughout the world, prominent examples being DDT and other pesticides, PCBs, and some organic solvents. Many are suspected human carcinogens, although our knowledge about the conditions under which they are transported, how they partition themselves into various environmental compartments (e.g., water, soil, atmosphere, living beings), and their short-term and long-term human and environmental impacts, is inadequate in many cases.

For lake and reservoir water quality management programs, the range of pollutants to be considered for most remedial programs includes:

- Particulate matter, including large debris,
- Plant nutrients (phosphorus, nitrogen),
- Biodegradable organic matter (biochemical oxygen demand (BOD), chemical oxygen demand (COD)),
- Heavy metals (lead, mercury, cadmium, chromium, nickel, etc.),
- Synthetic organic chemicals (pesticides, herbicides, organic solvents, etc.),
- Dissolved solids (salinity),
- Biological pollution, particularly by microorganisms,
- Acidifying compounds.

More detailed information on pollution is found in Section 2.2.2.

Large nutrient loads usually are generated from municipal wastewater treatment plants and large stockyards, as well as in storm-generated drainage or runoff from agricultural and urban areas. Agricultural runoff and construction sites, particularly in erosion-prone areas, also are major pollutant sources. Significant heavy metal loads are found in storm-generated runoff from urban and agricultural areas, particularly in industrialized regions. Synthetic organic materials are products of agricultural and urban storm-generated runoff, as well as industrial discharges, and also are transported via the atmosphere.

The locations of the "traditional" pollutants (e.g., nutrients, sediments, microbes) in a drainage basin are a significant factor in selecting a remedial or control strategy. These pollutants are subjected to various biological and chemical reactions and transformations in their transport from their sources to lakes and reservoirs. As a general rule, initial attention should be given to the pollutant sources closest to the location at which the water is to be withdrawn.

This general guideline, however, is not necessarily true for persistent pollutants, or pollutants that are capable of undergoing biotransformation to more toxic forms. The latter would include some heavy metals (e.g., lead, mercury) and some synthetic organic chemicals (PCBs, some pesticides and herbicides). The sources of these pollutants throughout the entire drainage basin must be considered in developing effective water quality management programs in such cases. For synthetic organic chemicals, the very nature of the sampling program must reflect the fact that such pollutants are hydropho-

bic (water-avoiding) in nature. As a result, they typically are present in very low or nondetectable concentrations in the water column. Instead, they tend to concentrate in the fatty tissues of fish and other aquatic organisms and organic-rich sediments in a lake, and must be measured in these components.

An effective remedial program or action plan, therefore, must be developed on the basis of the types, magnitude and sources of pollutants within the entire drainage basin. A general guideline to follow in developing and implementing an effective remedial program is to identify the sources of the greatest quantities generating biologically-available chemical forms of the water pollutants of concern, and to direct major attention to these sources.

*Treating the causes versus the symptoms*

Considerable past experience indicates that it is inevitably more cost-effective over the long term to treat the underlying, and most readily-controlled, *causes* of water quality deterioration, rather than attempting to alleviate the negative *symptoms* of such deterioration. In most cases, this means reduction or elimination of the pollutant inputs causing the degraded water quality in the first place.

The alternative strategy is to treat the symptoms of water quality deterioration. For a lake or reservoir suffering from excessive growths of aquatic plants (macrophytes) as a result of excessive nutrient loads (e.g., phosphorus), for example, one could mechanically harvest or remove the macrophytes from the waterbody. This is in contrast to treating the nutrient sources that are stimulating the macrophyte growths in the first place. In fact, such mechanical harvesting may be the most logical, and perhaps the only option to consider if the costs of reducing the nutrient inputs (the root cause) are too high, or if additional treatment may be necessary in a given case.

Other possible reasons for treating the symptoms of water quality degradation, rather than their underlying causes, include:

- The absence of an institutional or regulatory framework for identifying and treating the causes,
- An inability of government or other bodies to formulate and implement an effective management program directed to reducing pollution inputs, and/or
- Inadequate financial or human resources to carry out the management program (Rast, 1999).

In some cases, direct treatment of water pollutants in a lake or reservoir basin can offer temporary relief from the symptoms of water quality degradation. An example in regard to eutrophication would be the direct addition of chemicals (e.g., alum) to precipitate the nutrients in the lake water, thereby reducing excessive phytoplankton biomass. The direct addition of acid-reducing materials has been attempted in some Scandinavian lakes to reduce the impacts of acid rain on water quality and biological communities. Treating the symptoms, however, rather than the underlying causes or sources of water pollution usually only offers temporary relief. The water quality problems typically will re-occur as additional pollutants enter the waterbody over time.

*Reducing pollutant loads*

The nature of the pollutant sources is a major factor in designing an effective remedial program or action plan for managing a lake or reservoir. A first priority usually is to limit or reduce the pollutant load to a lake or reservoir from the sources in the drainage basin that contribute the greatest quantities of the "biologically-available" forms of the pollutants. This refers to the chemical forms of a pollutant that are readily assimilated or taken up by organisms in a waterbody.

Water pollutants can enter a waterbody from both distinct (point) and diffuse (nonpoint) sources in the drainage basin. Examples of nonpoint pollutant inputs to waterbodies include storm-generated runoff from agricultural or urban areas. Consideration of basic differences in the characteristics of the pollutant inputs from point and nonpoint sources are major factors in designing an effective water-pollution control program for lakes and reservoirs (Table 8.1).

Nonpoint pollutant sources are more difficult to identify and quantify than pollutants from point sources. Further, control of storm-generated runoff typically involves multiple administrative units and departments. In contrast, it usually is easier to reduce point-source pollutant loads by initiating control measures at their sources (Novotny and Olem, 1994; Thornton et al., 1999).

Table 8.1. Comparison of point and nonpoint sources of pollution and their management implications (modified from Thornton et al., 1999)

| Point sources | Nonpoint sources |
|---|---|
| Fairly steady volume and quality | Highly dynamic; occurs at random intervals closely related to hydrologic cycle |
| Variability of values typically is less than one order of magnitude | Variability of values can range across several orders of magnitude |
| Most severe water quality impacts occur during low-flow periods | Most severe water quality impacts typically occur during or after storm events |
| Pollutants enter receiving waters at identifiable points, usually via pipelines or channels | Pollutant entry point to receiving waters cannot easily be identified; sources typically arise from extensive land areas |
| Pollutant loads can be quantified with traditional techniques | Pollutant loads are difficult to quantify with traditional hydrologic techniques |
| Primary water quality parameters of concern include biochemical oxygen demand (BOD), dissolved oxygen, nutrients, suspended solids, and sometimes heavy metals and synthetic organic chemicals | Primary water quality parameters of concern include sediments, nutrients, heavy metals, synthetic organic chemicals, pH (acidity), and dissolved oxygen |
| Pollution control programs are typically applied by government agencies | Effective pollution control programs involve individuals not normally considered in such programs (e.g., farmers, urban homeowners) |

The magnitude of nonpoint-source pollutant loads arising from storm events depends on a number of factors, including:

- The physiography of the drainage basin,
- The soil type and chemistry in the drainage basin,
- The type and extent of vegetative cover,
- The density of drainage channels in the drainage basin,
- The types and quantities of the materials applied to the land surface,
- The duration of the dry period preceding a rainfall event, and
- The volume, intensity and quality of storm-generated runoff.

Using eutrophication as an example, Ryding and Rast (1989) and Thornton et al. (1999) discussed human and animal wastewater, which contains large quantities of biologically-available phosphorus and nitrogen, as primary targets of pollution control programs. Treatment to reduce these nutrients in wastewater is a cost-effective approach to reduce the nutrient loads to lakes and reservoirs, at least up to a certain advanced treatment level, after which some degree of nonpoint-source control may increase the cost efficiency. It is noted, however, that phosphorus and nitrogen are not the only nutrients needed by algae and aquatic plants for growth and reproduction. In specific cases, other elements or compounds may limit algal growth. Examples of other limiting factors controlling the maximum algal biomass in specific cases include silicon, molybdenum, specific vitamins, etc. Nevertheless, from the perspective of development and implementation of cost-effective eutrophication management programs, phosphorus and nitrogen are the most important nutrients, primarily because their input to lakes and reservoirs, particularly from point sources, can be readily controlled with existing technology.

An alternative method of reducing pollutant inputs to a lake or reservoir is to divert the inflowing waters from the drainage basin into a downstream basin. Although this approach often is effective for the lake or reservoir of concern, it will not eliminate the basic problem. It simply transfers it to a downstream waterbody which may (or may not) be more susceptible to the problem.

The possibility of structural versus nonstructural control options also must be considered. Studies in the North American Great Lakes, for example, indicated that the construction of new municipal wastewater treatment plants, or implementing greater levels of phosphorus removal at the existing plants, was a viable option for significantly reducing phosphorus loads to the lakes (PLUARG, 1978). In some cases, however, nonstructural control measures also offered a viable option for controlling nonpoint phosphorus sources in the Great Lakes drainage basin. The PLUARG study, for example, found that simple changes in agricultural practices could significantly reduce phosphorus inputs from agricultural runoff. Such factors as (i) the quantity of fertilizers or pesticides being applied, (ii) the time of their application and (iii) the degree of treatment after their application (e.g., being plowed into the soil versus simply being applied to the soil surface) were of considerable importance in demonstrating good land stewardship (see Section 9.2). It was possible to demonstrate to farmers that plowing manure into the soil as fertilizer, rather than dumping it onto the land surface to be washed into receiving waters in storm-generated runoff, was effective in reducing inputs of nutrients and other pollutants to the Great Lakes.

Further, implementing soil fertility tests not only allowed farmers to reduce their fertilizer applications, and still produce a crop yield similar to applying larger quantities, but also saved money for farmers that previously used excessive quantities of fertilizers.

The PLUARG study also explored the option of removing water pollutants from urban streets by use of sweeping versus vacuum removal. It found that neither approach was successful in removing the smallest-sized particles, which comprise the major source of many pollutants entering lakes and reservoirs. For urban areas reducing the generation of pollutants at the source was found to be an effective approach over the long term. Thus, actions directed to change human practices in rural and urban areas have the potential to significantly reduce the pollutant loads to lakes and reservoirs.

*The option of doing nothing*

The environmental, social and economic consequences of doing nothing also should be considered in evaluating management options. Considering the consequences of doing nothing can actually be useful in helping decide whether or not to implement such a program in the first place. The environmental and socioeconomic consequences of doing nothing offer a basis for comparing the potential impacts of initiating a control program.

Nevertheless, experience to date indicates that a proactive approach to addressing water pollution *before* it becomes a major problem results in greater environmental and socioeconomic dividends over time than attempting to treat a lake or reservoir *after* water pollution has become a major problem. Doing nothing usually is not a satisfactory solution to water quality problems. An inevitable consequence of the human settlement of a drainage basin is a deteriorating water quality over time, especially if the deterioration continues to be ignored. Ignoring lake or reservoir water quality deterioration will likely result in the situation becoming worse over the long term, thereby necessitating even more expensive control programs at a later date. It also will reduce the number of water use options. Further, some prevention or control measures actually may result in monetary savings over time, an example being the use of nutrient-rich waters for irrigation or aquaculture purposes (Ryding and Rast, 1989).

### *8.3.4 Analyzing Costs and Benefits of Alternative Management Options*

The costs and effectiveness of alternative pollution control options can vary significantly. The costs of pollutant removal at treatment plants, for example, vary as a function of the treatment process, the age of the plant, the pollutants to be removed and to what level, etc. Nonpoint-source control measures also exhibit a wide range of costs and effectiveness.

*Compare the available resources and goals*

There is no point in developing a water quality pollution control program if the available resources or administrative structure are inadequate to implement the program. Thus, the available resources should be clearly identified and compared with the needs of the control tasks to be undertaken. These resources include such items as technical expertise, administrative framework, and financial and human resources.

An initial assessment of existing resources also can help identify what new resources are needed before an effective pollution-control program can be implemented. A well-formulated statute or regulation, for example, is useless if not supported by an adequate administrative structure. On the other hand, the results obtained from a well-structured monitoring and assessment program are of no value if the necessary legislative base and/or the institutional framework to ensure that corrective or remedial actions are implemented are not adequate.

Experience suggests that an initial pollution control program should not be unnecessarily elaborate. Rather, it should be sufficiently flexible to have a realistic possibility of achieving its management goals. A small, successful program can be expanded as knowledge and experience is gained, and as new resources are made available, thereby building on success already achieved.

As noted above, if the management goal is to reduce the negative impacts of water quality deterioration, the most cost-effective approach is to reduce the pollutant load from the drainage basin to the lake or reservoir of concern. The control program should be directed to the major sources of the pollutants in the basin.

As previously noted, some land use and/or land management practices offer a nonstructural means of reducing some pollutant loads associated with storm-generated runoff. The implementation of some nonpoint source control methods, however, may require basic public education or changes in public attitudes regarding land use. Without proper public awareness efforts, for example, a farmer located a long distance from a lake or reservoir may not appreciate his/her role in generating pollutant runoff from his land to a waterbody. Thus, the farmer may understandably resist the suggestion that a change in fertilizer or pesticide application patterns, or in the methods of plowing, can help reduce pollutant loads to a lake or reservoir from agricultural runoff.

*The role of cost-benefit analysis*

There are a number of scientific and technical elements to be considered in developing effective lake and reservoir pollution control programs. Nevertheless, experience suggests that the scientist or engineer cannot ignore the economic, institutional or political realities of the program in favor of developing a strictly science-based solution to the problem. Likewise, the decision-maker or policy-maker cannot ignore the environmental realities.

Because of the tremendous growth in the size, complexity and expenditures of governments, the public reaction in many developed countries has been a reluctance to fund new environmental protection programs without first conducting a thorough social and economic analysis of such programs. The basic approach for assessing the desirability or worthiness of alternative management options is cost-benefit analysis, which has its basis in a branch of economic theory called welfare economics. Although a thorough discussion of welfare economics is beyond the scope of this chapter, Baumol and Oates (1975) provide a basic introduction to this topic.

In the broadest sense, cost-benefit analysis refers to a comparison of all the positive and negative elements of a decision, even if all the elements are not directly measurable in strictly monetary terms. In practice, cost-benefit analysis usually means comparing:

- The economic gains to be realized by implementing a particular program or activity, and
- The costs expected to be incurred by implementing the program.

If this type of analysis indicates that a "unit value" (e.g., dollars) required to implement the program is spent such that it generates more wealth than is sacrificed, the overall social welfare is increased. In regard to pollution control, this means that if the costs of a necessary remedial program do not exceed the expected benefits (e.g., enhanced water quality), it is usually desirable to proceed with the program.

Governments must normally choose between many potential projects and programs, including such concerns as national defense, food security, economic development and environmental protection. Using solely economic criteria to assess the worthiness of such programs must stress efficiency in assessing competing alternatives. Thus, the most effective use of scarce resources is required to maximize their benefits.

The economic costs are primary factors in evaluating the efficacy of national, regional and even global programs. A significant shortcoming with a strictly monetary-oriented approach in selecting between alternatives, however, is the assumption that a positive benefit: cost ratio alone is sufficient rationale to justify proceeding with a given program or project. Indeed, it is difficult to offer a good argument contrary to the notion that a cheap or inexpensive solution to a water quality problem is economically desirable. Unfortunately, such an approach also can be environmentally short-sighted because some program elements are not easily quantified, or else can only be quantified in an artificial or unrealistic manner. Examples of the latter include such diverse elements as cultural values, long-term sustainability of natural resources, political realities, and societal and/or government structure and stability. These components can be fundamental considerations to the inhabitants of a lake or reservoir drainage basin when they are deciding how (or whether) to address a given water pollution problem.

A significant step was undertaken by Costanza et al. (1997), in developing estimates of the economic value of life-supporting ecosystem services provided free-of-charge to humans (e.g., water supply, water regulation, waste assimilation, nutrient cycling) from a number of different types of ecosystems (wetlands, oceans, forests, rangelands, etc.) around the world. Such estimates, even with admitted limitations and uncertainties, provided initial information on the economic value of preserving and protecting rivers, lakes, wetlands and other freshwater and marine systems. The "value" of the human services provided in water systems, estimated to be approximately US $33 trillion annually, is especially impressive, particularly because the estimated combined Gross National Products of all the countries in the world at the time of their calculation was approximately US $18 trillion (also see Section 1.2.3).

Governments should proceed cautiously in using a strictly monetary-oriented cost-benefit analysis as the sole decision-making tool. This approach will preclude realistic consideration of the long-term environmental, social and/or public health consequences of a given pollution control program. By comparing the expected benefits of alternative pollution control programs, one can attempt to select the most preferable option either by:

- Comparing the benefit: cost ratios of alternatives programs and levels of expenditure,
- Comparing the absolute values of the expected benefits of alternatives, using a fixed level of monetary and other resources, and
- Determining the minimum cost program for achieving a specific water quality goal or benefit.

Ryding and Rast (1989) describe a simplified approach for conducting a cost-benefit analysis that addresses alternative technical and social concerns regarding the development of eutrophication control measures for lakes and reservoirs.

### 8.3.5 *Analyzing Legislative and Regulatory Frameworks*

*Institutional concerns*

The nature and adequacy of existing institutions dealing with water resources are fundamental concerns. Because a range of different forms of governments, national priorities, customs and socioeconomic frameworks exist around the world, however, guidelines for addressing institutional and regulatory concerns in one country or region may not be appropriate for another country or region. A substantial water quality monitoring and assessment program for identifying pollution and its sources in a drainage basin, for example, would be largely worthless if not accompanied by a regulatory framework for enforcing pollution control programs. Likewise, a fundamentally-sound institutional base will be ineffective in controlling the pollution of a lake or reservoir if it does not have accurate information on the nature, extent and impacts of pollution problems. The latter typically require an adequate monitoring program.

As a practical matter, it usually is more efficient at the central government level to assign environmental protection programs to a single agency structured to manage multiple environmental concerns (e.g., air, water and land resources) than to create a separate government unit to deal with each problem as it arises. The responsible agency or institution for carrying out such programs must be clearly identified, for the benefit of other government agencies and for the citizens in a drainage basin.

In order to avoid overlaps regarding political boundaries, agency mandates, etc., if an agency with a mission compatible with the goals of a pollution control program already exists, it would likely be the most logical one to carry out the new pollution control function. Care must be taken, however, not to assign a new program to an existing government agency with conflicting purposes or goals. If a control program is given to an agency with conflicting purposes, much effort will be wasted in resolving the conflict, rather than addressing the problem, or else one purpose or goal may advance at the expense of the other. The example of the conflicting goals of providing both "swimmable" and "fishable" waters was noted in the previous section.

*Regulatory concerns*

A well-formulated statute or regulation is of little value if the necessary monitoring program or pollution-alert network for determining compliance with it is inadequate or nonexistent. Further, lengthy, complex regulations should be discouraged, because of

the potential difficulty in both understanding and administering them. A proliferation of agency-oriented regulations that can burden many environmental protection programs also should be avoided. If a new statute or regulation cannot be properly implemented or enforced because of unrealistic components or expectations, achievement of the worthwhile objective also is delayed. In such cases, respect for the timely compliance with laws also is generally eroded.

An orderly progression in statutory complexity and development is preferred to the confusion that can result from attempting to accomplish too much, too soon, and with too little information, manpower and funds. As previously noted, initial regulatory statutes should contain provisions for securing sufficient data and information to allow the responsible agency to periodically re-evaluate a given water quality control program, and revise it as necessary over time.

### *8.3.6 Selecting a Water Quality Control Strategy*

There is no universal program to address all lake and reservoir water pollution problems. Our scientific knowledge of the characteristics and dynamics of lakes and reservoirs (particularly cause-effect relations for extreme events) is not yet adequate to be able to develop and implement a flawless water quality control program. Nevertheless, we do have sufficient knowledge and experience to develop a generalized approach which, if used in conjunction with an adequate monitoring program and continuing evaluation of the measured data, will likely work for the majority of cases to be encountered.

The most feasible control program in a given situation will vary from location to location, depending on the nature and magnitude of the problems, the resources available to address it, the management goals, etc. Nevertheless, the reduction of pollutants loads from their sources in a drainage basin represents the most effective, long-term strategy for attempting to control or reduce water quality deterioration of lakes and reservoirs. As noted previously, some in-lake treatment methods also can offer temporary relief from some pollution problems (e.g., nutrients, heavy metals).

*Related water quality and desired water use*

It was previously suggested that "good" (acceptable) or "bad" (unacceptable) water quality must be defined in terms of the degree of impairment of human use of the waterbody. In this way, it is possible to relate desired water uses to the minimally-acceptable and/or optimal water quality for these uses. Thus, a logical management approach is to determine the needed water quality for a given water use or uses, and then design the control program to achieve the needed quality.

This approach is prudent for lakes and reservoirs. However, many reservoirs have dendritic shapes and multiple "arms" with many possible pollutant inputs. In contrast to the bowl-like shape of most natural lakes, reservoirs exhibit a depth gradient from their shallow upstream end to the deeper, more "lake-like" end at the dam site (Thornton et al., 1996; Rast and Straškraba, 2000). As a result, reservoirs often exhibit water quality gradients along their length (Fig. 8.1). Nevertheless, experience from around the world suggests that

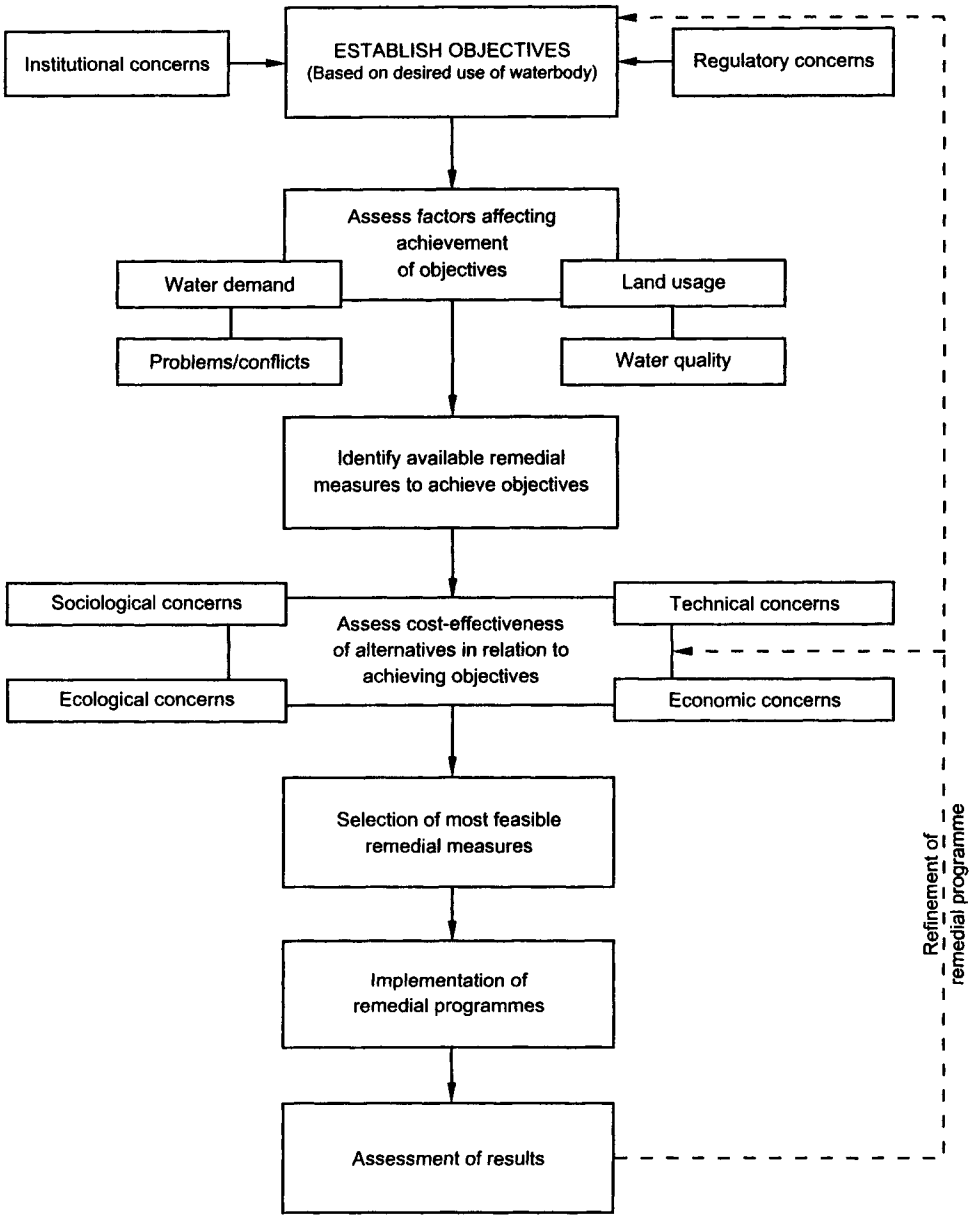



Fig. 8.1. Sequence of relevant decisions to be made by water managers in designing and implementing remedial programs to address water pollution (from Ryding and Rast, 1989).

the primary focus of reservoir water-pollution programs is the deeper, downstream end of the waterbody at the dam. Most water withdrawals are normally taken from this end. Thus, even though the shallower, upstream end of a reservoir may exhibit poorer overall water quality, the quality of the water at the downstream end is the usual focus of water-pollution management programs.

*Selecting a water quality control program*

As previously noted, an individual lake or reservoir water-pollution control program will depend on a number of factors, including the severity of the problem, the pollutants of concern, the number, types and locations of their sources in the drainage basin, the legislative and regulatory framework, the available institutions to carry it out, etc. Thus, it is difficult to provide unequivocal guidelines to be used in all cases. Nevertheless, a generalized sequence of decisions to be made by a decision-maker or water quality manager is illustrated in Figure 8.2. A practical, step-by-step application of the scientific and technical aspects of achieving the desired water quality in lakes and reservoir undergoing cultural eutrophication is provided in Figure 8.2. It is emphasized that virtually every component in this process, even if determined on the basis of the most rigorous scientific standards and criteria, also will necessarily involve the integration of a range of relevant socioeconomic elements as well. This is particularly true in regard to developing an effective water quality management program for the sustainable use of a lake or reservoir. As illustrated in Figure 8.2, these components include determining the nature of the problem, defining the control goals, assessing the adequacy of the current water quality, identifying logical and practical control measures, etc. Greater efforts also may be required to identify safe (acceptable) versus unsafe (unacceptable) pollutant concentrations, and the necessary reduction of the pollutant load needed to achieve desired in-lake conditions.

A similar approach should be feasible for other water pollutants as well, although the elements, particularly for evaluating the pollutant sources and impacts, and the possible control measures, will be unique to the problem.

The final decision on an appropriate water quality control program should be a "multi-judgment" decision, based on the relevant scientific, social, institutional, economic and political aspects of the problem. It is very important to set up a responsible monitoring program as well, both for defining the necessary condition of the waterbody prior to implementing the management program, and for evaluating the final outcome of the program after it is implemented.

Another point is reiterated here; the most prudent approach is to start with a simple management program, and increase its complexity as further knowledge and experience are gained. In this way, one can build on successes and generally reinforce the ultimate goals of the program.

As noted in Chapter 6, although they should never be used as the sole decision-making tool, mathematical models for predicting future trends in lake or reservoir water quality under a changing pattern of pollutant loads can be useful tools in the management and policy-making arena. Such models can be used to predict average values for specific time periods, as well as extreme or worst case conditions in some cases, depending on the

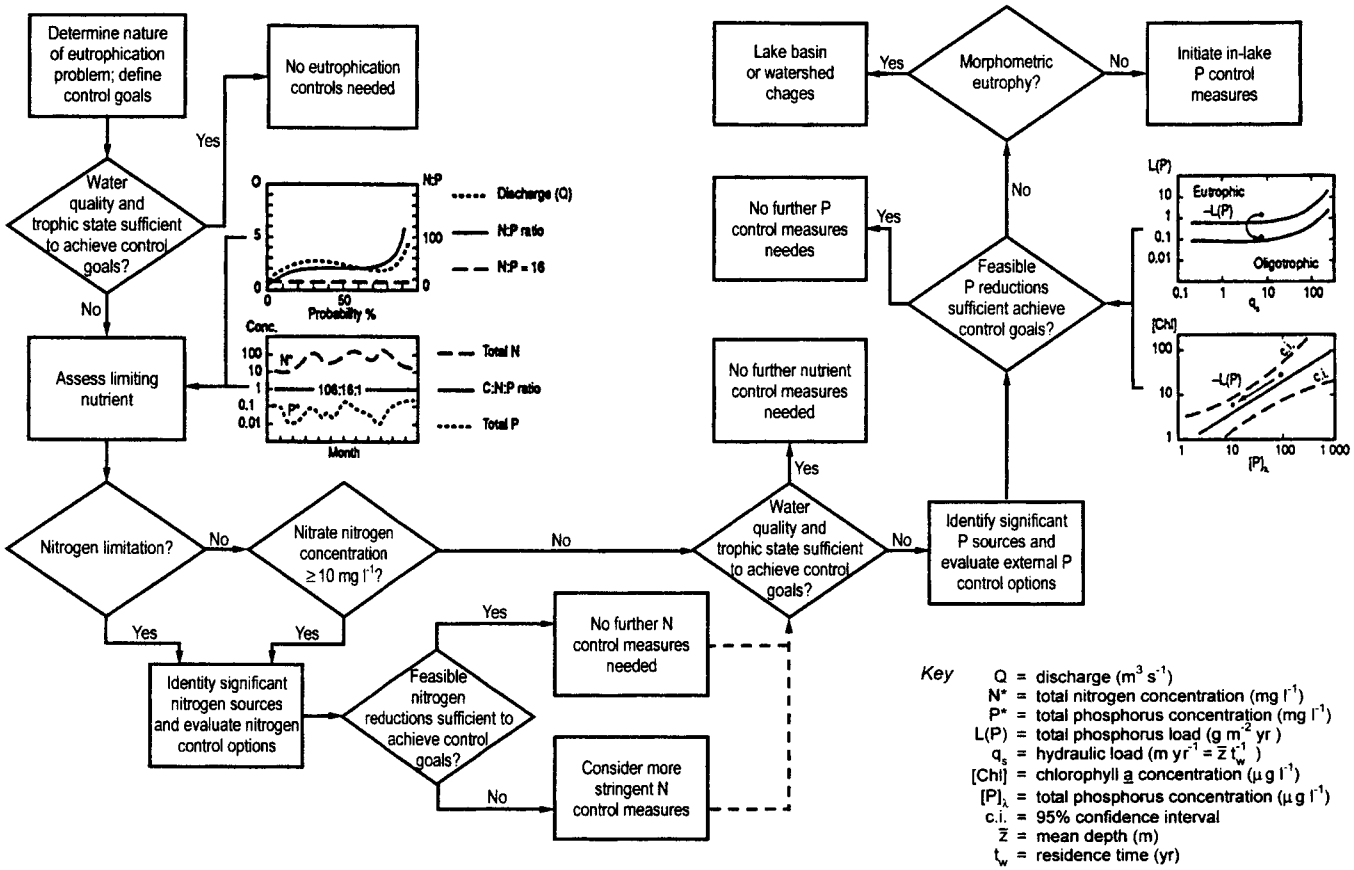



Fig. 8.2. Practical application of a control program for addressing lakes and reservoirs undergoing cultural eutrophication (from Ryding and Rast, 1989).

pollutants of concern. It is also noted that, in most industrialized countries, the main citizen interest in lakes and reservoirs tends to focus on the water quality conditions existing during the period of maximum direct human contact use of the waterbodies and the period of worst water quality. In flowing waters, however, recreational use may be prolonged throughout the year.

The problems to be identified and addressed, and the remedial programs to be developed and implemented, will necessarily dictate the elements that must be considered in developing a comprehensive, cost-effective water quality management program. Nevertheless, a number of principles exist that can be used to guide water quality management programs. These principles were developed as a collaborative effort of an international group of stakeholders concerned with the sustainable use of lake and reservoirs, including representatives of governments, non-governmental organizations, citizens organizations, international scientific organizations, academia and scientific institutions. As identified and discussed in the *World Lake Vision: A Call for Action*, these general principles are as follows (Rast, 2003b):

Principle 1. *A harmonious relationship between humans and nature is essential for the sustainable use of lakes.*

Principle 2. *A lake drainage basin is the logical starting point for planning and management actions for sustainable lake use.*

Principle 3. *A long-term, preventative approach directed to preventing the causes of lake degradation is essential.*

Principle 4. *Policy development and decision making for lake management should be based on sound science and the best available information.*

Principle 5. *The management of lakes for their sustainable use requires the resolution of conflicts among competing users of lake resources, taking into account the needs of present and future generations and of nature.*

Principle 6. *Citizens and other stakeholders should be encouraged to participate meaningfully in identifying and resolving critical lake problems.*

Principle 7. *Good governance, based on fairness, transparency and empowerment of all stakeholders, is essential for sustainable lake use.*

Coupled with consideration of the relevant scientific, socioeconomic and political realities, these principles provide valuable guidance to decision-makers and policy-makers involved in lake and reservoir water quality management.

### *8.3.7 What if a Lake or Reservoir Does Not Respond as Expected*

In some cases, even with the development of a seemingly-comprehensive water quality management program, a lake or reservoir will not respond as expected to the implemented control measures. In these instances, there are several factors that can be examined to determine why the expected results did not materialize.

*Assess the lag period for lake response*

A lake or reservoir will not necessarily respond immediately to a pollution control program, especially those directed to reducing external pollutant loads to the waterbody. Rather,

there typically is a time interval or "lag period" between implementation of a control program and the expected, measurable results. The lag period is the time necessary for the lake or reservoir to flush itself, or otherwise neutralize the effects, of any internal stores of a pollutant, particularly in the sediment at the bottom of the waterbody, once their input has been reduced or eliminated. A prominent example is the phosphorus in the bottom sediments of lakes undergoing cultural eutrophication. If the oxygen in the water layer overlying the bottom of the lake is depleted, the phosphorus in the sediments can be re-mobilized back into the water column. In contrast, in-lake methods, such as harvesting of aquatic weeds and precipitation of nutrients in the waterbody, may show smaller or no lag periods, since the symptoms are being directly treated in these cases. However, for in-lake methods, the negative symptoms may reappear after a short time period.

Citizens should be informed of possible lag periods. Otherwise, they may erroneously conclude that a control program has failed to achieve its pollution control goals, when the reality is that there simply has not been sufficient time after implementation of the program for the positive results to become visible. Sonzogni et al. (1976) and Ryding and Rast (1989) outline methods for calculating the expected duration of the lag period needed to show positive results from reducing the external phosphorus loads to lakes and reservoirs.

*Reassess the pollutant load estimates*

Another possible reason for a lake or reservoir not responding to a pollution control program may be inaccurate estimates of the pollutant load to the waterbody. Further, as noted above, a lake or reservoir can contain significant quantities of a pollutant in its bottom sediments that can be re-mobilized back into the water column under oxygen-depleted conditions in the bottom water layer in the waterbody. These latter sources must be considered in estimating their total load to a lake or reservoir.

As a general rule, direct measurement is the most accurate means of determining the pollutant load to a lake or reservoir. If it is not possible to directly measure the loads, one can estimate the load using unit area loads for specific pollutants and types of land uses in a drainage basin. Unit area loads provide an estimate of the average pollutant load expected to be generated from a given area of land surface over the annual cycle under average hydrologic conditions. As a simple example, a unit area load for a water pollutant might be one kilogram/hectare of agricultural land/year. Multiplying this value by the total number of hectares of agricultural land in a drainage basin provides an estimate of the annual load of the pollutant of concern expected from agricultural areas in the basin. The unit area load, however, must be used with caution, because "wet" or "dry" periods can significantly influence the quantities of a given pollutant generated in a given situation. The time interval between precipitation events also can influence the pollutant load (Thornton et al., 1999).

Although the most accurate values are obtained from direct measurements, the unit load approach also can be used for some point sources. One example is the phosphorus discharge from municipal wastewater treatment plants. Knowledge of the average production of phosphorus from each individual in a given area, and the number of people being served by a treatment plant, can provide an estimate of the total phosphorus load from the

plant. A similar approach could be applied to pollutant discharges from specific industries and manufacturing processes.

The multi-year, international study of nonpoint-source pollution of the North American Great Lakes Basin provides unit area load estimates for a number of water pollutants (PLUARG, 1978) affecting the Great Lakes. Rast and Lee (1983), Ryding and Rast (1989) and Thornton et al. (1999) also discuss methods for estimating the accuracy of phosphorus loads to lakes and reservoirs.

*Reassess the water quality sampling program*

Lakes and reservoirs can exhibit large, intermittent variations in water quality. Thus, an efficient, representative and economically-feasible sampling program is not always readily attainable unless such variations are incorporated into the program. This typically means more frequent sampling during periods of water quality changes, including during and after precipitation events.

Further, accurately forecasting hydrological and meteorological conditions from one year to another, especially because of low flow and flood periods, and for "ice-breaking" periods, is difficult over the long term. There is a need, therefore, for appropriate water quality sampling intervals both in a lake or reservoir, and in the major input streams, over the annual cycle as a baseline monitoring program. As noted above, flood and major rainfall events should always be sampled. A sampling period of at least three years should be considered a minimum sampling effort for the most accurate assessment of a waterbody. To this end, Ryding and Rast (1989), Thornton et al. (1996) and Rast and Thornton (1999) provide guidance for addressing water quality gradients in lakes and reservoirs.

*Reassess the geographic factors*

Physical factors such as mean air temperature, rainfall, snow cover, overland flow and soil erosion can significantly affect the water quality conditions to be found in a lake or reservoir. Further, because of their generally higher average water temperatures, waterbodies in tropical regions can exhibit considerably different chemical conditions and biological communities (also see Section 1.3.2—*Consequences of Geographic Differences for Water Quality Management*).

*Reassess the lake morphometric factors*

The shape or configuration of a lake or reservoir can significantly influence its response to reduced pollutant loads. It is generally difficult, for example, to accurate define the average conditions for reservoirs with substantial longitudinal gradients in water quality (Rast and Straškraba, 2000). As previously noted, the main sampling sites for reservoirs are normally situated at the dam end of the waterbody. In waterbodies containing multiple sub-basins, it is difficult to determine the "average" water quality conditions for the entire waterbody. Further, reservoirs subjected to periodic drawdowns also can experience widely-differing morphometric characteristics over the annual cycle. These factors can result in erroneous conclusions regarding the overall water quality of a lake or reservoir.

*Reassess the hydrodynamic factors*
The mixing regime in a lake or reservoir involves elements such as the wind-induced mixing of the water (and its "fetch") during different seasons, the ratio between the volumes of the bottom and surface water layers (hypolimnion: epilimnion ratio), the ratio between the mixing depth and the mean depth in a waterbody, and the ratio between the shoreline waters and the surface waters (littoral area: epilimnion ratio). These elements also can facilitate the remobilization of pollutants in the bottom sediments back into the water column, particularly with wind-induced water mixing.

*Duration of expected results*
Even when such factors are appropriately considered, a pollution control program still may not achieve the desired results. The only alternative in such cases may be more stringent control measures. Alternatively, one can simply accept the results obtained with the original program as the best that can be obtained for the lake or reservoir with the existing resources and efforts.

Fortunately, however, even when a pollution control program does not produce the desired results within the expected time period, such programs still generally work to the positive benefit of a lake or reservoir. An additional factor to consider in developing a pollution control program, therefore, is the expected duration of the program. Programs designed to be effective over the long term usually are preferable than programs designed only for short-term results.

### *8.3.8 Summarize Selected Control Strategy*

After developing a pollution control program or strategy, it is useful to develop a detailed working plan. The goal is to ensure that regulators, implementers and all other interested parties and individuals have adequate documentation of the program objectives, and the tasks necessary to achieve them. This approach also will foster cooperation, rather than confrontation, between the involved government units, and between government agencies, the citizens and other relevant stakeholders in the drainage basin. The program summary should identify the goals of the control program, and the obligations of the involved governmental agencies and other relevant stakeholders.

A brief control program overview also can be prepared for all interested parties inside and outside the government. It should contain a clear re-statement of the problem to be addressed and the goals of the program, and should be widely disseminated prior to its implementation.

### *8.3.9 Post-Project Monitoring*

It is essential to ensure appropriate monitoring efforts and studies continue for at least several years after implementing a lake or reservoir pollution control program. This will allow comparison of the conditions of a lake or reservoir before and after the implementation of the program.

Post-treatment monitoring also will allow governments, citizens and other interested parties to ascertain whether or not the expected results, often based on model calculations and predictions, have been achieved. Even if the measured results of a pollution control program fall short of the expectations, a post-project monitoring program still can be used to improve the predictions in question, and to reduce the uncertainty of model predictions for future planning purposes.

### *8.3.10 Periodic Progress Reports to the Public and Other Stakeholders*

Wherever feasible, citizen participation in developing and implementing water pollution control programs can be very valuable. Further, providing progress reports to citizens on the results actually achieved for a pollution control program are extremely useful.

The type, extent and format of the information will vary considerably on the basis of the target audience. Appropriate media include the press, television, radio and the Internet, as well as popularized science and technical publications. In view of the lay audience's nontechnical background, general information is usually more informative than highly-technical or scientific descriptions. Appropriately-illustrated information also is useful in communicating to citizens, with technical jargon kept to a minimum. Detailed technical discussions can be presented in appropriate scientific and technical reports and journals.

Experience has generally demonstrated that, if citizens can be persuaded of the severity of a water quality problem, and the environmental, health and economic consequences if the problem is ignored, they can more easily and readily appreciate the need to implement a control program. It also can result in the development of a proprietary interest on the part of citizens in the work involved in the control program. In many cases, it can assist in making citizens more amenable to the associated expenses of the program.

Rast et al. (1990) and Thornton et al. (1999) provide detailed discussions of these various principles as applied to control nutrient loads to lakes and reservoirs undergoing cultural eutrophication. Their approach can be generalized to other pollutants as well.

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*Chapter 9*

# LAKE AND RESERVOIR CASE STUDIES

Many aspects of lake and reservoir management have been discussed in the previous chapters, including many individual management aspects. This chapter adopts another approach: it focuses on an overview of situation and management problems of a few selected lakes. The selection of lakes was made in such a way as to get broad geographical coverage, including lakes and reservoirs from developed and developing countries, from temperate and tropical regions, from shallow to deep waterbodies, and from waterbodies used predominantly for recreation, for drinking water supply and for other purposes. The first example, *Neusiedlersee* at the border between Austria and Hungary, is a very shallow lake largely overgrown by vegetation, with a focus on nature protection, since the lake has been declared a Biosphere Reserve by UNESCO (Section 9.1). Management of the *Laurentian Great Lakes* Basin Ecosystem of North America, including identification and implementation of the appropriate and most cost-effective mix of point and nonpoint source control measures for external phosphorus loads, as confronted by the International Pollution From Activities Reference Group (PLUARG) of the U.S.–Canada International Joint Commission, is discussed in Section 9.2. A major goal of the Great Lakes management effort was to decrease the external phosphorus load to the lakes, which was achieved via a combination of scientific study and analysis, public education and awareness, and concerted binational, state/provincial and local actions. *Lake Fure*, a lake in a highly-urbanized area near Copenhagen, Denmark, is an example of major water quality deterioration and subsequent improvement based on community activities. Restoration difficulties related to the phosphorus accumulated in the sediments over time are now the major threat (Section 9.3). *Lake Ichkeul* is another UNESCO Biosphere Reserve situated near the coast of North of Africa in Tunisia. This shallow lake is endangered by increasing salinity, due to the damming of some of the inflows (Section 9.4). *Biesbosch Reservoirs* in The Netherlands are an example of a system of embankment reservoirs created by surrounding an area by dams and pumping water from the river into the reservoirs. Drinking water supply is the sole use of the reservoirs, with artificial mixing and biomanipulation being the dominant management options (Section 9.5).

The stratified *Římov Reservoir* in the Czech Republic is an example of a small drinking water supply reservoir with multiple outlets, where hydraulic operation and biomanipulation are used in concert with drainage basin protection activities to provide good quality raw water (Section 9.6). In contrast, *Lake Kariba* is one of the largest reservoirs located between Zimbabwe and Zambia in the heart of Africa. The lake was exposed to a long period of continuing invasion of the floating water fern *Salvinia*, which interfered with navigation and fishery. The reservoir is a rare example of successful fish introduction, due to the filling of an empty ecological niche in the open waters of the reservoir (Section 9.7). South

America is represented by a reservoir cascade, the River Tietê Reservoir System in the São Paulo State of Brazil. This system of six reservoirs, with 50 million people living in the headwaters of the river, is an example of a difficult management problem in an area that is rapidly developing economically. The decrease of heavy pollution and eutrophication leading to toxic algal blooms is merging with navigation, tourism, recreation and aquaculture interests (Section 9.8).

## 9.1 NEUSIEDLERSEE–FERTÓ (AUSTRIA AND HUNGARY)

Neusiedlersee is a very shallow, and the largest Austrian lake (Table 9.1), with its southern part extending into Hungary (Fig. 9.1). It occupies a tectonic depression, which came into existence during the end of the Pleistocene, some 12–13,000 years B.P. (Löffler, 1981, 1990). Originally an endorheic area, according to historical and other evidence, it switched to exorheic stages when, during periods of exceptional precipitation, the lake area increased by more than 200 $km^2$ and eventually flowed toward tributaries of the Danube. At least 100 such events may have occurred since the lake came into existence. Moreover, it seems most likely that the main area inundated during such high water periods was the plain, which extends to the southeast portion of the lake—the so-called Hanság, which was obviously the precursor of the present lake. Due to a later subsidence of the latter it turned into a peat accumulating wetland. The bottom of the early lake stage is about 1 m above the deepest parts of the present lake basin.

Since 1965, an Austrian–Hungarian agreement grants a maximum lake level at 115.4 m above sea level (a.s.l.), which is maintained by a recently-improved sluice. In contrast to the control of high water levels, no provision exists for possible desiccation of the lake, such as that which took place in late-1860s, and critical low levels such as occurred in 1902 and 1929/30, when the salinity increased by about 16‰. The latter was due to the Tertiary brackish-water sediment, and was characterized by raised alkalinity values.

This large reed belt (Table 9.1), which represents a remarkable sink of turbidity and nutrients from the open lake, has accumulated about 150 million $m^3$ of sediment since its onset during the 19th Century (Fig. 9.2). It is known as a unique site for a variety of marsh-breeding waterfowl, such as the Spoonbil (*Platalea leucorodia)*, purple heron (*Ardea purpurea)*, Marsh Harriers (*Circus aeruginosus*) and others. Nevertheless, and in spite of an evaluation of the dry lake bottom during the last desiccation (1860s), which proved to be unsuitable for any kind of agriculture actions, it was decided in 1882 to completely drain the lake (Winkler, 1923). The idea subsequently was revised, however, for a number of reasons, such as maintenance of the fishery and a lack of money during the 1890s. After many controversial discussions, it was finally agreed to replace an inefficient drain dating from the 18th Century by a large channel (with a simple sluice for the lake) across the Hanság toward a Danube tributary, in order to control future inundation. This construction began, with several failures, in 1895 and was completed 20 years later. However, only recently (1994) did this artificial outflow and its sluice get their final and functional shape.

Table 9.1. Background data for Neusiedlersee (chemical data—5-year average value)

| | |
|---|---|
| *Physical variables* | |
| Latitude | 47°38′–47°57′ |
| Longitude | 16°41′–16°52′ |
| Altitude | 115.4 m |
| Length | 33.5 km |
| Width | 12 km |
| Volume | 300 million $m^3$ |
| $z$ (mean) | 0.8 m |
| $z_{max}$ | 2.2 m |
| $z_{max}/z$ | 0.36 |
| $A$ | 321 $km^2$ (233 $km^2$ Austria, 88 $km^2$ Hungary, 178 $km^2$ reed belt, 133 $km^2$ open lake) |
| $A'$ | [a]1400 $km^2$ |
| $A'/A$ | 4.4 |
| *Main tributary* | |
| Average inflow | 2.1 $m^3\ sec^{-1}$ |
| Average inflow subsurface | 1.3 $m^3\ sec^{-1}$ |
| *Extent of recreation* | |
| Number of catchment inhabitants | 92,000 |
| Overnight bookings | 1,500,000 |
| Number of sewage treatment plants | 30 |
| Number of sailing boats | ≅5000 |
| *Chemical variables* | |
| Order of major cations | $Na^+ : Mg^{2+} : Ca^{2+} : K^+ = 51:3:7:3$ |
| Order of major anions | $CO_3^{2-} : SO_4^{2-} : Cl^- = 42:40:18$ |
| pH range | 7.4–9.2 |
| Average total P concentration (mg $l^{-1}$) | 50–400 |
| Average total N concentration (mg $l^{-1}$) | 0.5–2.5 |
| *Biological variables* | |
| Dominant phytoplankton | *Microcystis pulverea, Monoraphidium contortum, Cyclotella meneghiniana* |
| Zooplankton | *Leptodora kindti, Arctodiaptomus spinosus, Diaphanosoma brachyurum, Keratella quadrata* |
| Fish | Among fish the cyprinids clearly dominate with about two thirds in Neusiedlersee. For unknown reasons some species such as *Umbra krameri* have dissappeared from the lake, whereas late immigrants like *Pseudoasbora parva* entered the lake about two decades ago |

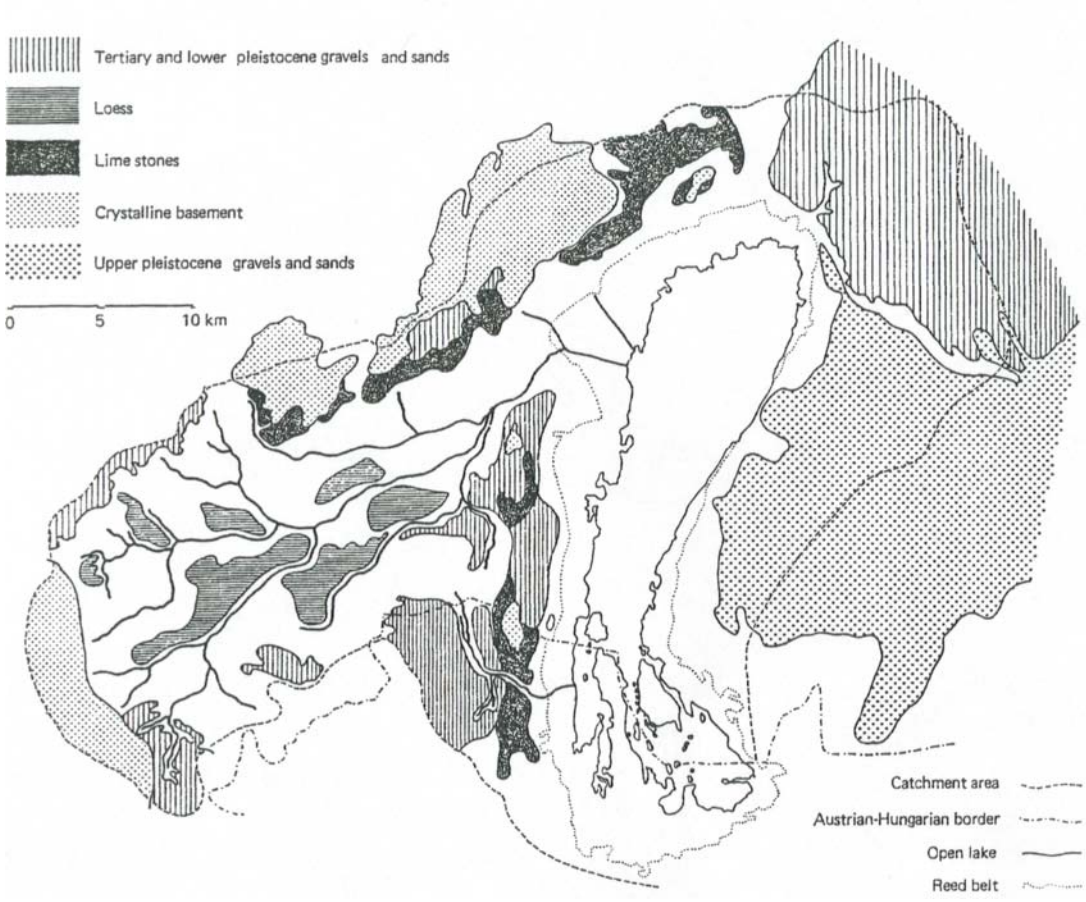



Fig. 9.1. Map of Neusiedlersee, showing extent of macrophyte beds until 1845 and since 1870. The Wulka River flows into the lake from the upper end. (Orig.)

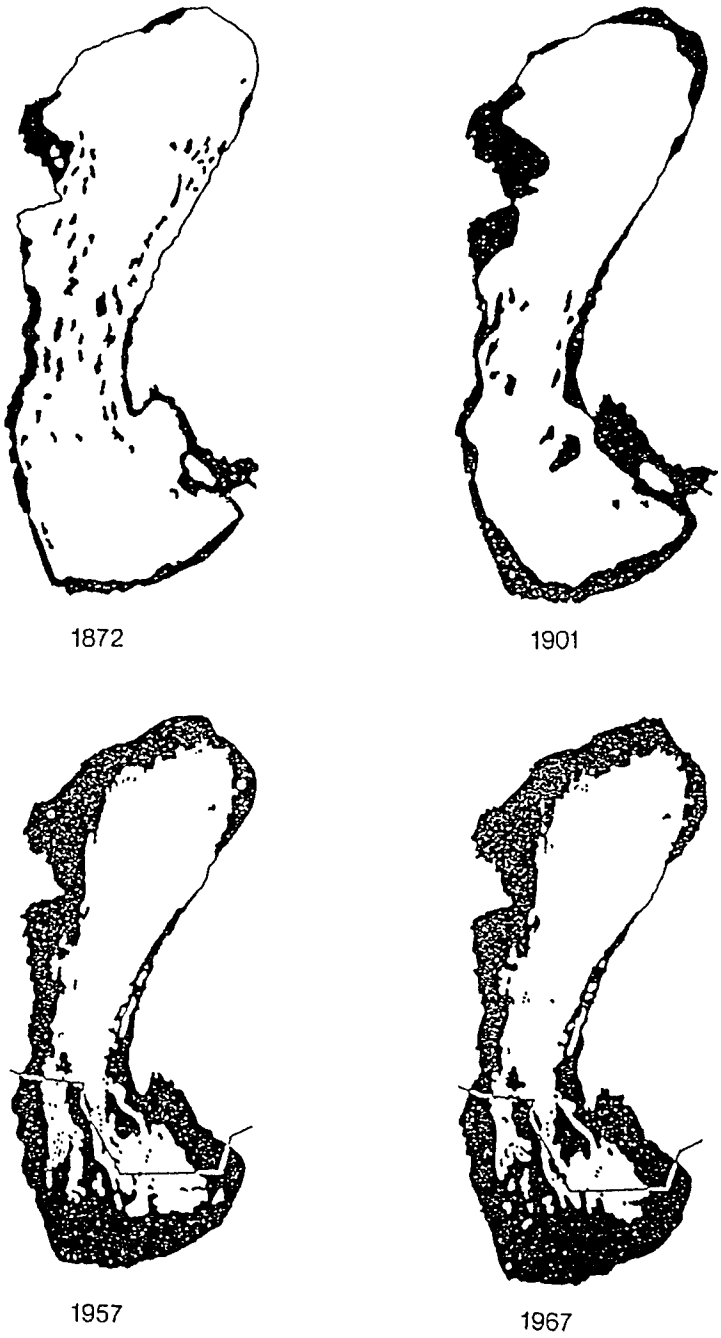



Fig. 9.2. Development of the Neusiedlersee reed belt from 1872 to 1967. (Orig.)

This lake, presently known as one of the most important recreation sites in Central Europe, had its unique flora and fauna doomed to destruction during the 1920s, when Prince Esterhazy in a 1918 letter stated that the lake had no importance and, therefore, should be claimed for agricultural purposes (Winkler, 1923).

Between the World Wars, when the lake became shared between Hungary and Austria, it was gradually developed into a modest tourist site. Reed harvest and fishery increased on a small scale. However, the latter came to an abrupt end when an extremely long freezing period coincided with an anomalous low water level, resulting in an overall fish-kill.

From 1958 on, regular annual stocking of the lake with eel (5–7 million fry individuals) became a major impact which may have contributed to the extinction of several fish species, such as *Umbra krameri, Misgurnus fossilis* and *Proterorhinus marmoratus*. This stocking effort has been given up a few years ago, due mainly to economic reasons, but may be reactivated. Another impact related to the lake was fish—the irresponsible introduction of the grass carp (*Ctenopharyngodon idella*) and silver carp (*Hypophthalmichthys molitrix*) during the 1960s and 1970s. One of the obvious consequences was the destruction of the submerged vegetation (pond weed, *Potamogeton pectinatus,* and milfoil, *Myriophyllum spicatum*). These plants began to recover only after about 20 years, when most of the grass carp became gradually removed by commercial fishing. Tourism in the Neusiedlersee region started to increase exponentially after 1960, and the lake became increasingly eutrophic during the 1970s. The impact included *Microcystis* blooms and a dramatic increase of the planktivorous fish *Pelecus cultratus* which, in contrast to the fishery in Hungary, is not harvested commercially. At the end of the 1970s, as perhaps a consequence of its eutrophication, a raptorial cladoceran species (*Leptodora kindti*) invaded the lake.

Serious consequences from tourism pressures have so far involved the reclamation of remarkable salt meadows northeast Oggau, large sections of the eastern and northern shore and the establishment of a yachting harbor and village within the reed belt near Jois. The transformation of recreation areas into amusement park zones (Podersdorf) became economically supported by the European Union, although in 1977 (Austria) and 1979 (Hungary), UNESCO declared Neusiedlersee a Biosphere-Reserve, followed by a Ramsar Site declaration in 1983 and a nomination as a National Park in 1992.

During the 1980s and early-1990s, the campaign against eutrophication became more successful when most of the relevant villages were gradually supplied with tertiary sewage treatment plants. However, additional activities for the desired improvement of the lake are still needed.

*Management problems*

In our time, the Neusiedlersee area presents one of the most important tourist areas in Austria (sailing, surfing, boating, skating, ice-sailing, sport fishing, hiking, cycling, etc.). At the same time, however, it is one of the unique Biosphere Reserves (more recently also a National Park) valued for its bird life and for peculiar aquatic organisms typical for this alkaline lake, including *Surirella peisonis* (diatom), *Hexarthra jenkinae* (rotifer) and *Diaptomus spinosus* (copepod, endemic in alkaline lakes of the western Palearctic Region). Reed harvest is still of economical, but also ecological importance. The commercial fishery has declined, although the carp, pike and pike-perch catches are still of importance.

Thus, the main conflicts are mainly between nature protection and tourism, including tourism-oriented measures such as boat harbors, and the maintenance of a consistent water level controlled by the sluice of the Hanság Channel. In addition, the lake's main tributary, the Wulka River, which drains almost half of the drainage basin (Fig. 9.1), and which is characterized by large settlements, intensive agriculture and stock-farming, presents one of the main nutrient and pollutant load sources (Löffler and Gunatilaka, 1994).

Another problem is the increasing sedimentation processes within the reed belt area. As previously mentioned, about 150 million $m^3$ of sediment, mainly from the open lake, have accumulated in the reed area since its onset after the last desiccation of the lake some 130 years ago. In contrast to other opinions (Dick et al., 1994), the increase of the reed belt, and its rapidly-growing function as a turbidity sink, was caused by the decline of cattle stock-farming (until the shift of the century by 10,000 heads) and the trampling effects along the lake shores. The present volume of the water-covered reed belt area is fast decreasing. Thus, its potential as a turbidity and nutrient sink will become accordingly restricted. A monitoring system for the regression of the water-covered reed belt area (in connection with a change of the water level) seems most desirable.

Different options for the management of the reed belt including the following:

- Optimizing the Neusiedlersee drainage basin with respect to nutrient and pollutant sources, accepting further sedimentation within the reed belt with possible increasing (willow-) shrub formation and future vegetation stages which may contribute to the diversity of wetland structures.
- Fluctuating water-levels within a permissible extent, thereby allowing for an increased sink function of the reed belt and, with an increased water level during spring, improved spawning conditions for fish (e.g., pike). To some extent this could also contribute to some erosion of the reed belt.
- Apart from strictly-protected reed areas, regular reed harvest would contribute to nutrient removal, and help control reed parasites. In addition, reed removal along the landward fringe would allow for an increase of wet grassland important for many species of water fowl.

All these options must be carefully examined using relevant scientific methods, and also considering the economic resources. One of the more important concerns is the possible desiccation, or at least critically-low water levels, of the lake due to prolonged drought periods, such as happened during the 19th Century and early-20th Century. Although such dramatic events (probably more than 100 desiccations of Neusiedlersee happened since its origin) may be greatly mitigated by the modern sluice construction, undesirable hydrological conditions cannot be excluded. In order to predict and prevent such events in this complex shallow lake system, much more information is needed, which can only be achieved with long-term investigative programs. With regard to the water-budget of Neusiedlersee, high priority should be given to the monitoring of lake. According to earlier sources, such inflows can exceed 8 million $m^3\ yr^{-1}$ (Löffler, 1979). More recently, they have been estimated at less than 2.5 million $m^3\ yr^{-1}$. Since this enormous difference cannot be explained just by improved methodology it must be connected with the dramatic

change of the former pasture-land into farming and particularly viniculture areas, from the 1950s until the 1980s. Obviously, this alternation has resulted in increased evapotranspiration which is intensified by irrigation from local wells. Therefore a long-term monitoring of the groundwater level and possibly also the irrigation activity seems desirable. In addition to this important missing link of the hydrological cycle of Neusiedlersee, realistic values of evapotranspiration based on the present extent of the reed areas are missing. On the one hand, it is because of the increased percentage of the lake area covered by the reed. On the other hand, large (but unknown) portions toward the landward sides of the lake are little, or not at all, involved in the lake's water budget. Thus, studies of the extent of the water-covered reed areas should be carried out, along with denivellation influences by the lake, the latter being completely unknown so far. A network of monitoring stations set up at appropriate sites within the reed area should provide essential information on this topic. Finally, for the optimization of Neusiedlersee, the evaluation of its anion and cation dynamics, essential for the persistence of many organisms, seems indispensable.

An outstanding example of the long-term development of local population awareness is presented by Neusiedlersee, which started with planned destruction, and has presently become a Biosphere Reserve and a National Park. When the lake dried up during the late-1860s, after a period of low precipitation, a commission was established to evaluate the possible use of the area for agriculture. However, due to a rapid increase of the lake level shortly thereafter, this consideration became obsolete. A similar discussion early in the 20th Century, however, again stressed agricultural use of the lake area, calling for draining activities. In 1953, a plan was presented for a dam across the southern portion of the Austrian part of the lake. Arguments put forth for the project, all of which were inadequate, included the improvement of fishery conditions and the stabilization of the lake level. No relevant assessments had previously been made. The financing was to come from the Marshall Aid. For ecological reasons, however, the lake dam project was eliminated from serious consideration in the early-1960s. Instead, the idea of a bridge across the lake, to improve access to the capital of the county Burgenland, was proposed in 1972, eventually resulting in the first large ecological demonstration in Austria.

Along with the "bridge event", awareness of the value of Neusiedlersee and its environment by the public, and more slowly by decision-makers, began to grow, eventually resulting in the declaration of the Austrian part of Neusiedlersee as a Biosphere Reserve in 1977, followed by the Hungarian portion in 1979. Nevertheless, the annual stocking of the lake with the exotic eel, which began during the late-1950s, continued for an additional 15 years, resulting (like the illegal stocking with grass carp during the 1960s) in many ecosystem disturbances, which were only evaluated more than 25 years after the onset of the eel-stocking activities and, thus, could not provide accurate information as to their full impacts. Other adverse activities promoted by decision-makers focused on rapidly-increasing sailing sports. Large areas of the reed belt were sacrificed to construct yachting harbors and holiday settlements, thereby neglecting the recommendations given

for biosphere reserves. Meanwhile, large adverse activities of this kind vanished and, in 1994, the lake became a National Park (in addition to its UNESCO status as a Biosphere Reserve).

The area east of Neusiedlersee, known as "Seewinkel", is covered to a large extent by a great variety of soil types, including large areas of salt soils ("solontchak" and "solonetz") on top of Danube gravel from the late Pleistocene Era. About 120 small and shallow lakes, with a wide range of chemical and physical properties, were still spread throughout this flat country of about 400 $km^2$ at the beginning of the 20th Century. These lakes are not only outstanding with regard to their physiographic properties, a large number being highly alkaline, but also with respect to their flora and fauna. They are of different origin and age, stemming from the late Pleistocene Era and much later. Lack of understanding, and mismanagement, have resulted in a decrease in their number by 50% up to the present time. In most cases, reclamation for agriculture, gravel mining and tourism was responsible for their destruction during the 1950s and 1960s.

Neusiedlersee presents an outstanding example for a long-term development of awareness which at its beginning was oriented toward the destruction and abuse of the lake area (e.g., 1860s and 1880s). Among numerous other lake projects, which fortunately were never realized, the plan of a dam across the lake in order to stabilize the lake level and to "improve" fishery evoked first public protests which finally culminated in a nation-wide demonstration and refusal when the idea of a bridge across the lake "to improve the access to the capital of the county Burgenland" was invented in 1972. Along with a fast growing acceptance of Neusiedlersee as a unique landscape, its protection became increasingly desirable which finally resulted in the Biosphere Reserve, the Ramsar and National Park declaration.

Nevertheless, the pressure of still growing tourism, and due to this development, the economic consequences also are a permanent threat to rules for the conservation of lakes.

## 9.2 THE LAURENTIAN GREAT LAKES (UNITED STATES AND CANADA)

The drainage basin of the Laurentian Great Lakes of North America (Superior, Michigan, Huron, Erie and Ontario) covers an area of 755,200 $km^2$, consisting of 538,900 $km^2$ of land surface and 216,300 $km^2$ of water, and the lakes collectively contain about 18% of the liquid freshwater on the earth's surface. In addition to being the home to more than 37 million people in the United States and Canada, it also has been described as the industrial heartland of both countries.

As a joint international agreement signed by both countries to protect the integrity of the Great Lakes Basin Ecosystem, the 1972 Great Lakes Water Quality Agreement requested the U.S.–Canada International Joint Commission (IJC) to:

- Investigate the extent to which the boundary waters of the Laurentian Great Lakes were being polluted by land drainage (nonpoint source pollution),
- Identify the magnitude and locations of the major nonpoint pollutant sources in the basin, and

Table 9.2. Great Lakes phosphorus loads (PLUARG, 1978)

| Lake | Total load (tons/year) | Atmospheric load (% of total load) | Diffuse load (% of total load) | Estimated contributions of major land uses to diffuse loads (% of diffuse loads) | | |
|---|---|---|---|---|---|---|
| | | | | Agriculture | Urban | Forest/other |
| Superior | 4200 | 37 | 53 | 7 | 7 | 86 |
| Michigan | 6350 | 26 | 30 | 71 | 12 | 17 |
| Huron | 4850 | 23 | 50 | 68 | 12 | 20 |
| Erie | 17,450 | 4 | 48 | 66 | 21 | 13 |
| Ontario | 11,750 | 4 | 28 | 66 | 19 | 15 |

- Make recommendations regarding the most practical remedial measures and probable costs for controlling nonpoint source pollution of the Great Lakes.

To address this complex task, the IJC subsequently established the Pollution from Land Use Activities Reference Group (PLUARG), requesting it to address these questions and to make recommendations for reducing the nonpoint-source phosphorus load to the Great Lakes, in order that phosphorus target loads in the Great Lakes Water Quality Agreement could be achieved.

At the time of the PLUARG study, the USA and Canadian population in the Great Lakes Basin was approximately 30 million and 7 million, respectively. The drainage basin area was approximately 539,000 km$^2$ (Lake Superior—138,600 km$^2$; Lake Michigan—117,400 km$^2$; Lake Huron—128,900 km$^2$; Lake Erie—78,300 km$^2$; Lake Ontario—75,300 km$^2$). Approximately 61% of the whole drainage basin consisted of forested/wooded land, 24% of agricultural land (cropland and pastures), 3% of urban land (residential, industrial and commercial areas), and 12% was recreational land, wetlands, transportation corridors, waste disposal sites, extractive industries and idle lands (Fig. 9.3).

PLUARG found that Lakes Erie and Ontario were most affected by phosphorus, with intensive agricultural activities being the major nonpoint phosphorus source (Table 9.2). Further, the most important land-related factors affecting pollution from given land use activities were soil type, land use intensity and materials usage. Intensive agricultural activities (i.e., growing row crops on fine-textured, high clay content soils) contributed the largest phosphorus loads. The loads did not arise uniformly from whole watersheds or even sub-watersheds. Rather, small portions of a sub-basin ("hydrological active areas") can contribute large pollutant loads, because of their proximity to streams or aquifer-recharge areas. Developed urban areas, for example, have large hydrological active areas because of their highly impervious connected surface areas and extensive alteration of their natural hydrology.

*PLUARG overview model*

To better evaluate the significance of nonpoint-source phosphorus loads, and alternative control options, PLUARG developed an "overview model" (Johnson et al., 1978), with the goal of utilizing its study results in the decision-making process to select specific

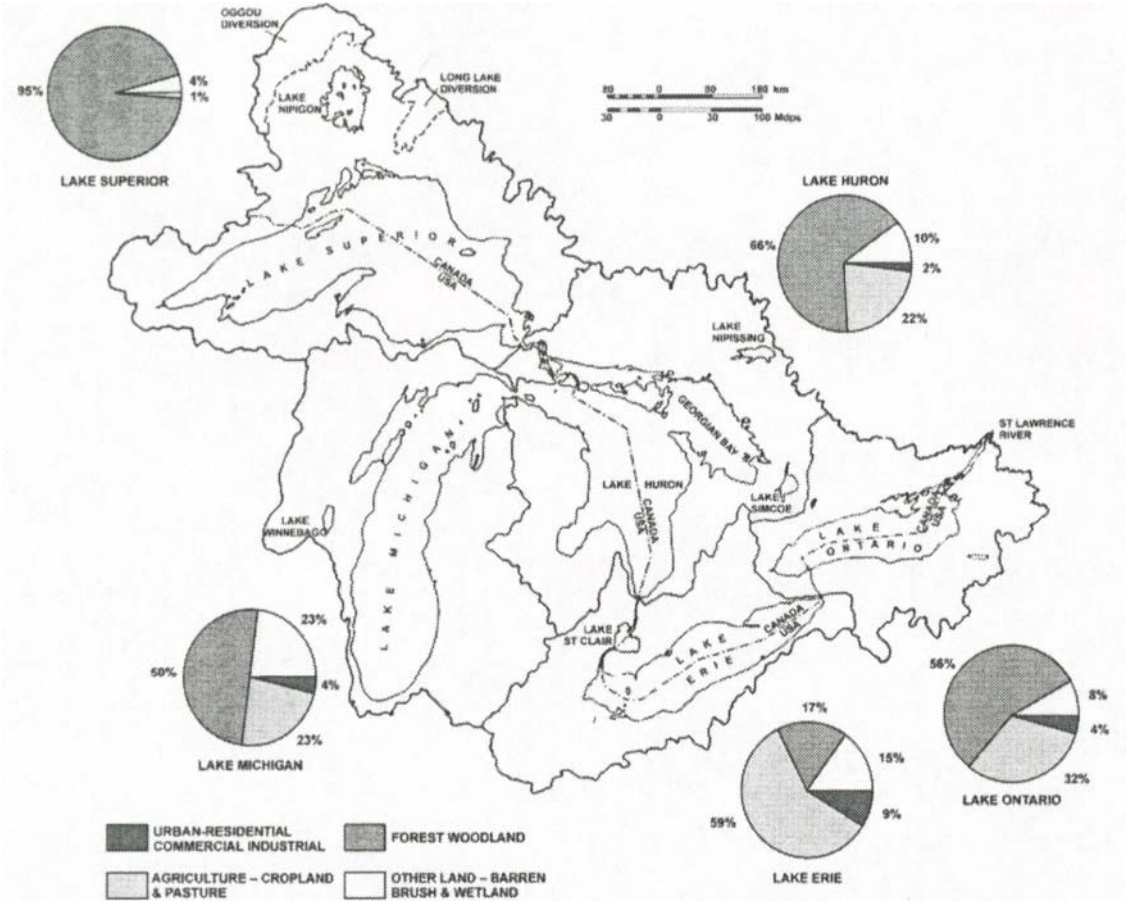



Fig. 9.3. Great Lakes drainage basin and land use percentages for each lake (PLUARG, 1978).

remedial programs for reducing nonpoint phosphorus loads to the Great Lakes. A basic concept was that of "land use" and "land form", and their interaction to determine the nonpoint phosphorus load from a given land use area. Land use is the principal factor affecting diffuse phosphorus loads. Several other factors, however, also directly affect the nonpoint phosphorus loads, including land slope, soil texture and type, extent of impervious surface areas, drainage density and vegetative cover. These latter factors ("land form") represent the interposition of land relief features onto soil texture maps. All other factors being equal, row cropping and fine-textured soils combined to produce the largest nonpoint phosphorus unit-area loads. Thus, a representative phosphorus unit area load from a given sub-basin reflected the predominant land use and land form characteristics of the sub-basin. Using this approach, Johnson et al. (1978) produced a matrix of rural and urban phosphorus unit-area loads for different land use/land form combinations for the Great Lakes Basin.

The overview modeling process comprised the following protocol:

- Determining the phosphorus unit area loads for combinations of land characteristics and land use intensity,
- Delineating the Great Lakes Basin into relatively homogeneous sub-basins, based on land use and intensity and land characteristics,
- Determining municipal point source loads from given basins,
- Compiling base phosphorus loads to the Great Lakes in the absence of remedial programs,
- Simulating remedial measures, and
- Selecting an appropriate mix of point and nonpoint-source remedial measures, based on cost-effectiveness or other desired criteria.

The protocol was a three-step process. The first step was a mathematical description of the Great Lakes drainage basin, dividing its watersheds into relatively homogeneous sub-basins, based on their land use and land form characteristics. The sub-basins were arranged in a cascading sequence, with land drainage flowing from the headwaters, through a series of discrete entry points, to the rivermouth. An example is shown in Figure 9.4, with the watershed treated as ten separate phosphorus-contributing units whose individual outputs were summed.

The second step was development of a matrix of phosphorus unit-area loads, based on land use and land form characteristics. Each sub-basin was described by a pair of numbers defining basic land use and land form characteristics. The two numbers intersected at some point in the unit-area load matrix to produce a discrete unit-area load for the sub-basin as shown for the U.S. portion of Lake Erie basin (Figs 9.5 and 9.6).

The final step was to manipulate the matrices by various remedial program functions iteratively to produce annual and projected demographic, land use and phosphorus load data for given phosphorus control scenarios. This allowed prediction of expected phosphorus load reductions and estimated costs for alternative control scenarios, including the effects of demographic and land use pattern changes over time. The percentage phosphorus loads from major land use categories are summarized in Table 9.3.

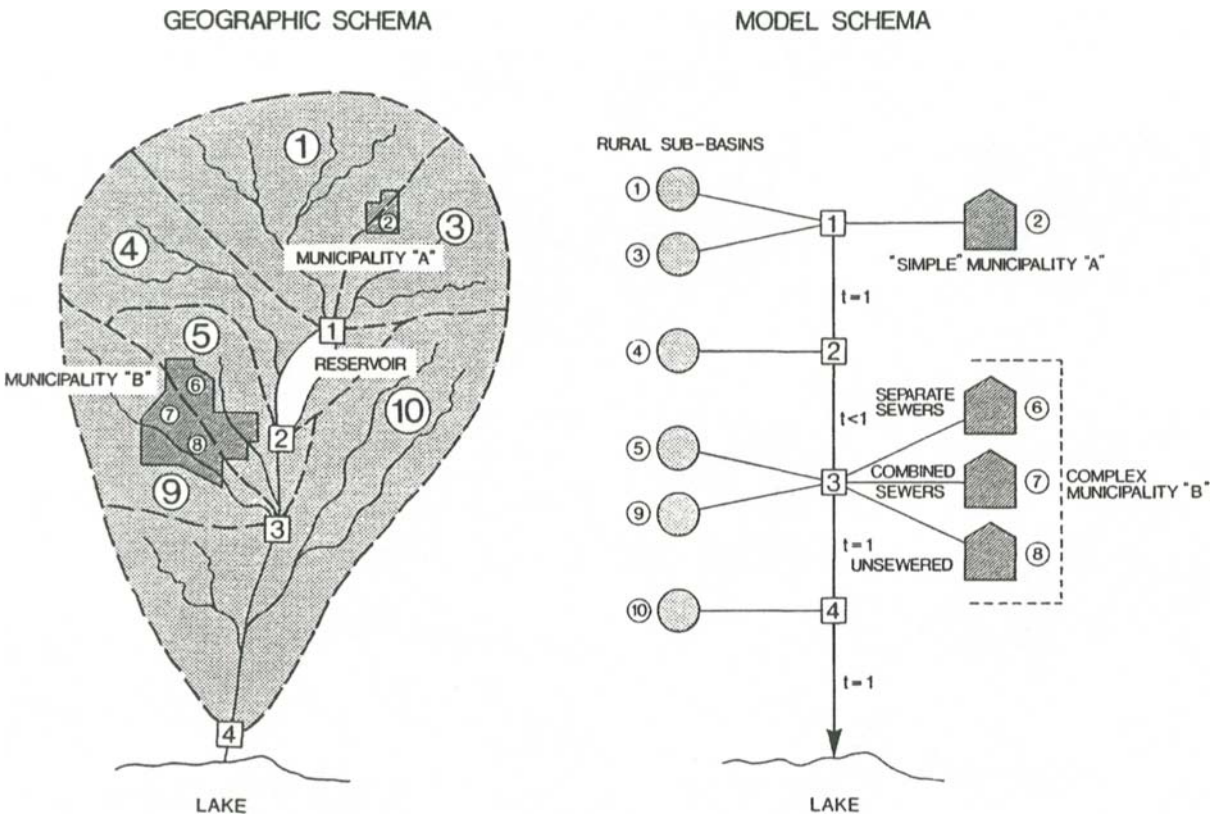



Fig. 9.4. Watershed model for Great Lakes used during the PLUARG study (PLUARG, 1978).

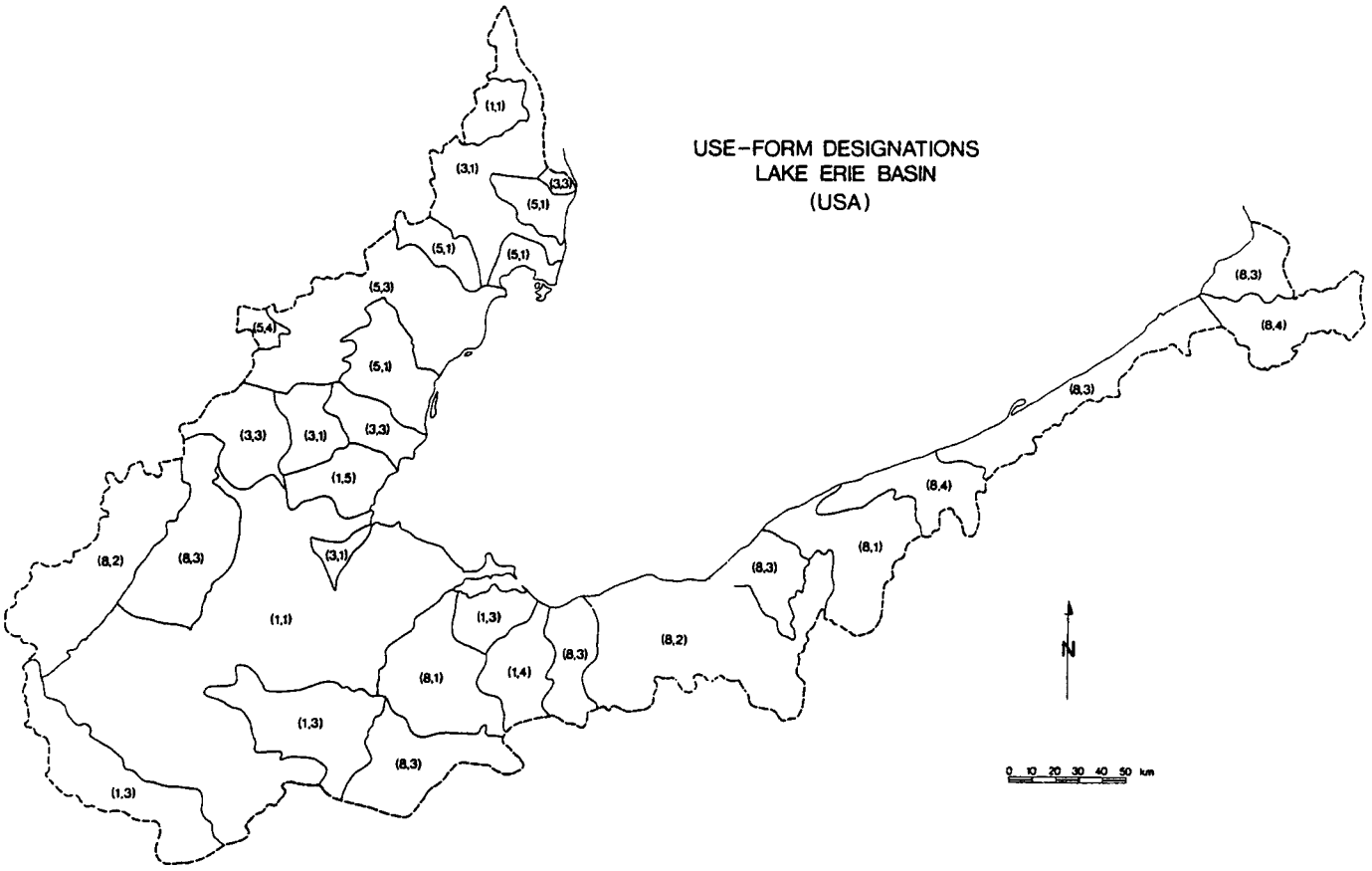



Fig. 9.5. Land use—land form designation for the U.S. portion of the Lake Erie drainage basin (Johnson et al., 1978).

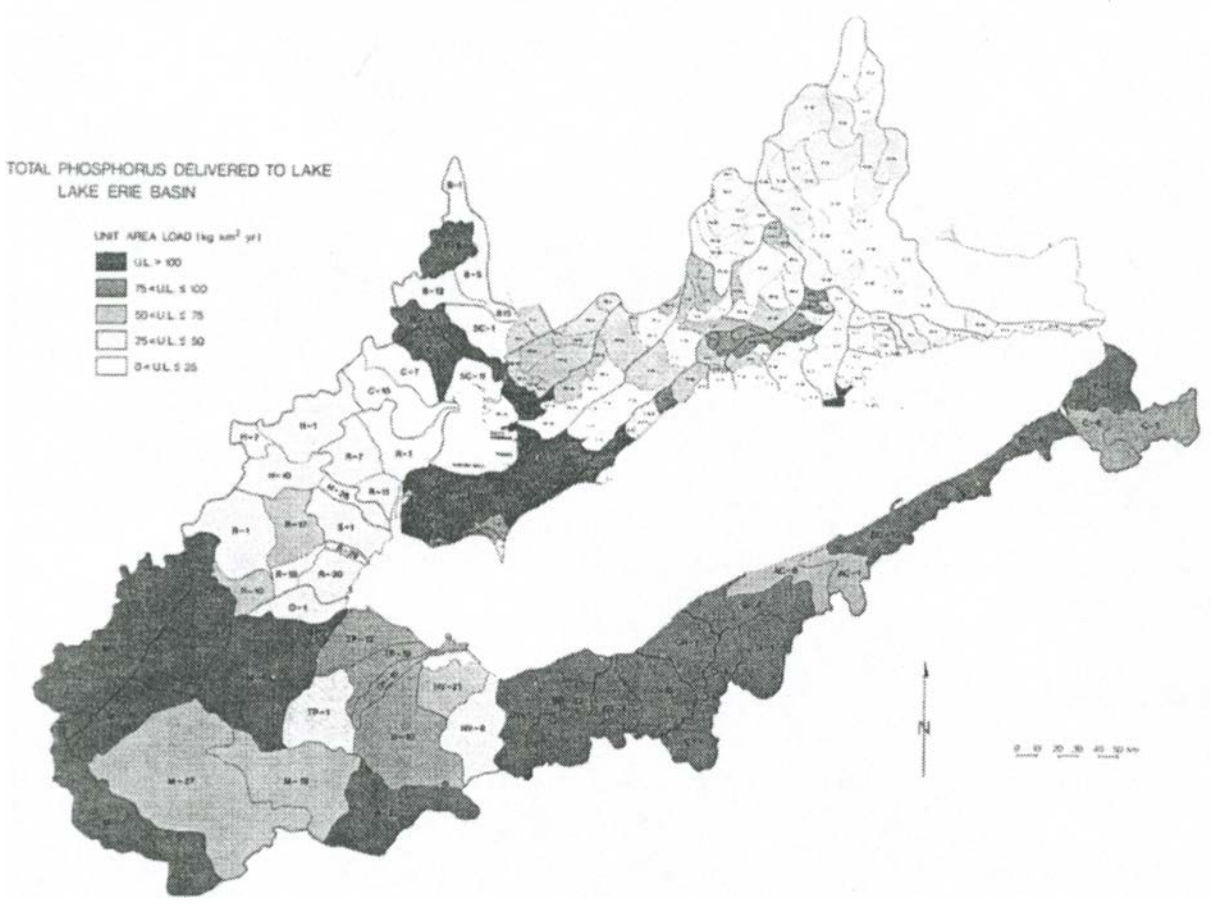



Fig. 9.6. Total phosphorus unit area loads for Lake Erie drainage basin (Johnson et al., 1978).

Table 9.3. Relative diffuse tributary phosphorus loads from major land use categories in the Great Lakes Basin (PLUARG, 1978)

| Land use | Percent of total phosphorus load, estimated from Universal Soil Loss Equation Lake | | | | |
|---|---|---|---|---|---|
| | Superior | Michigan | Huron | Erie | Ontario |
| Urban | 7 (<1)[a] | 12 (3) | 12 (2) | 21 (9) | 19 (4) |
| Cropland | 4 (<1) | 64 (12) | 61 (9) | 61 (39) | 55 (11) |
| Pasture | 3 (1) | 7 (11) | 7 (13) | 5 (20) | 11 (21) |
| Forest | 74 (94) | 3 (50) | 11 (66) | 1 (17) | 3 (56) |
| Other | 12 (4) | 14 (23) | 9 (10) | 12 (15) | 12 (8) |

[a]Percent of lake basin in identified land use.

PLUARG examined alternative management options for reducing Great Lakes phosphorus loads, and selected several point and nonpoint remedial control options, as follows:

- Treatment of municipal sewage treatment plants discharging in excess of 3800 $m^3$/day from ambient effluent phosphorus concentration to concentrations of 1.0, 0.5 or 0.3 mg $l^{-1}$,
- Two levels of treatment for urban runoff (see below),
- Three levels of treatment for rural runoff (see below).

The agricultural remedial programs were defined by the percent reduction in the unit-area loads and the annual costs per $km^2$ of land to which the treatment was applied. The Level 1 agricultural program consisted of measures such as:

- Proper incorporation of fertilizers and manure into the soil,
- Avoiding excessive use of organic fertilizers,
- General conservation plowing techniques,
- Minimizing tillage,
- Mulching,
- Avoiding farming on slopes.

The above measures were considered "common sense" techniques applicable to all agricultural land, resulting in a 10% percent reduction in the diffuse rural load at a "minimal" cost per $km^2$. The Level 2 rural nonpoint measures were comparable to "best management practices", consisting of Level 1 measures, plus more rigorous programs in areas where row crops are grown on fine-textured soils (e.g., conservation tillage, buffer strips, strip cropping, better drain construction, and winter cover crops). Depending on the specific measures, estimated costs ranged from US $3.80–23.80/ha. For the Great Lakes, the Level 2 rural measures would result in a 25% reduction in the rural nonpoint phosphorus load at a cost of approximately US $15/ha. The Level 3 rural program comprised the Level 2 program, plus more intensive application of Level 2 measures (e.g., increased crop cover, conservation tillage and strip cropping, plowing in spring instead of fall, pasture establishment/management, gradient terracing, grassed waterways), with the costs ranging from US $2.50–125/ha. For the Great Lakes, the Level 3 rural program would

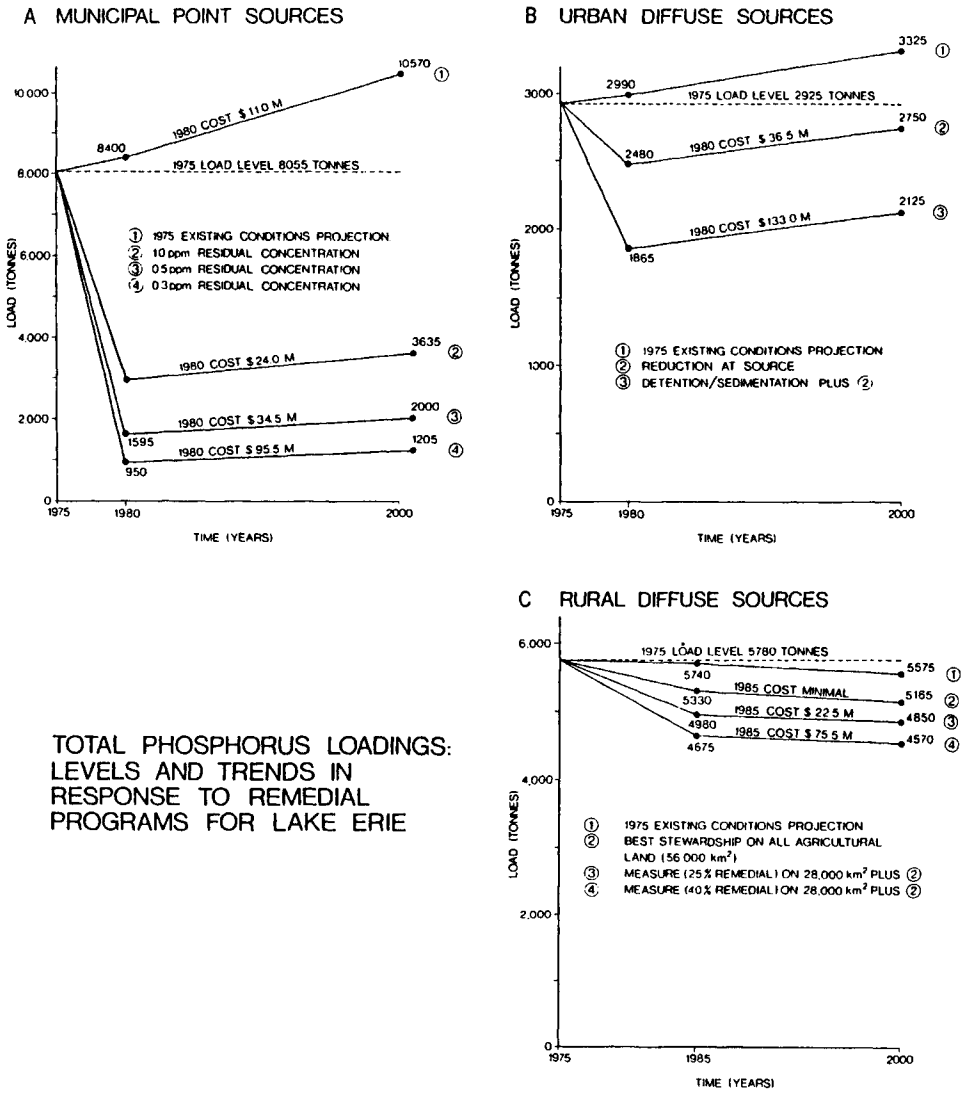



Fig. 9.7. Total phosphorus loading levels and trends in response to remedial programs for Lake Erie (Johnson et al., 1978).

result in a 40% reduction in the rural nonpoint phosphorus load at a cost of approximately US $55/ha. Fig. 9.7 and Table 9.4 summarize these suggested phosphorus load alternatives for Lake Erie. Further details on the nonpoint source control measures considered in the overview model, including their costs and effectiveness, are discussed by Marshall Macklin Monaghan Limited (1977) and Skimin et al. (1978).

Table 9.4. Phosphorus load reduction options for Lake Erie[1] (PLUARG, 1978)

| Remedial measure options | Estimated incremental phosphorus reduction (t) | Estimated cumulative phosphorus reduction (t) | Estimated incremental annual cost (US $ million) |
|---|---|---|---|
| *Urban point sources* | | | |
| Reduction of municipal sewage treatment plant effluent concentrations | | | |
| (a) 1 mg $l^{-1}$ to 0.5 mg $l^{-1}$ | 1305 | 6685 | 10.5 |
| (b) 0.5 mg $l^{-1}$ to 0.3 mg $l^{-1}$ | 645 | 7330 | 61.0 |
| *Rural nonpoint sources* | | | |
| Level 1 (sound management on all agricultural lands; = 10% phosphorus reduction) | 450 | 450 | minimal |
| Level 2 (Level 1 measures, plus buffer strips, strip cropping, improved municipal drainage practices; = 25% phosphorus reduction) | 350 | 800 | 22.5 |
| Level 3 (Level 2 measures at greater intensity; = 40% phosphorus reduction) | 305 | 1105 | 53.0 |
| *Urban nonpoint sources* | | | |
| Level 1 (program of pollutant reduction at source) | 445 | 445 | 36.5 |
| Level 2 (Level 1, plus detention, sedimentation) | 615 | 1060 | 96.5 |

[1] A reduction of 2400 tons/year was recommended, based on 1976 datum.
[2] Reduction in 1980 from 1976 load; values cumulative only within each urban and rural category.

The PLUARG overview model was not developed to select a specific scheme of remedial programs for achieving Great Lakes target phosphorus loads. Rather, it was to provide decision-makers with a means of comparing various phosphorus management options for the Great Lakes Basin, thereby allowing them to assess the cost-effectiveness of alternative nonpoint and point source phosphorus programs prior to implementing them. The overview model also helped PLUARG identify the land areas contributing the greatest quantities of nonpoint source phosphorus loads to the Great Lakes.

*Cost-effective phosphorus control programs for the Great Lakes*

PLUARG also used the overview model to identify the least expensive mix of phosphorus control programs to achieve the phosphorus target loads in the 1978 Great Lakes Water Quality Agreement.

The required annual phosphorus load reductions to achieve the target loads in the Agreement are:

- 680 tons for Lake Huron (including 100 tons for the whole lake, and 580 ton for Saginaw Bay),
- 2400 tons for Lake Erie,
- 2400 tons for Lake Ontario.

Based on their ambient phosphorus conditions, the only necessary control measures for Lakes Superior and Michigan was to achieve a 1.0 mg $l^{-1}$ phosphorus concentration in effluents from municipal sewage treatment plants discharging in excess of 3800 $m^3$/day.

In order to achieve the target load of 11,000 tons/year for Lake Erie, the point and nonpoint source phosphorus control measures being considered would result in an annual phosphorus load reduction of 2400 tons from the 1975 estimated load of 13,400 tons. Assuming the 1.0 mg $l^{-1}$ phosphorus effluent requirement was achieved, the overview model indicated that a further decrease in the effluent concentration to 0.5 mg $l^{-1}$ would reduce the phosphorus load an additional 1305 tons/year (approximately half the reduction needed to achieve the target load), for a total annual cost of US $34.5 million. This would include US $24 million/year to reduce the effluent concentrations from their ambient 1975 level to the required 1.0 mg $l^{-1}$ for municipal treatment plants discharging in excess of 3800 $m^3$/day, plus an additional US $10.5 million/year to reduce the effluent concentrations from 1.0 to 0.5 mg $l^{-1}$. Achieving a 0.3 mg $l^{-1}$ effluent concentration would cost an additional US $61 million/year, for a load reduction of only 645 tons/year. Implementing a Level 1 urban nonpoint control program would achieve a 445 tons/year load reduction at a total annual cost of US $36.5 million. The Level 2 urban nonpoint program would cost an additional US $96.5 million/year, but would only achieve an additional load reduction of 615 tons/year. In contrast, a Level 1 rural nonpoint program would achieve a 450 tons/year phosphorus load reduction at a "minimal" cost. The Level 2 rural program would decrease the rural load by an additional 350 tons/year at an annual total cost of US $22.5 million, while the Level 3 rural program would decrease the rural load by an additional 305 tons/year for an additional total annual cost of US $53 million/year.

Lakes Ontario and Huron were analyzed in the same manner. Based on the overview model, the most cost-effective mix of point and nonpoint source phosphorus control programs for Lakes Huron, Erie and Ontario are illustrated in Table 9.5. An important conclusion of the overview model was that the most cost-effective phosphorus control program for all three lower Great Lakes included at least some level of nonpoint source phosphorus control, rather than attempting to achieve the most stringent municipal point source effluent phosphorus concentration of 0.3 mg $l^{-1}$. The reader is referred to other sources for additional information on the PLUARG study results (IJC, 1970; Johnson et al. 1978; PLUARG, 1977, 1978, 1979; Rast and Gregor, 1979; U.S. Department of State, 1978).

Information on more recent developments in the Laurentian Great Lakes can be found in several summarizing publications, including the Environmental Atlas of Botts and Krushelnicki, the review of toxic contaminants by Allan and Ball (1990), the monograph on Lake Huron by Munawar et al. (1995) and on Lake Erie by Munawar et al. (1999).

Table 9.5. Summary of most cost-effective mix of point and nonpoint source control options to achieve Great Lakes phosphorus target loads (PLUARG, 1978)

| Lake basin | Phosphorus load reduction (tons/year) | Annual cost (US $) |
|---|---|---|
| HURON | | |
| (A total reduction of 680 tons/year from 1975 levels is necessary to achieve target phosphorus loads for whole lake (100 tons/year) and Saginaw Bay (580 tons/year)) | | |
| (1) Voluntary sound management on all agricultural land | 90 | "minimal" |
| (2) 1.0 mg $l^{-1}$ effluent concentration from municipal sewage treatment plants | 285 | (2.0) 1.0—new program costs |
| (3) 0.5 mg $l^{-1}$ effluent concentration from municipal sewage treatment plants | 125 | 1.5 |
| (4) Second level of effort on all cropland in fine-textured soil areas | 75 | 4.0 |
| (5) First level of effort for urban areas | 105 | 8.0 |
| *Total* | 680 | 14.5 |
| ERIE | | |
| (A total reduction of 2400 tons/year from 1975 levels is necessary to achieve target phosphorus loads for whole lake) | | |
| (1) Voluntary sound management on all agricultural land | 450 | "minimal" |
| (2) 0.5 mg $l^{-1}$ effluent concentration from municipal sewage treatment plants | 1305 | (34.5) 10.5—new program costs |
| (3) Second level of effort on all cropland in fine-textured soil areas | 350 | 22.5 |
| (4) First level of effort for urban areas | 445 | 36.5 |
| *Total* | 2550 | 69.5 |
| ONTARIO | | |
| (A total reduction of 2400 tons/year from 1975 levels is necessary to achieve target phosphorus loads for whole lake) | | |
| (1) Voluntary sound management on all agricultural land | 80 | "minimal" |
| (2) 0.5 mg $l^{-1}$ effluent concentration from municipal sewage treatment plants | 1000 | (23.5) 7.5—new program costs |
| (3) First level of effort for urban areas | 140 | 14.0 |
| *Sub-Total* | 1220 | 21.5 |
| (4) Reduction to Lake Ontario from successful implementation of Lake Erie program | 1200 | |
| *Total* | 2420 | |

## 9.3 LAKE FURE (DENMARK)

Lake Fure is located near Copenhagen (15–20 km from the center of the city). It is surrounded on three sides by forest and wetlands used as recreational areas by the citizens of Copenhagen. The lake is used almost entirely for recreational purposes, including sailing, rowing, angling and swimming (mainly for children, since the distance to the sea is only 12 km).

During the 1950s and 1960s, the city of Copenhagen expanded toward the lake, because many people found it attractive to live close to nature, and the districts north of Copenhagen, where Fure Lake is situated, is characterized by many lakes and forests. Many one-family houses were built on the north side of Copenhagen between 1950–1975. This was accompanied by an increased discharge of municipal wastewater. In the 1960s, Lake Fure received the mechanical–biological treated wastewater from more than 60,000 inhabitants in the three communities adjacent to the lake. This resulted in increased eutrophication of the lake, with the transparency of this beautiful lake decreasing during the 1960s from about 3–4 m to about 1 meter during spring and summer algal blooms.

*Wastewater treatment*

In 1972, two communities discharging wastewater to Lake Fure decided to construct a pipeline to the sea (about 10 km) to discharge the mechanical–biological wastewater into Øresund, the water between Sweden and Denmark. A third community decided to introduce more advanced wastewater treatment, including chemical precipitation with pH adjustment to ensure a high efficiency of treatment. Chemical precipitation was used twice in the treatment, first for direct precipitation of the wastewater before the primary sedimentation phase (using iron (III) chloride), and also for post precipitation after the secondary sedimentation (using aluminum sulphate). With this method, it was possible to ensure a phosphorus concentration of $\leqslant 0.2$ mg $l^{-1}$. In the 1960s, 33 tons of phosphorus were discharged to the lake each year. After the described measures were taken, the phosphorus discharged directly to the lake was reduced to about 3 tons, half coming from stormwater overflows. About 300 kg of phosphorus is still coming from wastewater, with the remaining 1200 kg from the tributaries, particularly from the Mill River, which passes several lakes and drains agricultural areas before it reaches Lake Fure.

Eutrophication was reduced as a consequence of these measures, but only very slowly because the lake's water retention time is about 20 years. Today, more than 30 years after the measures were taken, the transparency during the spring and summer algal bloom is slightly more than 2 m.

*Environmental management considerations today*

Nitrogen removal also has been introduced to treat the wastewater. In accordance with Danish rules, all wastewater treatment plants receiving wastes from more than 15,000 inhabitants must introduce nitrogen removal. The required standard is 8 mg N $l^{-1}$. Although the possibility of removing phosphorus from stormwater overflows is being considered as

an additional means of further reducing the phosphorus load to the lake, no final decision has yet been made.

Ecological engineering also is being considered as a means of more rapidly reducing the external phosphorus load to the lake. The release of phosphorus from the lake bottom sediments (internal load) has been measured to be as much as 12 tons/year (1997), compared to an external phosphorus load of about 2.5–3 tons/year. Thus, it would be beneficial for the recovery of the lake to remove more phosphorus from the lake by siphoning the hypolimnic water, treating it with an activated aluminum oxide column, and discharging the treated water (phosphorus concentration 50 μg $l^{-1}$ downstream of the lake). This project would cost approximately US $3.5–4 million, while the treatment of all stormwater would cost in the order of US $1.5 million. The first approach would reduce the annual phosphorus load by about 10 tons after a few years, while the latter approach would only reduce the load by about 1500 kg $yr^{-1}$.

Model computations have suggested that, if no additional measures are taken, it will take another 30 years before the lake has recovered consistent with the present reduced phosphorus load of about 2.5–3 tons/year. This would imply an increased transparency to about 3 m. It would, however, be possible more rapidly to achieve a better result (i.e., within 10–15 years) if either the proposed ecological engineering method was applied or the stormwater was treated. The realization of both proposals would imply a gradual improvement of the transparency to 4–5 m in 15–20 years.

Conveying the wastewater from the third community to the sea also has been considered. However, the model results suggest this would be a waste of money, because the siphoning of hypolimnic water and/or treatment of stormwater would give better results at less cost.

In the summer of 2003 another ecotechnological solution was launched: Aeration of hypolimnion and biomanipulation. Prognoses based on the Lake Fure model have not yet been performed, but it is expected that the two projects, costing about the same as the above mentioned alternative methods, would have the same effect: The lake transparency will increase to 4–5 m over a period of 15–20 years.

*What can we learn from this case study?*

The environmental management story of Lake Fure clearly demonstrates that preventing pollution is very important (in contrast to treating it after it has become a problem). If the measures taken in 1972 had been taken in 1962, ten years earlier, the quantity of phosphorus in the lake bottom sediment would have been considerably lower, and the recovery of the lake faster and cheaper.

It has also been demonstrated that eutrophication management plans must consider all major phosphorus sources. Further, it is always beneficial to reduce all sources as early as possible. It is evident from a calculation of the phosphorus balance of the lake that the treatment (phosphorus removal) of stormwater is crucial. Initiation of stormwater treatment as early as 1972 would imply that the lake bottom sediment today would have contained 30–40% less phosphorus, thereby facilitating a faster recovery.

Finally, the study clearly demonstrated that it is beneficial to use ecological models to obtain a good overview and comparison of the various environmental management plans.

Modelling is an excellent management tool, provided that it is not oversold and not used to assess waterbodies lying outside its range of applicability.

## 9.4 LAKE ICHKEUL (TUNISIA)

Among shallow lakes, Lake Ichkeul (Table 9.6, Fig. 9.8) is not only one of the most important wintering sites in North Africa for numerous species of waterfowl (e.g., Flamingo, *Phoenicopterus ruber*, Gray-lag, *Anser anser*, Wigeon, *Anas penelope*, Pochard, *Athya ferina*, Tufted duck, *Athya fuligula*), but also a unique aquatic system in which freshwater influence dominates during the winter while seawater enters the lake basin during the summer (Anonymous, 1993, 1997).

The influence of seawater on the lake has increased considerably since 1984, when damming activities of the lake's inflows were initiated for irrigation and for a water supply for the rapidly-increasing populations of cities and major settlements. One of the major impacts is a reduction in the former large stands of bulrush *Scirpus maritimus*, and the sites of common reed. This development is of concern in regard to the waterfowl, as well as the lake's former animal diversity. A sluice was constructed in 1988 to retain freshwater from the winter precipitation and exclude seawater inflow in the summer. Nevertheless, it is clear that the former low-salinity periods cannot be regained, and that the maximum salinity of the lake is increasing (Lemoalle, 1987; Zaouali, 1995).

*Management problems*

In 1977, Ichkeul became a UNESCO Biosphere Reserve and, three years later, a National Park and a UNESCO World Heritage site. At that time, the management strategy for the lake appeared feasible. Before the 1977 declaration, minor drainage activities within the marshes and the ongoing fishery were only of little concern. The fishery was, and still is, focused on coastal fishes (e.g., mullet (*Mugil cephalus*), eel). The annual harvest totals about 230–250 tons, and is related largely to the efficient weir-basket fishery in the upper Tinja River. In addition to these activities, tourism in the area has steadily increased. At present, the number of visitors exceeds more than 25,000/year. After the 1980 declarations, however, the Tunisian Government started damming the major tributaries, Joumine, Ghezala and Sejnane, for drinking water supply and irrigation (Table 9.6). A statement on the need to protect the other tributaries (e.g., Rivers Tine and Malah) has recently been made, with plans for their damming after the year 2000 being dismissed. With more than 80% of the freshwater needed to keep the lake water's seasonal salinity, Lake Ichkeul is endangered by permanent high salinity conditions.

Figure 9.9 demonstrates the winter salinities for 1995–1997, which are far above the values observed before the final reclamation of the tributary water in 1988 (Joumine, Ghezala and Sejnane Rivers). As a result, freshwater invertebrates are disappearing, being replaced by marine and saline inland water species (Table 9.6). In addition, the decrease of emergent and submerged vegetation greatly influences the presence of many birds, such as

Table 9.6. Basic variables for Lake Ichkeul

| | |
|---|---|
| *Physical variables* | |
| Geographic location | 37°17′N, 9°40′E |
| Area, watershed (without the lake and its marshes) | About 2000 $km^2$ |
| Area, lake | 90–126 $km^2$ (including marshes which total about 30 $km^2$) |
| $z_{max}$ | 2–4 m |
| $z_{cryp}$ | −1.25 m a.s.l. |
| Salinity | 1–3 g $l^{-1}$ (winter season until 1984)<br>−32 g $l^{-1}$ (winter 1997)<br>72–75 g $l^{-1}$ (summer 1995, 1996) |
| Secchi depth | 5–80 cm |
| Length, Tinja, Connection L. Ichkeul and the Bizerta Bay: | 4.5 km |
| Total annual inflow from tributaries | 271 $10^6$ $m^3$ (tributary Sejnane 44%)<br>(tributary Joumine 38%) |

| Dam | Opened | $10^6$ $m^3$ | $km^2$ |
|---|---|---|---|
| Sejnane Dam | 1988 | 138 | 7 |
| Joumine Dam | 1984 | 130 | 6.6 |
| Ghezala Dam | 1985 | 12 | 1.4 |

*Biological variables*

Emergent vegetation of the marsh area: Bulrush *Scirpus maritimus*, Common Reed *Phragmites australis* (at present reduced to only small stands, Bulrush *Scirpus lacustris*, reed *Juncus acutus*; landwards tamerisk *Tamarix africana*)

Submerged vegetation: Skunk Weeds *Chara* sp., *Nitella* sp., *Callitriche* sp., *Zannichellia palustris*, Widgeon grass *Ruppia maritima* (not observed during the excursion in February 1997). The pond weed *Potamogeton pectinatus*, which used to be the dominant species is still abundant

Dominating algae (February 1997): Dinophyceae (*Prorocentrum micans*, *Exuviella micans*, diatoms, such as *Chaetoceras* sp., *Campylodiscus* cf. *clypeus*, *Nitzschia* spp.

Invertebrates: Brackish-water and inland saline water species such as *Cyprideis torosa littoralis*, *Cletocamptus* sp., *Gammarus locusta*, *Sphaeroma hookeri*, *Hydrobia ventrosa*; marine species such as *Nereis diversicolor*, Balanidae spp.

Fish species: Eel *Anguilla anguilla*, Twaite Shad *Alosa fallax*, *Syngnathus abaster*, *Aphanius fasciatus*, Guppy *Gambusia affinis*, mullets *Mugil cephalus* and *Mugil ramada*, Bass *Dicentrarchus labrax*

Amphibians: *Discoglossus pictus*, *Bufo viridis*, *Rana ridibunda*

Birds: About 230 species are recorded. Among breeding species, Great Crested grebe *Podiceps cristatus*, Cattle Egret *Bubulcus ibis*, Purple Heron *Ardea purpurea*, Little Egret *Egretta garzetta*, Purple Gallinule *Porphyrio porphyrio*, Kentish Plover *Charadrius alexandrinus*, Black-Winged Stilt *Himantopus himantopus*, etc. Mainly summer residents: Flamingo *Phoenicopterus ruber*, Spoon Bill *Platalea leucorodia*. Mainly winter residents: Gray-Lag Goose *Anser anser*, Wigeon *Anas penelope*, Pochard *Aythya farina*, etc.

Mammals: Otter *Lutra lutra*, Water buffalo *Bubalus bubalis* (introduced)

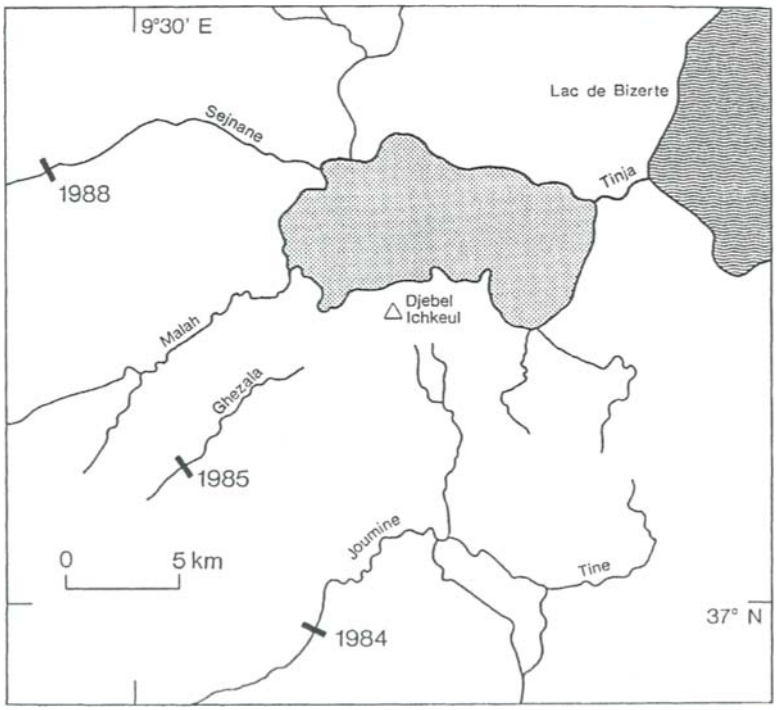



Fig. 9.8. Map of Lake Ichkeul.

Widgeon (*Anas penelope*). Apart from natural fluctuations in numbers, the winter residents decreased from 112,000 individuals (November, 1973) to 3500 (January, 1983; January, 1987), and were not observed at all in February, 1997. Other herbivorous waterfowl species also have decreased significantly in numbers.

In spite of these shortcomings, the value of Lake Ichkeul was again emphasized with the further designation of the area as a Ramsar Site in 1987. A third and major dam was completed on the Sejnane River in 1998. At approximately the same time, the construction of a sluice at a lakeward position of the Tinja River began operation. This weir, proposed by the University College London group (Hollis et al., 1983) and French engineers (Skinner et al., 1987) is supposed to retain the freshwater accumulated in the lake basin from the winter precipitation, and also control the sea water inflow during the summer. For the latter task, a pumping device was established at the weir site. In addition, the sluice provides devices for fish migration (mullet, eel).

Nevertheless, there is no doubt that, due to the dramatic reduction of the freshwater potential of the lake, its size will be confined to about 70–80% of its former winter extension. Under these conditions, the marsh area will be considerably reduced, thereby also reducing the potential sites of waterfowl dependent on the emergent vegetation for breeding.

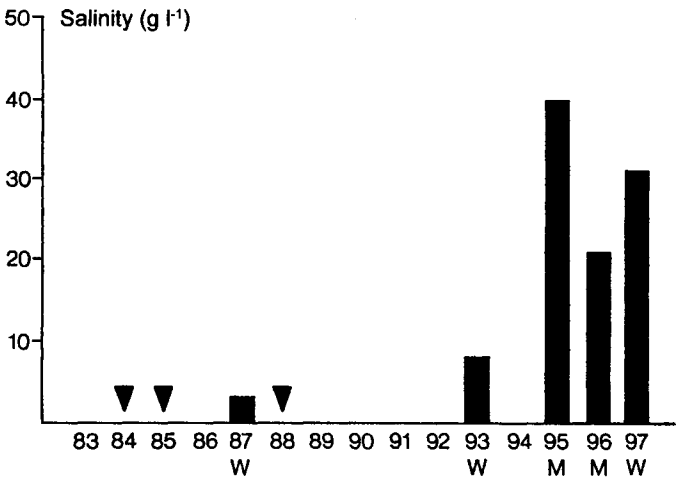



Fig. 9.9. Lake Ichkeul. Arrows indicate years when dams on tributaries were constructed; W—winter, M—March.

With this type of development, Lake Ichkeul belongs to an increasing group of shallow lakes that have become endangered as a result of human impacts on their water budgets.

However, according to communications from the Tunesian environmental agencies (December, 2002) the condition of Ichkeul since 1997 has to some extent been stable, although the sluice for preventing excess seawater and retaining freshwater in the lake has not yet been finished. No further damming of inflows seems to be considered and some of the macrophytes such as *Potamogeton pectinatus* and *Bulboschoenus maritimus* are still abundant. During the late 2000, heavy rainfall resulted in a water level of about 2 m and abundant waterfowl was reported.

## 9.5 BIESBOSCH EMBANKMENT RESERVOIRS (THE NETHERLANDS)

The Water Storage Corporation Brabantse Biesbosch (WBB) manages three interconnected, stratified reservoirs to supply raw water to several waterworks in the southern and western parts of The Netherlands (Table 9.7). The embankment above terrain reservoirs were constructed during the 1970s, being located near the confluence of the Rhine and Meuse Rivers (Fig. 9.10). The De Gijster Reservoir receives moderately-polluted, nutrient-rich water from the River Meuse. The water retention time is about 5–6 months, in which water quality improves considerably, due to in-lake chemical, physical and biological processes (Oskam, 1982; Oskam and Van Breemen, 1992).

With a water retention time in each reservoir of 1–3 months, and an average doubling time of algae of 2–4 days, the effects of the hydraulic washout can be neglected in the Biesbosch reservoirs (Van Breemen and Ketelaars, 1995). The relative magnitude of the other losses varies in time (seasons) and space (reservoirs), being influenced by reservoir

Table 9.7. Characteristics of Biesbosch reservoirs and air injection system (Oskam and van Breemen, 1992)

| Reservoir | De Gijster | Honderd en Dertig | Petrusplaat |
|---|---|---|---|
| Surface area (ha) | 305 | 210 | 100 |
| Perimeter (m) | 8500 | 6400 | 4300 |
| Top embankment a.s.l. (m) | 8.5 | 8.5 | 8.5 |
| Bottom depth max. a.s.l. (m) | −20 | −20 | −9 |
| Average water level a.s.l. (m) | 6.5 | 6.5 | 6.5 |
| Volume (million $m^3$) | 40 | 33 | 13 |
| Maximum depth (m) | 27 | 27 | 15 |
| Mean depth (m) | 13 | 15 | 13 |
| Water retention time (weeks) | 11 | 9 | 4 |
| Injection units | 6 | 3 | 3 |
| Water flow | 166 million $m^3$ $yr^{-1}$ | | |
| Air flow | 0.8 $m^3$ $s^{-1}$ | | |
| Compressor power | 250 kW | | |
| Compressor working hours | 2200 hr $yr^{-1}$ | | |
| Annual energy consumption | 550,000 kWh $yr^{-1}$ | | |

morphology, changes in temperature and light climate, stability of the water column and zooplankton-grazing activity. Because each reservoir can be considered an independent ecosystem, these loss factors vary per reservoir (Van Breemen and Ketelaars, 1995).

*Destratification system*

The destratification system developed for the Biesbosch reservoirs uses air injection at the lake bottom to maintain isothermal conditions (Oskam and Van Breemen, 1992). Air injection takes place in so-called injection units, with the number of units varying from three to six per reservoir, depending on the size of the reservoir. The injection unit is very compact, consisting of a 1-foot by 3-foot meter steel frame, on which six polyethylene foam plastic blocks are installed. The land-based compressors feed the injection units by means of plastic pipes, forcing air through the foam blocks and producing a bubble column of very finely-divided air, thus entraining large amounts of water.

This system has several advantages. Because the foam blocks do not clog, the maintenance costs are very low. Further, because the foam block expands when the air injection is re-started, materials deposited during long periods of nonuse (autumn and winter) crumble. The installation is easy and very flexible, since the position of the units can be easily altered if their original configuration does not prove to be optimal. The overall investment is moderate. Table 9.7 gives the operational characteristics of the air injection systems in the three Biesbosch reservoirs. The total operating cost of the reservoir mixing system is about €40,000 per year. The capital cost is €75,000 (a total investment of Euros 0.75 million), so the total cost is calculated at less than €0.0007 /$m^3$ of delivered water, a very moderate figure (Oskam and Van Breemen, 1992). The air injection is very effective in maintaining homogeneous conditions in the reservoirs. Measurements have shown that, under optimal

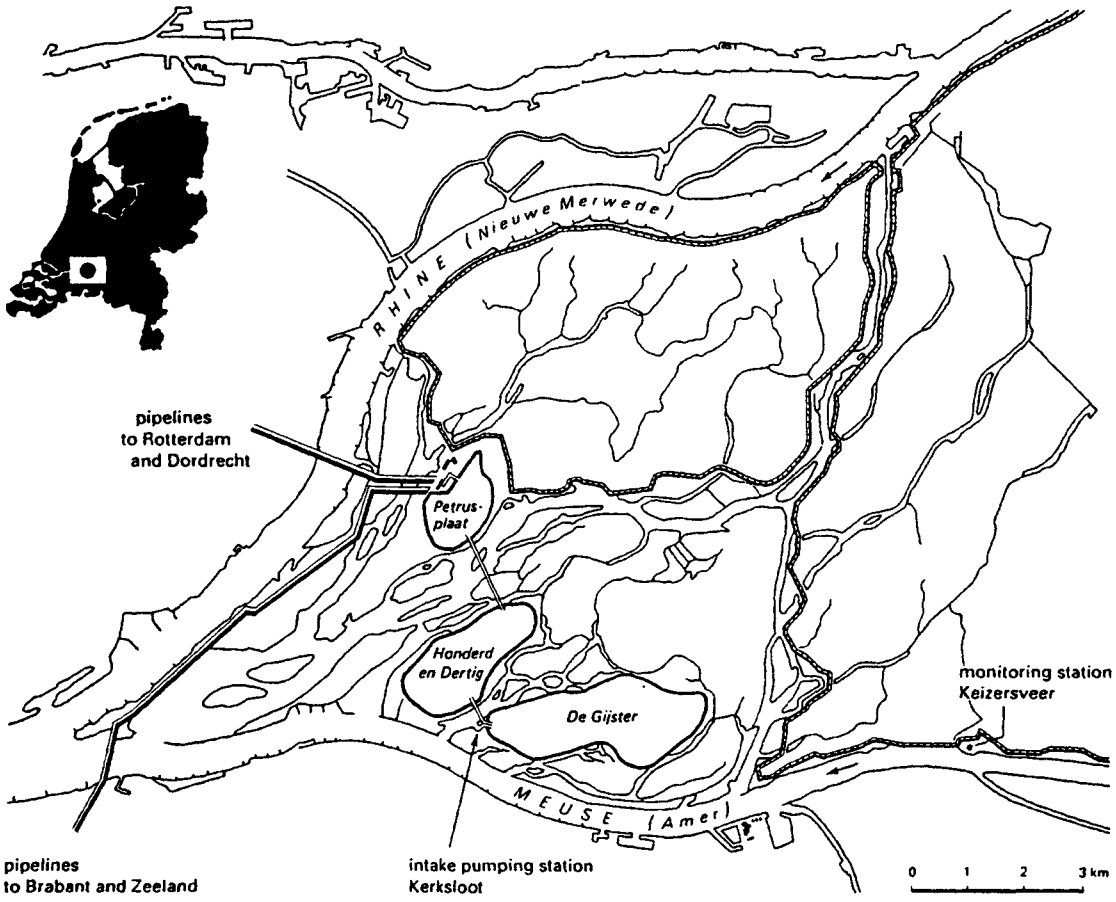



Fig. 9.10. Location of Biesbosch reservoirs in The Netherlands.

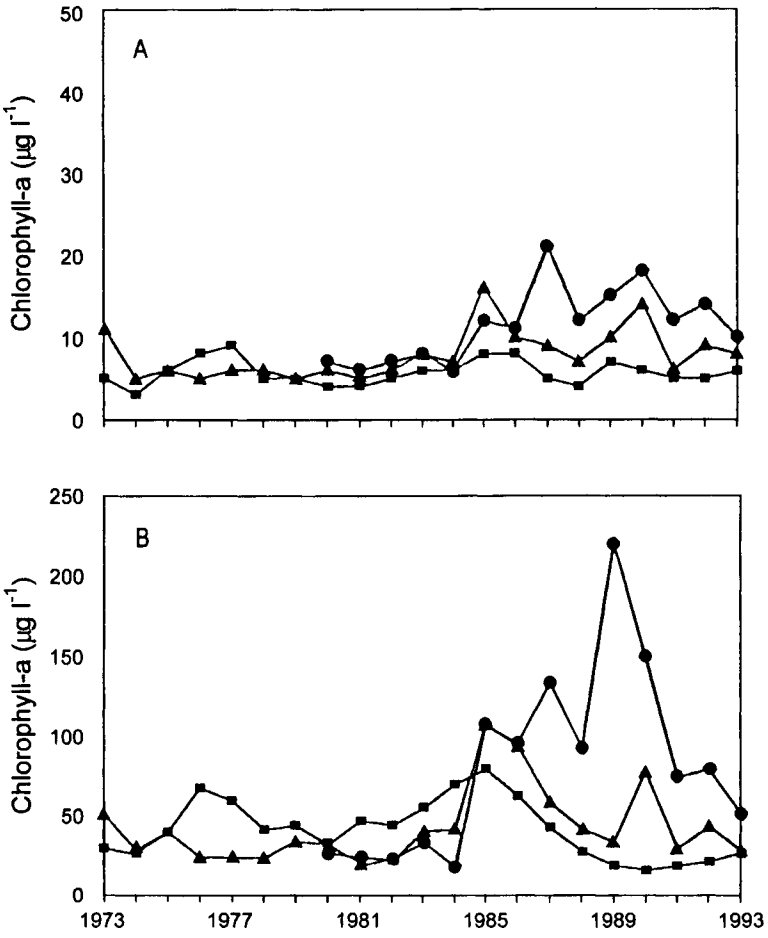



Fig. 9.11. Annual mean (A) and maximum (B) chlorophyll-a concentrations in Biesbosch reservoirs outlets, 1973–1993. ●, De Gijster; ▲, Honderd en Dertig; ■, Petrusplaat (from van Breemen and Ketelaars, 1995).

conditions, 1 $m^3$ of air entrains up to 250 $m^3$ of water, thus ensuring efficient vertical and horizontal mixing (Oskam and Van Breemen, 1992). The mixing time of the reservoirs is small, compared to the average residence time, so the reservoirs can be considered as a series of fully-mixed reactors.

*Eutrophication degree and its management*

Control of algal growth by artificial mixing of lake water is based on the principle of light limitation, depending mainly on the available mixing depth in relation to the optical properties of the Biesbosch reservoirs (Oskam, 1978). According to the phosphorus load–

lake response model of Vollenweider (1976), the mean annual phosphorus concentration of 18–30 $\mu g\ l^{-1}$ in the Meuse is expected to result in a mean annual chlorophyll-a (CHA) concentration of 30–50 $\mu g\ l^{-1}$ in the Biesbosch reservoirs. The annual mean CHA concentrations at the reservoir outlets from 1973–1993 were well below the predicted values (Fig. 9.11). Sedimentation of particulate phosphorus and phosphorus uptake by algae decreases the quantity of phosphorus in the three interconnected reservoirs. Based on a simple kinetic model of light-limited algal growth, the maximum CHA concentration in the Biesbosch reservoirs is predicted to vary between 40–70 $\mu g\ l^{-1}$ (Oskam and Van Breemen, 1992). Without light limitation, the Lund (1978) model predicts maximum chlorophyll-a concentrations in the De Gijster, Honderd en Dertig and Petrusplaat reservoirs of 590, 370 and 280 $\mu g\ l^{-1}$, respectively. The maximum CHA concentrations in the reservoirs were well below the predicted concentrations (Fig. 9.13). The mean and maximum CHA concentrations are sometimes much lower than the predicted values because, in addition to light limitation, other factors also may reduce the algal biomass (Van Breemen and Ketelaars, 1995).

*Buoyant Cyanobacteria*

Growths of buoyant Cyanobacteria (mainly *Microcystis*) in the Biesbosch reservoirs starts around June and ends in September/October, with the maximum biomass being found in August/September. Although a wide variation in the algal biomass is observed in different years, the highest biomass is always found in the De Gijster Reservoir, and the lowest in the Petrusplaat Reservoir (Oskam and Van Breemen, 1992). The higher biomass of Cyanobacteria in De Gijster Reservoir can be explained by the different morphologies of the littoral zone of the Biesbosch reservoirs (Fig. 9.12). Only the De Gijster Reservoir has a shallow zone (width 100 m, depth 3–7 m) around its entire perimeter, which is suspected to result in insufficient mixing and favorable growth conditions for Cyanobacteria (Van Breemen and Ketelaars, 1995).

Because buoyancy changes during the day, and the photosynthetic characteristics, were different at the shallow and deep sites, the shallow and the deep part of the reservoir can be considered as two different systems with restricted horizontal mixing (Visser et al., 1996). Thus, the Cyanobacteria are entrained in the shallow part, receiving a higher daily light dose, which results in a higher growth rate in the shallow part. The mixing velocity in the reservoirs depends on the quantity of the injected air, and the wind speed. Below a critical mixing velocity, Cyanobacteria that possess buoyancy regulation may succeed in staying in the upper layers, forming algal scums. Their horizontal distribution depends on a critical wind-speed of 2–3 $m\ s^{-1}$, which is needed to mix the algae. Below this speed, the wind blows the algae toward the downwind shore, with algal scums being formed. In very tranquil weather-conditions, *Microcystis* scums were observed occasionally in all three reservoirs.

*Benthic Cyanobacteria*

Benthic Cyanobacteria and algae can increase because of the reduced algal biomass and increased transparency caused by use of the vertical-mixing method. Because of

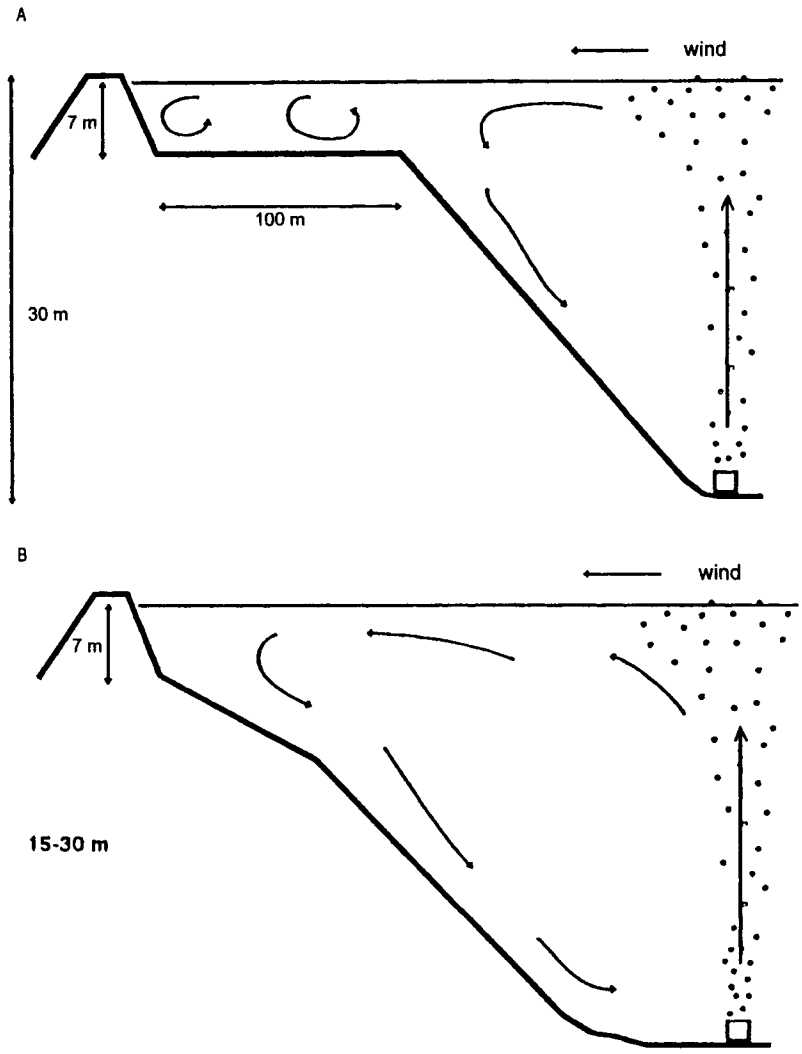



Fig. 9.12. Schematic cross-section of the littoral zone of Biesbosch reservoirs. A—De Gijster Reservoir; B—Honderd en Dertig and Petrusplaat Reservoirs. (Orig.)

the wind-exposed location, and the strong wave action of the Biesbosch reservoirs and their asphalt-concrete embankments, rooted plants are absent (Van Breemen and Ketelaars, 1995). Growths of benthic algae, however, are less susceptible to wave action, meriting close attention because of the potential production of toxins, and taste and odor compounds. Ever since April/May 1984, spring blooms of benthic Cyanobacteria

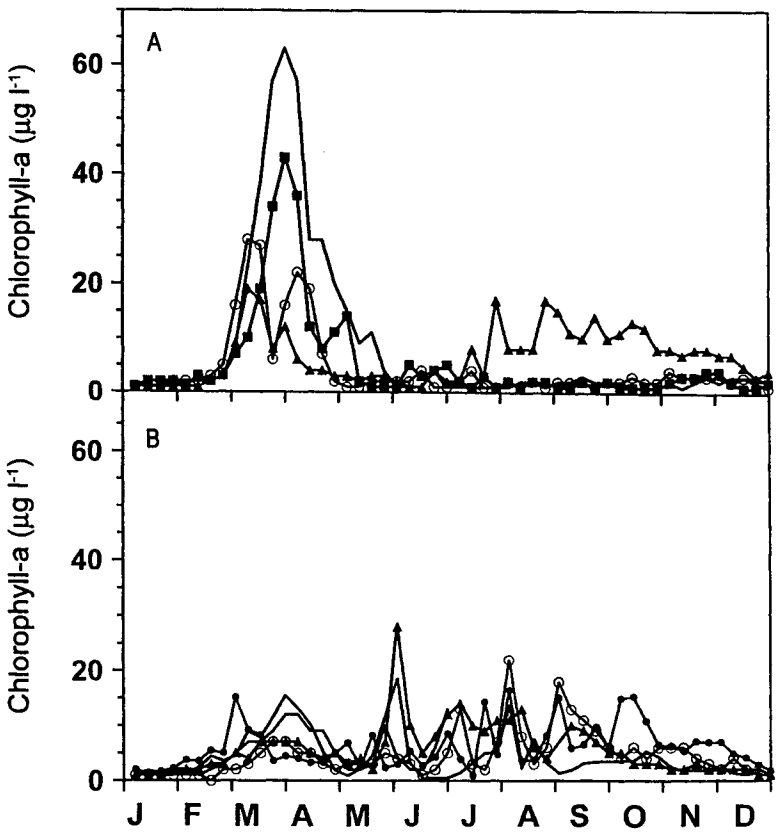



Fig. 9.13. Seasonal variation of chlorophyll-a concentrations in the outlets of Biesbosch reservoirs. A—Petrusplaat Reservoir 1986–1989. B—Petrusplaat Reservoir 1990–1994. 1986; 1987; 1988; 1989; 1990; 1991; 1992; 1993; 1994 (from van Breemen and Ketelaars, 1995).

(*Oscillatoria* spp.) have been a recurring phenomenon in the Petrusplaat Reservoir. Mechanical disturbance of the bottom has proven to be a successful remedy for this problem (Van Breemen et al., 1991).

*Nutrients*

The influence of nutrients on algal biomass is illustrated by the interaction between diatoms (Bacillariophyceae), silica and CHA in the Petrusplaat Reservoir during the spring of 1984 (Fig. 9.13). The algal biomass was dominated by the centric diatoms *Stephanodiscus hantzschii* and *Stephanodiscus astraea* var. *minitula*. When it reached its maximum value of 80 $\mu g\ l^{-1}$ CHA, the silica concentration dropped below the detection level of 0.2 $mg\ l^{-1}$, with the diatom cells being weakly silicated at that time (Van Breemen and Ketelaars,

1995). Both phenomena suggested that silica-depletion limited diatom growth in the reservoir, although other factors (e.g., parasitism, sedimentation) also may have contributed to the collapse of the diatom bloom. A second example of the influence of abiotic factors on the plankton composition and biomass in the Biesbosch reservoirs is the conspicuous absence of Chrysophyceae from the Petrusplaat Reservoir. The water in this reservoir is softened by adding caustic soda or lime, resulting in an increased pH to values above 9 and carbon dioxide ($CO_2$) concentrations dropping to almost zero, with limitation of Chrysophyceae taking place.

*Zooplankton*

Due to the low abundance of zooplankton at the start of the spring pulse, the algal biomass depended entirely on the light limitation caused by artificial mixing. The low algal biomass during the rest of the growing season was the result of light limitation, and grazing pressure of the herbivorous zooplankton, consisting mainly of *Daphnia galeata*, *Daphnia pulex/pulicaria* and *Eudiaptomus gracilis*.

Beginning in 1989, an entirely different seasonal pattern emerged. The characteristic diatom spring pulse almost vanished, possibly because of a series of mild winters that stimulated the early appearance of herbivorous zooplankton. The summer period, on the other hand, showed an increase of algal biomass to maximum levels of about 20 µg CHA $l^{-1}$. It was hypothesized that this increase could be related to the increasing abundance of the predatory cladoceran, *Bythotrephes longimanus*, that started to invade the Biesbosch reservoir system in 1987 (Ketelaars and Van Breemen, 1993). Calculations suggested that as much as 75% of the daphnid population in the De Gijster Reservoir could be eliminated by *Bythotrephes* and *Leptodora kindti* during periods of maximum predator density (Ketelaars and Van Breemen, 1993). In fact, peak abundance of *Bythotrephes* and *Leptodora* did coincide with drastic declines in daphnid numbers. However, this phenomenon only indicates a short-term influence of predatory cladocerans on the herbivorous zooplankton. There is no convincing proof so far that the invasion of *Bythotrephes* has significantly affected the zooplankton composition in the reservoirs.

A new Ponto–Caspian invader, the mysid *Hemimysis anomala* was recorded in 1997 for the first time in The Netherlands (Ketelaars et al., 1999). In the summer of 1998, extremely high densities (> 6 individuals/l) of this species were recorded in one of the Biesbosch reservoirs (Honderd en Dertig Reservoir). This invasion had dramatic effects on the zooplankton composition and abundance (Ketelaars et al., 1999). From the end of August, almost no Cladocera, Ostracoda, Rotifera and invertebrate predators (*Leptodora kindti* and *Bythotrephes longimanus*) were present in the waterbody. The copepod densities, however, were not influenced (Fig. 9.14).

The chlorophyll-a concentrations were significantly lower, compared to previous years, possibly the result of mysids feeding on the algae. Laboratory experiments revealed that *Hemimysis anomala* is a voracious predator, as well as being an omnivorous feeder. If *H. anomala* also exploits the phytoplankton biomass, the overall water quality may appear to be the same as when other zooplankters are present. However, by reducing the number of

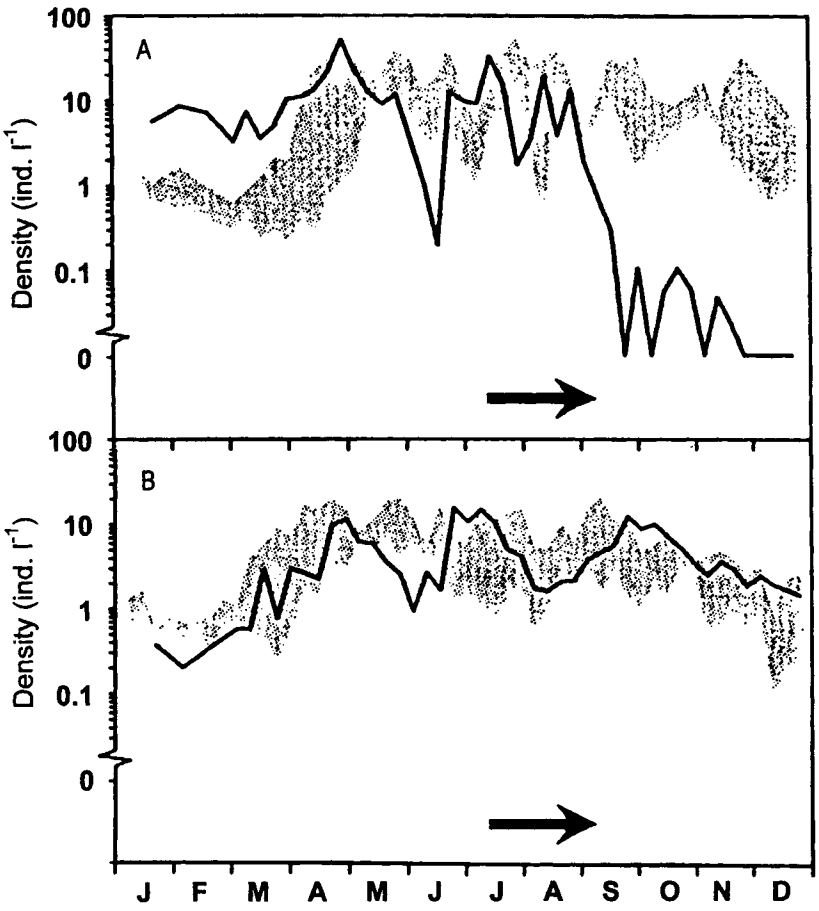



Fig. 9.14. Filtering zooplankton densities in the Honderd en Dertig Reservoir. A—Densities of *Daphnia* spp. and *Bosmina* spp. in 1998 (solid line) compared to those in 1995–1997 (dotted area); B—The same for adult copepoda. Horizontal arrow indicates the start of the period that *Hemimysis anomala* was found in vertical plankton hauls (13 July, 1998) (from Ketelaars et al., 1999).

trophic levels in the waterbody, it will make the ecosystems less stable and more vulnerable to small disruptions.

*Fish*

It is common knowledge that the zooplankton biomass and composition can be strongly influenced by planktivorous fish, particularly juvenile cyprinids. Large zooplankton-like daphnids are a preferred prey species, being prone to disappear first under high fish predation pressure. Fishery surveys with gill-nets (Fig. 9.15) have shown that cyprinids (mainly

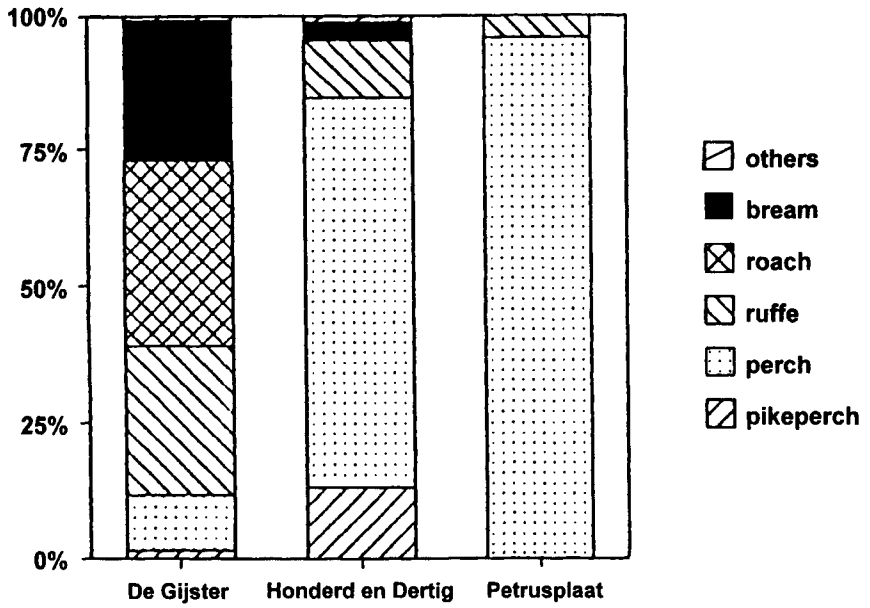



Fig. 9.15. Relative abundance of fish caught in gill-nets in the Biesbosch reservoirs: De Gijster (1991), Honderd en Dertig (1992) and Petrusplaat (1993) (from van Breemen and Ketelaars, 1995).

roach *Rutilus rutilus* and bream *Abramis brama*) are of minor importance, or even totally absent, in two of the three reservoirs.

The situation in the De Gijster Reservoir is different because fish are pumped in from the river. In the absence of vegetation in the littoral zone of the reservoirs, it appears that natural recruitment is restricted to perch (*Perca fluviatilis*) and ruffe (*Gymnocephalus cernua*). Eels (*Anguilla anguilla*) are present in all reservoirs, as shown by fyke-net catches (gill-nets are unsuitable fishing gear for eel). Gut-content analysis of adult perch from the Petrusplaat Reservoir in 1987 indicated that their food consisted mainly of zooplankton. It seems possible that predation on zooplankton by perch is one of the reasons that summer levels of phytoplankton in the Petrusplaat Reservoir are now higher than in the past (Van Breemen and Ketelaars, 1995). This reservoir was almost void of fish between 1973–1979 because of caustic-soda dosing for softening (Van Breemen and Ketelaars, 1995). As a result, the pH exceeded 9.0, with $NH_4^+$ being transformed to $NH_3$. Because of high $NH_4^+$ concentrations (maximum 1.0 mg $l^{-1}$) between 1973–1979, the $NH_3$ concentrations (maximum 0.6 mg $l^{-1}$) in the Petrusplaat Reservoir exceeded the $LC_{50}$–level for fish (0.35–0.50 mg $l^{-1}$) most of the time. After this period, the $NH_3$ levels decreased below 0.05 mg $l^{-1}$. With the gradual decrease of $NH_4^+$ concentrations in the River Meuse, and commissioning of the De Gijster Reservoir in 1979, fish reappeared in the Petrusplaat Reservoir in growing numbers. In 1978, gill-nets yielded only 0.6 fish per night. However, this figure rose to 48.7 fish per night by 1987. Beginning in 1980, large shoals of 0+ perch

were observed regularly near the inner embankment of the reservoir. If the hypothesis that the increase of predation on zooplankton by fish is the main factor for increasing algal biomass during the summer is valid, the experiences in the Petrusplaat Reservoir indicate that it may take 5–7 years before the overall effect on the ecosystem becomes manifested. After a transition period of 5–7 years, the establishment of a fish community has changed the previously simple food-web structure of the Petrusplaat Reservoir. Because of their potential negative influence on the herbivorous zooplankton, the future development of predatory cladocerans, mysids and planktivorous fish in this reservoir must be carefully monitored.

## 9.6 ŘÍMOV RESERVOIR (CZECH REPUBLIC)

The Římov Reservoir (Table 9.8) is an example of a small temperate drinking water valley reservoir with an intermediate water retention time and a riverine shape (Fig. 9.16).

The long-term, average theoretical water retention time of this reservoir designates it as a transient type, with its stratification influenced by water flows. The annual average phosphorus retention capability ranges between 60–75% in different years, with the average values for the winter portion of the year usually being much lower than the values for the summer portion of the year. Initially after flooding, the values were as low as 19% in 1980 in connection with the aging process ("trophic surge") during reservoir filling. During the summer, there is a pronounced horizontal variability along the reservoir, with phytoplankton peaks starting in the inflow zone. Later, most of the phytoplankton shifts along the reservoir until it reaches the dam. There is no correspondence between the phytoplankton production and biomass at the dam, because the biomass comes from the inflow region. The critical nutrient for phytoplankton development is phosphorus, which reaches mean annual levels between about 20–40 $\mu g\ l^{-1}$ at the reservoir surface in different years, and CHA values between 20–70 $\mu g\ l^{-1}$, including an exceptional year of 110 $\mu g\ l^{-1}$ (Fig. 9.17; Hejzlar et al., 1989). Phytoplankton has a spring Cryptophyceae and small centric diatom peak, followed by a "clear water phase". It was discovered that, when stratification starts to develop in the waterbody, phosphorus limitation plays a role in the "clear water phase". This is in addition to zooplankton grazing, which is apparently the dominant force (Fig. 9.18; Vyhnálek et al., 1991). The summer phytoplankton composition is primarily large-celled and colonial algae and Cyanobacteria species.

Because this reservoir has been consistently monitored since its construction, the data provide detailed observations of the aging process of reservoirs (Straškrabová and Procházková, 1992; Straškraba et al., 1995). The data support the new idea that aging is not just a consequence of increased nutrient and organic matter content during the first few years of existence. Rather, an important role is also related to differences in development rates of major groups of organisms, and subsequent changes in biotic interactions. Development of fish populations takes a few years (Fig. 9.19; Sed'a and Kubečka, 1997) and, during this time, there is no control of lower trophic elements by predators (i.e., weak or no top–down control).

Considerable attention was focused on the effects of fish on plankton, particularly in regard to biomanipulation techniques. One condition for the use of these techniques is complete knowledge of fish populations in the reservoir. The estimated biomass of

Table 9.8. Basic data for Římov Reservoir

| | |
|---|---|
| *Physical variables* | |
| Location | Czech Republic |
| River (% of total inflow) | Malše (90%) |
| Geographical coordinates | 48°50′N, 14°40′E |
| Surface elevation | 471 m a.s.l. |
| Catchment area (40% forest, 50% intensive agriculture) | 444 $km^2$ |
| Population in the watershed | 13,000 inh. |
| Forested area | 4% |
| Agricultural area | 50 |
| Annual average inflow | 4.4 $m^3 s^{-1}$ |
| Volume | 33.6 million $m^3$ |
| Surface area | 2.1 $km^2$ |
| Maximum depth | 47 m |
| Mean depth | 17 m |
| Length | 13 km |
| Shape: riverine, width | average 300 m; maximum 700 m |
| Outlet structure | 5 outlets between 10 and 35 m above the bottom in distances of about 6.5 m |
| Overflow at | 466.1 m |
| Filled in the year | 1978/1979 |
| Normal rage of annual water level fluctuations | 1 m |
| Theoretical retention time, long term average | 96 days |
| Primary use | drinking water supply (1–2 $m^3 s^{-1}$) |
| Other uses | — |
| Trophic status | mesotrophic–eutrophic |
| *Chemical variables* | |
| Total phosphorus load (1983–1997) | 14–17 g $m^2 yr^{-1}$ |
| Total phosphorus concentration | 34 mg $m^{-3}$ |
| Nitrate nitrogen ($NO_3$-N) | 2440 mg $m^{-3}$ |
| Ammonia nitrogen ($NH_4$-N) | 36 mg $m^{-3}$ |
| Nitrite nitrogen ($NO_2$-N) | 18 mg $m^{-3}$ |
| Chemical oxygen demand (COD) | 14 mg $m^{-3}$ |
| Dissolved organic carbon (DOC) | 6 mg $m^{-3}$ |
| Calcium ($Ca^{2+}$) | 16 mg $m^{-3}$ |
| Magnesium ($Mg^{2+}$) | 4 mg $m^{-3}$ |
| Sulfate ($SO_4^{2-}$) | 30 mg $m^{-3}$ |
| Chloride ($Cl^-$) | 6 mg $m^{-3}$ |
| Alkalinity | 0.5 meq $l^{-1}$ |
| Conductivity at 25°C | 158 μS $cm^{-1}$ |

Table 9.8 (*continued*)

| | |
|---|---|
| *Biological variables* | |
| Chlorophyll-a (0–3 m) | 5 mg $m^{-3}$ |
| Biochemical oxygen demand ($BOD_5$) at surface | 1.2 mg $O_2$ $l^{-1}$ |
| Bacteria cultivated on DAPI, surface | 3.5 million $ml^{-1}$ |
| Bacteria cultivated on beef-pepton agar, surface | 450 CFU $ml^{-1}$ |
| Heterotrophic nannoflagellates, surface | 0.45 $10^3$ $ml^{-1}$ |
| Ciliates, surface | 3.3 $ml^{-1}$ |
| Rotifers (0–3 m) | 0.14 $ml^{-1}$ |
| Nauplii (0–3 m) | 0.002 $ml^{-1}$ |
| Biomass of herbivorous Cladocera (protein N) (0–40 m) | 81 mg $m^{-2}$ |
| Biomass of Copepoda (protein N) (0–40 m) | 77 mg $m^{-2}$ |
| Total zooplankton biomass (protein N) (0–40 m) | 162 mg $m^{-2}$ |
| Most common zooplankton species: *Daphnia galeata, Daphnia cucullata, Bosmina longirostris, Diaphanosoma brachyurum, Cyclops vicinus, Mesocyclops leuckarti, Eudiaptomus gracilis* | |
| Most common phytoplankton species: *Cryptomonas reflexa, C. marssonii, C.curva, Rhodomonas minuta, Synedra ulna, Stephanidiscus hantzchii, Cyclotella pseudostelligera, Fragilaria crotonensis, Aulacoseira granulata, Coelastrum microsporum, Planktosphaeria gelatinosa, Eudorina elegans, Anabaena mendotae, Microcystis aeruginosa, Aphanizomenon flos aquae, Anabaena cf. spiroides* | |
| Most common fish species (with biomass over 1 kg $ha^{-1}$, arranged according to their biomass in 1996, detected by split-beam echosounder): *bream Abramis brama; roach Rutilus rutilus; pike Esox lucius; perch Perca fluviatilis; Aspius aspius; ruffe Gymnocephalus cernuus* | |

dominant planktivorous fish (larger than 10 cm) varied between nearly 0 and 250 kg $ha^{-1}$, although up to 650 kg (1982) were present in the aging period. Reproduction of the most common fish species, perch (*Perca fluviatilis*), was intentionally reduced in some years by lowering the reservoir's water level after egg-laying, so that the eggs dried out near the shore. In combination with predatory fish stocking and net fishery, the planktivorous fish biomass was considerably reduced (Fig. 9.19; Sed'a and Kubečka, 1997). Changes in the plankton composition were observed in years with both high and low fish biomass. Specific field experiments were conducted to examine the effects of fish on possible prolongation of the "clear water phase", and lowering of the phytoplankton concentrations, both being favorable for a drinking water supply. An important function of the juvenile fishes was disclosed, with a double beam echosounder being used to gain more exact data about their numbers and biomass. This modern technique provides accurate data about the density and biomass of all fish sizes in the reservoir, particularly when coupled with classic techniques that simultaneously survey species representation. Figure 9.20 shows the comparative evaluation of the functioning of biomanipulation by means of fish control of the presence of large Cladocera in the zooplankton biomass.

The microbial food chain was intensively studied (Fig. 9.21). The results indicate that this previously-neglected component has a major effect on the cycling of matter in the reservoir. The multi-year study and comparison with other reservoirs and lakes has shown

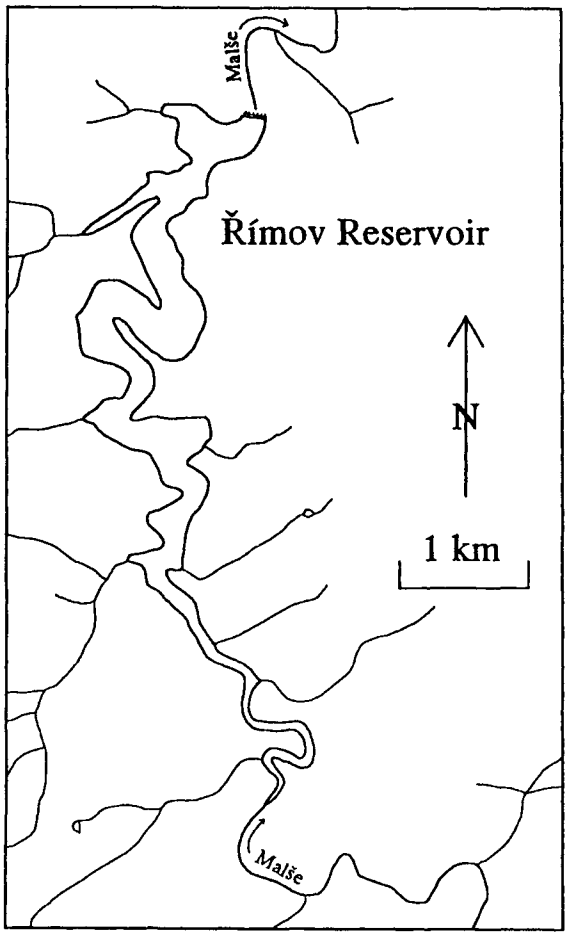



Fig. 9.16. Map of Římov Reservoir showing the riverine character, main inflow—Malše River, and tributary brooks with their bays (from Komárková and Hejzlar, 1996).

seasonal differences in the interrelations between the phytoplankton, its exudate feeding bacteria, heterotrophic nannoflagellates feeding on bacteria and algae, proto-zooplankton predominated by ciliates, meta-zooplankton and fish in different trophic conditions and under different fishstock, as well as in the oxic and anoxic layers (Straškrabová and Šimek, 1993; Straškrabová et al., 1994). In the biomanipulated reservoir, changes were observed during the decrease of the fish biomass from about 450 to below 150 kg $ha^{-1}$, with the biomass of small-size cladocerans and ciliate densities decreasing, while the percentage of larger cladocerans, the numbers of heterotrophic nannoflagellates and bacterial growth rates increased.

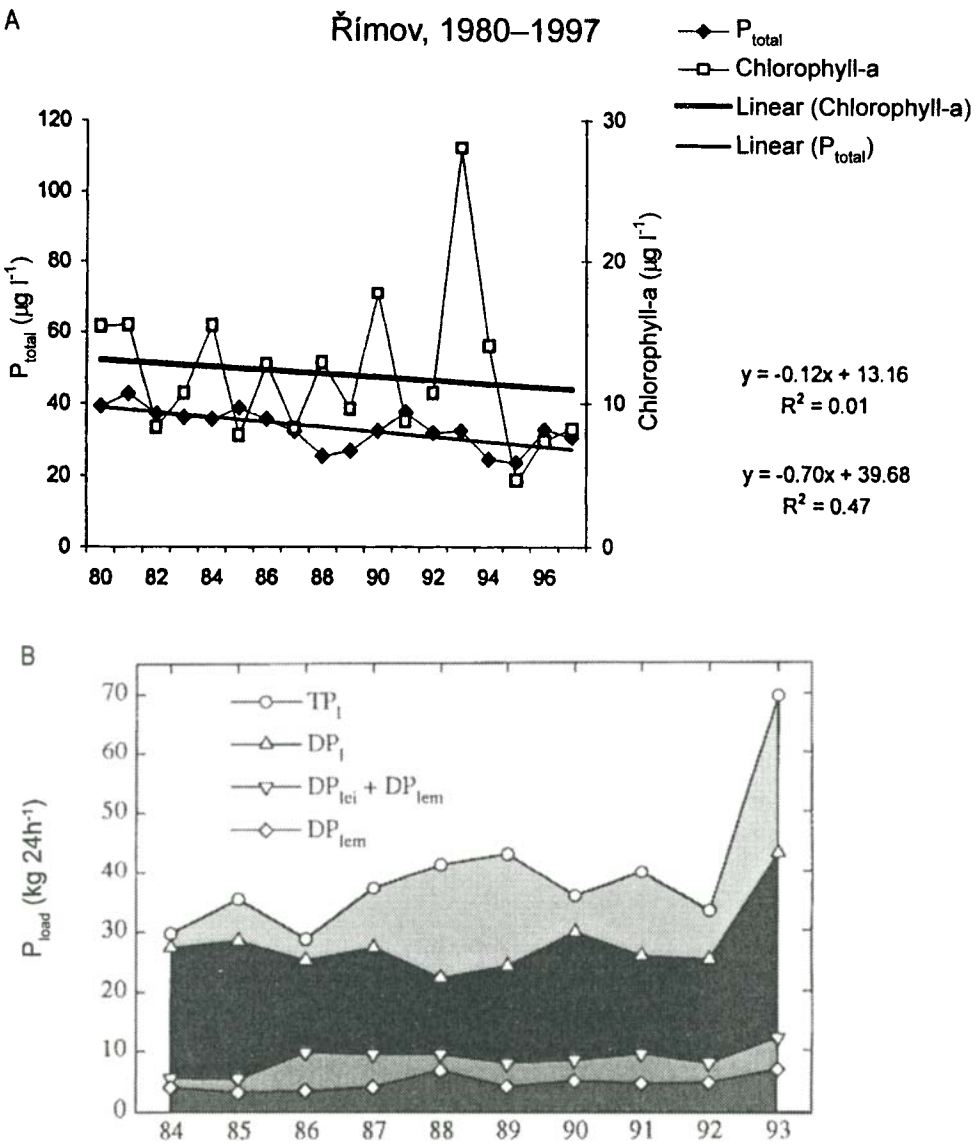



Fig. 9.17. Long-term eutrophication trends in Římov Reservoir. A—Trends of TP and CHA concentrations in the reservoir are decreasing since the reservoir filling, but the amplitude of CHA oscillations among years are increasing (from 38th Annual Report, Hydrobiological Institute, Academy Sciences Czech Republic); B—Phosphorus load of the reservoir, particularly the TP load, shows an increasing tendency (from Komárková and Hejzlar, 1996).

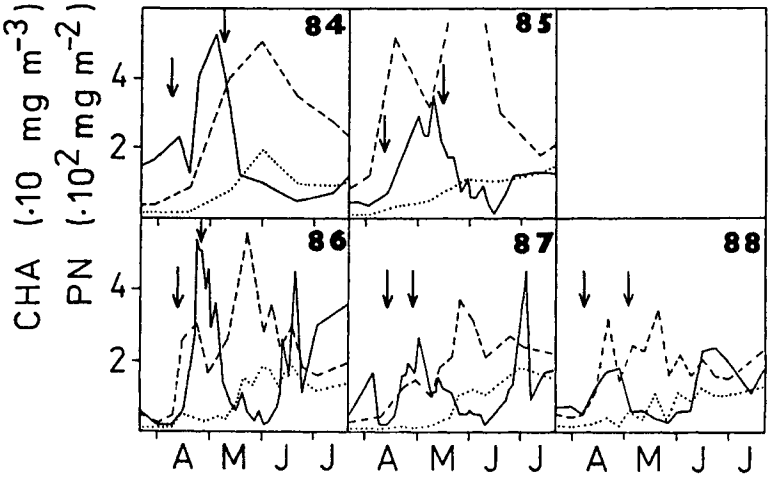



Fig. 9.18. The "clear water phase" in the reservoir, exhibiting low phytoplankton when its control by zooplankton is high, is a regular phenomenon occurring after the onset of summer stratification (from Vyhnálek et al., 1991).

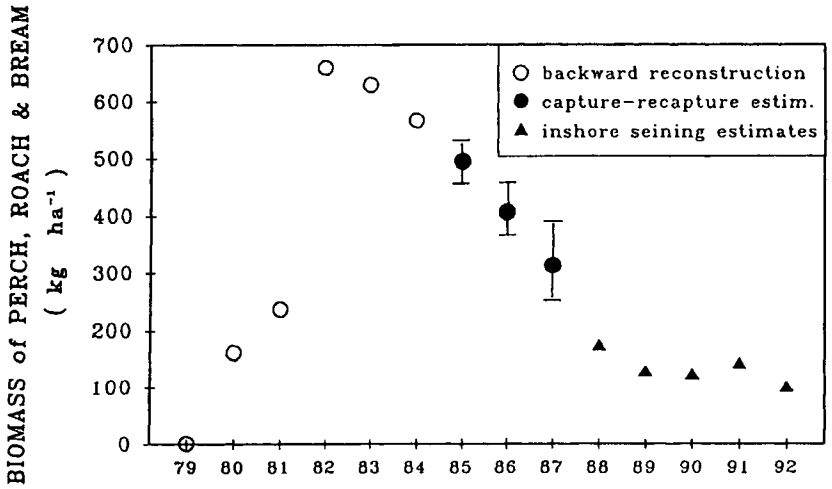



Fig. 9.19. Long-term changes in the annual biomass of three main planktivorous species, perch (*Perca fluviatilis*), roach (*Rutilus rutilus*), and bream (*Blicca bjoerkna*). Only fish older than one year are included (from Sed'a and Kubečka, 1997).

*Management problems*

A new automatic technique ("clean layer") was designed to decrease the quantity of coagulants used for drinking water treatment. The technique is based on the uneven water

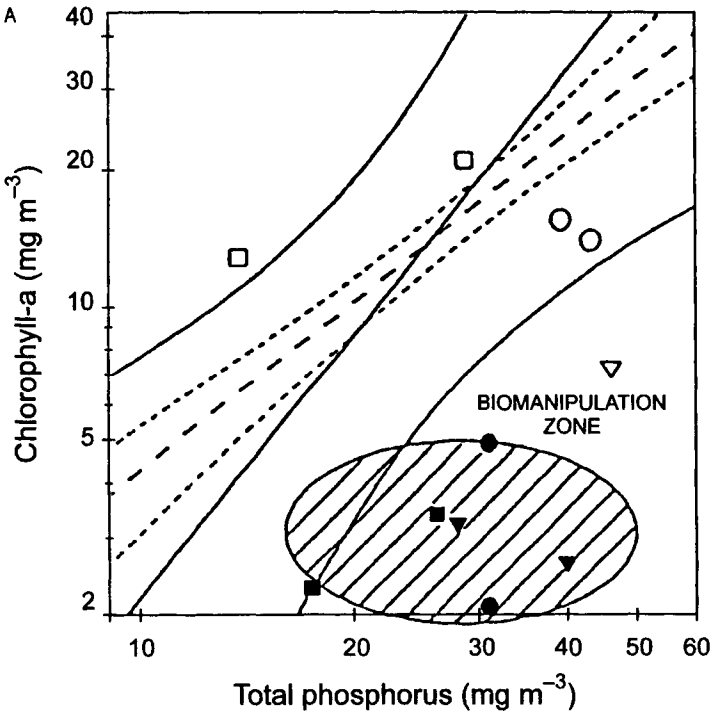



Fig. 9.20. Biomanipulation in lakes and reservoirs. A—Comparison of chlorophyll–phosphorus relationships from unmanipulated reservoirs, and the zone where a significant chlorophyll-a decrease due to biomanipulation was observed in three sets of biomanipulations (Hubenov Reservoir, Round Lake and Haugatjem Lake). Filled symbols indicated the post-manipulation situation in the same waterbodies, while empty symbols are for unmanipulated situation in the same waterbodies.

quality by depth in the reservoir, and the correlation between organic matter content and water layer transparency. Automatic transparency measurements with a submersible instrument enabled the depth of the cleanest water layer to be determined. Because the Římov Reservoir possesses multiple outlets, it was possible to take water for treatment from the optimum water layer.

Successful cooperation between scientists and agencies responsible for reservoir management and water treatment enabled the practical application of research results. However, controversy remained over the use of the reservoir for hydroelectricity production. The optimal strata for water releases differs in regard to the water quality and hydroelectricity production goals. Additional complications are caused by summer cottages along the river below the reservoir, with the owners protesting the release of the cold water strata from the reservoir.

The drinking water from the reservoir suffers from excess algae, associated organic matter and increasing nitrate concentrations. An increase of the water uptake to supply

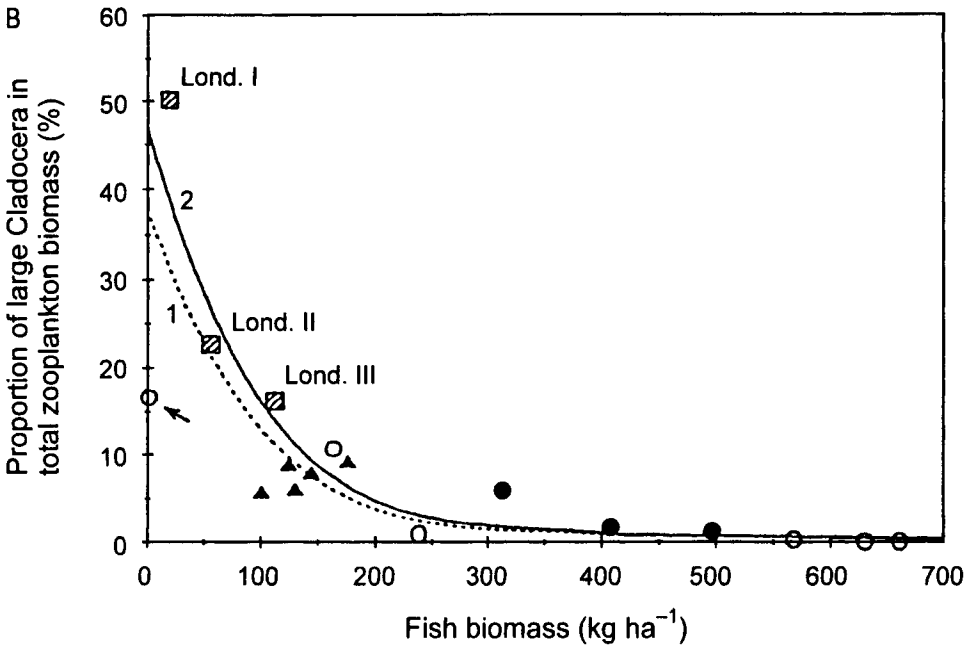



Fig. 9.20 (*continued*). B—Biomanipulation is related to the control of larger zooplankton filtrators by planktivorous fish. Only larger zooplankton appear to be capable of efficiently controlling phytoplankton. The relation is derived from observations in Římov Reservoir and three water supply embankment reservoirs for London (both from Sed'a and Kubečka, 1997).

additional inhabitants was suggested, with the formulation of alternative options for improving the water quality, as follows:

- Construction of a pre-reservoir,
- Tertiary treatment of effluents from the town located upstream of the reservoir,
- Changes in agricultural practices to decrease the nitrogen load, and
- Application of biomanipulation techniques within the reservoir.

In fact, tertiary treatment of effluents and changes in agricultural practices were initiated for the reservoir. Biomanipulation techniques were attempted, although in a very unsatisfactory and ineffective manner. Cultivated fry of predatory perch-pike (*Lucioperca lucioperca*) were introduced to the reservoir in large quantities, although their conditions before release did not guarantee their survival. The construction of a pre-reservoir is part of a larger goal of increasing water capacity. A suggestion, made in regard to the expected construction of an upstream reservoir for water transfer from the larger Vltava River, was to decrease the phosphorus load to the reservoir by construction of two pre-impoundments (in addition to tertiary treatment plant for the major town in the watershed). These measures together will result in a significant improvement in water quality (Hejzlar, 1989).

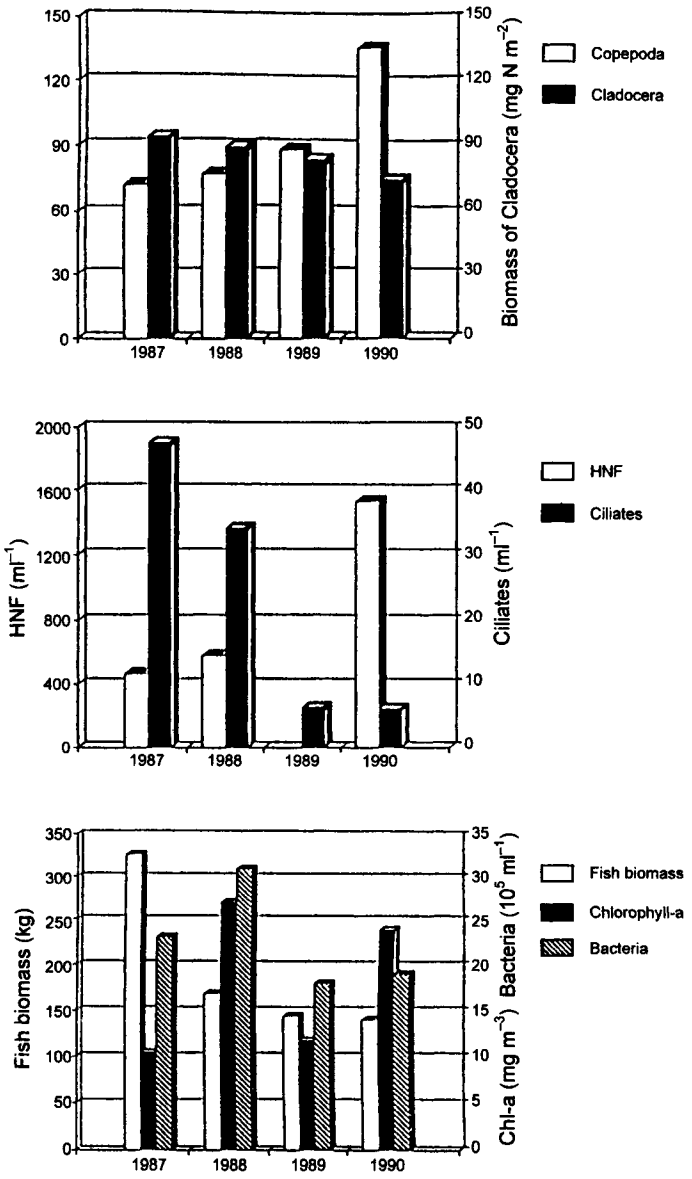



Fig. 9.21. Interannual changes in the annual mean values of microbial and classical food chain in Římov Reservoir. HNF—heterotrophic nannoflagellates. The Copepoda and Cladocera biomass is expressed in nitrogen (from Straškrabová et al., 1994).

This option, however, was never implemented because cheaper solutions for a drinking water supply for a larger region were found.

## 9.7 KARIBA (ZIMBABWE AND ZAMBIA)

The Kariba Reservoir is the world's 3rd largest, man-made lake, as well as being the first large dammed lake in the tropical region (Table 9.9). It was dammed in 1958 and filled completely during 1963.

Preparation for the construction of the reservoir in this vast valley area with a varied topography included the expulsion of more than 50,000 inhabitants of the Batonga tribe,

Table 9.9. Background data for Lake Kariba

| | |
|---|---|
| *Physical variables* | |
| Longitude | 16°28′–18°06′S |
| Longitude | 26°40′–29°03′E |
| Length of Zambezi | 2600 km |
| Area watershed | 1,193,500 km$^2$ |
| Area lake | 5364 km$^2$ |
| Length | 277–300 km |
| Width (max) | 40 km |
| Shoreline lenght | 2668 km |
| $z$ (mean) | 29.2 m |
| $z_{max}$ | 120 m |
| Volume (mean) | 156 km$^3$ |
| Level fluctuation | 2–3 m, up to more than 10 m |
| Surface temp. (max., December–March) | 28–32°C |
| Surface temp. (min., April–July) | 22°C |
| Type of circulation | monomictic (during the period of low surface temperature) |
| *Biological variables* | |
| Macrophytes: *Lagarosiphon ilicifolius* (dominant), *Vallisneria aethiopica*, *Ceratophyllum demersum*, *Najas pectinata*, water fern *Salvinia molesta* | |
| Phytoplankton: Cynophyta 60% (yearly mean): *Cylindrospermopsis raciborskii* (dominant species) | |
| Zooplankton: Dominant crustaceans: *Bosmina longirostris*, *Mesocyclops cf. leuckarti*; rotifers: *Keratella cochlearis*, *Keratella tropica*, *Trichocerca chattoni* | |
| Fish: Decline of stream-dwelling species (belonging to the Cyprinidae and Distichoidae). Increase of more lacustrine cichlids such as *Oreochromis mortimeri* (algal feeder), *Serranochromis condringtonii* (snail feeder), *Tilapia rendalli* (macrophyte feeder), *Synodontis zambezensis* (Mochocidae, Siluriformes, mainly snail and chironomid feeder, see Sanyanga, 1998), *Hydrocynus vittatus* (Characidae), *Limnothrissa miodon*. Altogether more than 40 species | |
| Birds: More than 30 species of potential piscivorous bird are reported (dominated by Cormorant (*Phalacrocorax africanus*) and the Great White Heron (*Egretta alba*)) | |

drawing global attention and criticism. However, it was not a unique event and, since that time, similar conflicts on a larger scale have taken place elsewhere in the world with other large damming projects. Another step in the construction of the reservoir included the removal of about 18% of the forested area within its drainage basin. With the exception of very hard wood trees, the remaining trees were destroyed after two years of flooding (totaling about 25% of the inundated forest area). The persisting trees in the reservoir water basin present an important substratum for algal growth, which has been estimated to total about 90,000 tons dry weight/year (Cronberg et al., 1988). The preparation phase of the reservoir water basin, however, did not include any attempt to assess the eutrophication potential resulting from the terrestrial biomass and the organic soil material that would be inundated. This lack of evaluation eventually resulted in costly and unnecessary strategies for attempting to control eutrophication, and its consequences (i.e., the invasion of the undesirable fern *Salvinia molesta*). Without relevant knowledge, the duration of this eutrophication period could not be predicted, and attempts to control this floating water fern by biological measures were not only unnecessary, but also failed. Finally, during the filling of the basin, a "Noah's Ark" action had to be performed in order to save about 6000 larger mammals. The present state of the reservoir is described in Ramberg et al. (1987).

*Management problems*

As noted above, one of the most obvious and disturbing consequences of the early lake post-construction stage was the explosive growth of the water fern *Salvinia molesta*, in regard to the easily-calculated, but never evaluated, period of reservoir aging. In 1962, almost 25% of the lake area became covered by the water fern, thereby being of great concern in regard to navigation and fisheries (Marshall and Junor, 1981). It was predictable that, after its temporary tropic upsurge due to the flooded terrestrial biomass, Lake Kariba would become a nutrient-poor, soft-water lake, with total phosphorus concentrations between 7–55 $\mu g\ l^{-1}$ and CHA concentrations between 2.5–11 $\mu g\ l^{-1}$ (values from 1986; Cronberg et al., 1988). Nevertheless, costly attempts at chemical and biological control (*Paulinia* sp.—Orthoptera, and *Cyrtobagus singularis*—Coleoptera, both from South America) were tried, but failed. Because of the exhausted nutrient sources, the area covered by the water fern had decreased by 1987 to only 1% of the original value, and has recently disappeared.

A few years after it was filled, the grass *Panicum repens* entered the inundation zone of Lake Kariba. As the dominant plant, it contributed greatly to the return and/or increase of hippos and other grazers (i.e., the antelope Lechwe (*Kobus lechwe*), the African buffalo). An increase in browsers, such as the Greater Kudu (*Tragelaphus strepsipteros*) and the impala (*Aepyceros melampus*), was also observed. The management of the reservoir's fishery deserves special attention with respect to the introduction of the freshwater sardine *Limnothrissa miodon* in 1967–1968 (Marshall et al., 1982). This planktivorous species, which attains its maturity within only 5–6 months, was endemic to Lake Tanganyika. A rapid increase in its numbers contributed greatly to the total annual fish harvest of 35,000–40,000 tons, compared to less than 5000 tons in 1964 (Magadza, 1988). The estimate of the harvest of the freshwater sardine in 1997 was about 32,000 tons. Thus, this sardine ("kapenta") has become very important, especially for the local people, because it

provides cheap and nutritious food for the local fish market (Kautsky et al., 1997). The predatory tigerfish, *Hydrocynus vittatus*, also has increased, compensating for the decline of other species bound to the river environment. At the same time, *Hydrocynus vittatus* is an important attraction for sport fishing and therefore tourism. Moreover, with an increasing population, small-scale fish-pond farming is carried out on the lake shore, and experimental cage-culture have recently been initiated. The main species farmed are cichlids (*Tilapia redalli*, *Oreochromis mossambicus* and also *Oreochromis niloticus*), silver and grass carps (*Hypophthalmichthys molitrix* and *Ctenopharingodon idella*). Plans for an expanded aquaculture exist mainly in Zimbabwe (Kautsky et al., 1997).

Health problems in the Lake Kariba area include malaria and tsetse flies and, related to these vectors, the application of DDT (Magadza, 1988), which has been banned in all industrial countries. Moreover, a striking increase in molluscs, although mainly mussels such as *Caelatura mossambicensis, Corbicula africana*, *Mutela dubia* and *Aspartharia wahlbergi*, also includes vectors of schistosomiasis (*Biomphalaria pfeifferi*, *Bulinus* spp.) (Machena and Kautsky, 1988). During the 1970s, *Bulinus* spp. accounted for only 1% of the snail biomass, which totaled 5000 tons for the whole lake and which was distributed within the macrophyte zones of shallow and sheltered areas. The extent of schistosomiasis infection is not well known, but has been documented as a general feature since the onset of the final filling stage of Lake Kariba. It should also be mentioned that, during the last stage of the filling of the Lake Kariba water basin in 1963, an earthquake occurred, which might have been a consequence of the enormous pressure of 147 billion tons of water in the reservoir. Similar observations have been made elsewhere in connection with the establishment of other large dams.

In conclusion, Lake Kariba has multipurpose uses. In addition to energy production, they include wildlife and water-based activities, recreation, and tourist attraction. Fishing, performed at both a commercial and sporting level, has become most significant, with the lake producing two-thirds of Zimbabwe's total fish production. And with the successful introduction of the freshwater sardine, a most interesting and productive action in the history of fishery has been achieved.

*Impact of flooding*

More recently (February, 2001), however, unusual heavy seasonal rainfall requirement the opening of floodgates of Kariba which caused extensive flooding and put an even greater strain on the Cahora Bassa. Since infrequent and highly regulated flooding in the Zambezi Basins has created a false sense of security for the people living downstream when, before the construction of the dams, they migrated from the farming sites in the floodplain areas before floods arrived.

The unexpected disaster, which caused about 100 casualities and left approximately 100,000 homeless, demonstrates that dams on the Zambezi exacerbated the flood problems, which need adequate strategies for the future such as at least an efficient warning system.

It also must be kept in mind that Lake Kariba was constructed at a time when important long-term effects on the region, such as the natural flooding periods below the dam, were

not yet recognized. When Cahora Bassa Dam (Mozambique) was completed much later (1975), for example, it soon became clear that, in spite of better ecological understanding, it turned out to be possibly the least environmentally-acceptable dam project in Africa. Such impacts as the choking of the river delta's channels with invasive vegetation, loss of grazing land and sections of the mangrove forest and coastal erosion, have resulted in a tremendous decline in the wildlife, including waterfowl and fishery. Moreover, flash floods caused by flood pulses from Cahora Bassa, an example being in 1978, which killed 45 people, displaced more than 200,000, and resulted in a loss of nearly 600 $km^2$ of cropland. During a 1977 workshop on the sustainable use of the Cahora Bassa Dam, it was concluded that the Cahora Bassa Dam's outflow must be managed to simulate the river's natural variability in water flows. However, the implementation of a prescribed flooding program at the scale of the lower Zambezi Basin will be a long-term process.

## 9.8 TIETÊ RIVER RESERVOIR SYSTEM (BRAZIL)

The Tietê River (Fig. 9.22) is a historic waterway. Since the 17th Century, it was used for development of the interior of the present State of São Paulo and Brazil. The river is 700 km long. The construction of reservoirs on the river has been carried out in a sequence since 1963. Six reservoirs were constructed (Barra Bonita, Bariri, Ibitinga, Promissão, Nova Avanhandava, Três Irmãos), with the primary purpose of producing hydroelectricity (Table 9.10).

The changes in the river system are well known and described elsewhere. As an example, the character of the flushing rate variation is shown for Barra Bonita Reservoir, the first in the cascade (Fig. 9.23). The major part of the water flux within about 2000–4000 $m^3\ s^{-1}$ is used for power generation. For short periods of summer floods, excess water is also released from the reservoir as spill water. Major consequences of reservoir construction include (1) hydrological changes, as observed by the relatively level water flow rate below the reservoir (Fig. 9.23), and (2) a decrease in fish diversity. Human activities in the lake's drainage basin cause eutrophication and pollution. Effects of land use in the drainage basin, and such human activities as recreation and navigation, increased the load of suspended material and overall toxicity.

The water quality in the whole reservoir cascade, as measured in the summer period (February, 1998), is presented in Table 9.10 (Barbosa et al., 1999). Longitudinal profiles along the cascade, relative to the situation in the Tietê River at the inflow to Barra Bonita Reservoir is presented in Figure 9.24. Although some water quality variables show an improvement as water passes the cascade, this is not the case for all variables, particularly total inorganic (soluble reactive) phosphorus and chlorophyll-a. The reason is that there are major contributions of these materials to the reservoirs from local watersheds.

The six reservoirs represent an important economic development. Since their construction, the reservoirs are permanently utilized for multiple activities (in addition to the originally-planned hydroelectricity production), including:

- Waterpower,
- Navigation,

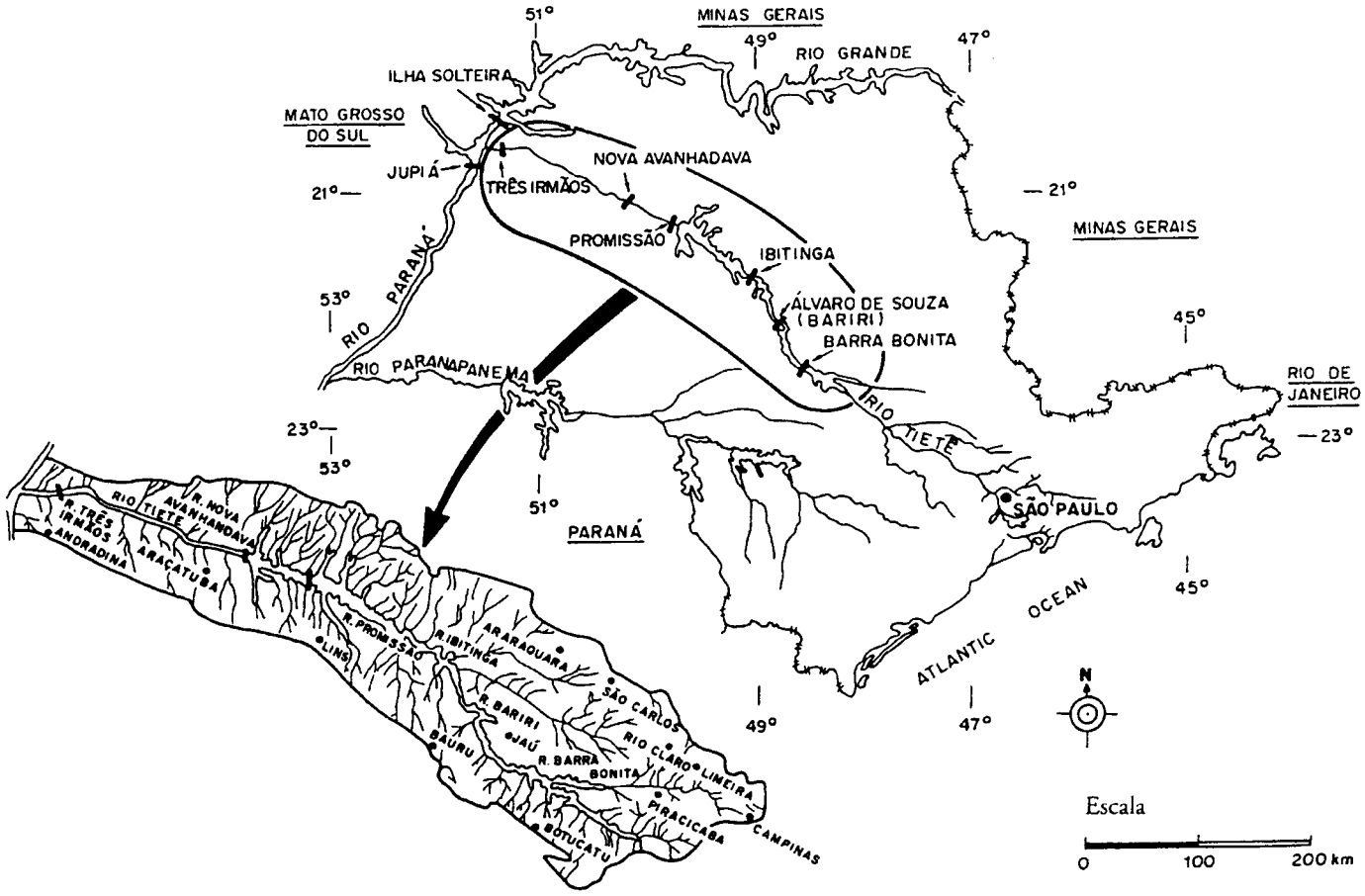



Fig. 9.22. Map of reservoir system on the Tietê River and its location in the São Paulo State, Brazil (from Tundisi et al., 1999a).

Table 9.10. Background data on the cascade of reservoirs on River Tietê, São Paulo State, Brazil

| | B. Bonita | Bariri | Ibitinga | Promisao | Anhandava | Tres Irmaos | Jupia |
|---|---|---|---|---|---|---|---|
| *Physical variables* | | | | | | | |
| 1 | 310 | 55 | 115 | 605 | 210 | 817 | 330 |
| 2 | 10.1 | 8.6 | 8.6 | 14.0 | 13.0 | 17.2 | 11.2 |
| 3 | 37–137 | 7–24 | 12–43 | 124–458 | 32–119 | 166–615 | |
| 4 | 1964 | 1969 | 1969 | 1975 | 1985 | 1991 | 1974 |
| *Chemical variables* | | | | | | | |
| 5 | 62.6 | 87.0 | 36.9 | 34.1 | 23.6 | 29.9 | 21.8 |
| 6 | 4.73 | 5.77 | 15.16 | 14.72 | 4.27 | 3.03 | 5.59 |
| 7 | 3.690 | 2.750 | 1.550 | 1.250 | 760 | 930 | 210 |
| 8 | 299.4 | 84.6 | 112.3 | 77.7 | 29.6 | 9.3 | 16.8 |
| 9 | 194.4 | 96.1 | 23 | 17.3 | 7.8 | 2.5 | 2.8 |
| 10 | 786.4 | 51.2 | 26.4 | 12.5 | 23.7 | 33.8 | 23.0 |
| 11 | 1,280.2 | 231.9 | 161.8 | 107.5 | 61.1 | 45.5 | 42.6 |
| 12 | 3.91 | 5.97 | 5.90 | 5.44 | 5.73 | 6.12 | 6.74 |
| 13 | 4.4–5.0 | 3.3–7.0 | 2.6–5.6 | 9.0–9.3 | 5.9–6.8 | 6.8–6.9 | 7.6–8.0 |
| 14 | 18.2 | 7.3 | 4.8 | 4.2 | 5.4 | 6.2 | 3.5 |
| 15 | 8.6 | 3.4 | 1.7 | 2.3 | 2.4 | 2.3 | 1.2 |
| 16 | 9.58 | 3.88 | 3.1 | 3.13 | 2.97 | 3.83 | 2.2 |
| 17 | 270.7 | 40.2 | 10.7 | 9.7 | 14.3 | 15.0 | 7.6 |
| 18 | 59.0 | 31.6 | 42.0 | 34.6 | 32.2 | 31.1 | 9.7 |
| *Biological variables* | | | | | | | |
| 19 | 57 | 65 | 65 | 95 | 64 | 51 | 129 |
| 20 | 3.18 | 1.88 | 1.78 | 1.64 | 2.53 | 1.09 | 2.53 |
| 21 | 42.6 | 55.8 | 54.9 | 17.1 | 11.0 | 6.7 | 4.5 |
| 22 | 38 | | | | | | |
| 23 | 34 | 75 | 78 | 12 | 9 | | |
| 24 | | | | 73 | 67 | 86 | 61 |

1—Area ($km^2$); 2—Average depth, $z_{av}$ (m); 3—Water retention time (days) monthly basis; 4—Reservoir completion year; 5—Total phosphorus ($\mu g\ l^{-1}$); 6—Total inorganic phosphorus ($\mu g\ l^{-1}$); 7—Total nitrogen ($\mu g\ l^{-1}$); 8—Nitrate nitrogen, $NO_3$-N ($\mu g\ l^{-1}$); 9—Nitrite nitrogen, $NO_2$-N ($\mu g\ l^{-1}$); 10—Ammonia nitrogen, $NH_4^+$-N ($\mu g\ l^{-1}$); 11—Total inorganic nitrogen ($\mu g\ l^{-1}$); 12—Soluble reactive silica ($mg\ l^{-1}$); 13—Dissolved oxygen, $O_2$ in upper 5 m ($mg\ l^{-1}$); 14—Total suspended solids ($mg\ l^{-1}$); 15—Inorganic suspended solids ($mg\ l^{-1}$); 16—Organic suspended matter ($mg\ l^{-1}$); 17—Dissolved N/P ratio; 18—TN/TP ratio; 19—Phytoplankton species number; 20—Shannon diversity index; 21—Chlorophyll-a ($\mu g\ l^{-1}$); 22—% unicellular centric diatoms; 23—% *Microcystis*; 24—% *Coelastrum*.

- Water supply,
- Recreation,
- Fisheries,
- Aquaculture.

There is a concentration of approximately 50 million people in the Tietê River drainage basin, particularly in its headwater areas. There are vast urban areas in which water scarcity is becoming an urgent problem. The costs of water treatment are increasing 3–4 fold,

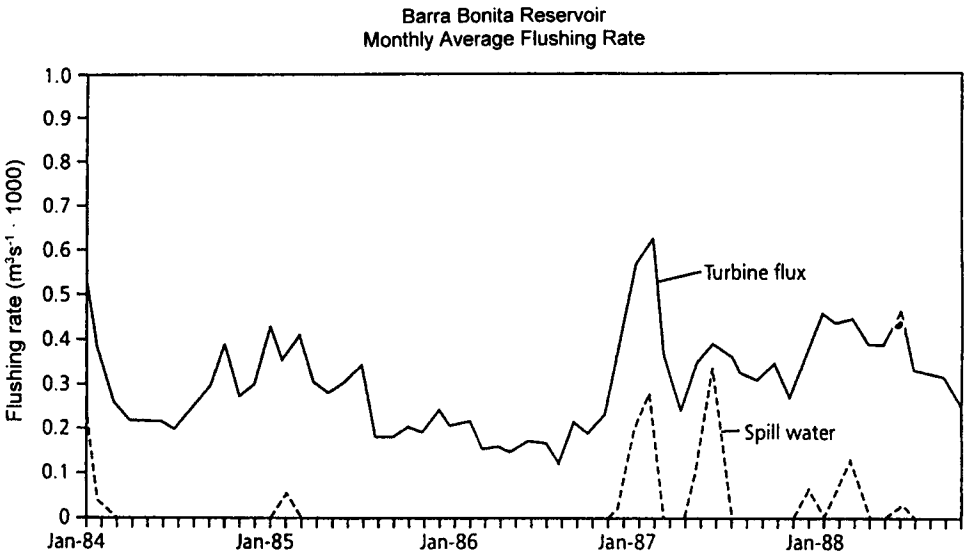



Fig. 9.23. Variation of outflow rates in Barra Bonita Reservoir as an example of the seasonal differences in functioning (from Tundisi et al., 1999b).

because of the decreasing water quality. Thus, the management of water quality for supplying cheaper water for domestic use is an important issue.

The management of these cascades of reservoirs in the Tietê River is a complex operation because of the multidisciplinary processes that must be addressed, including:

- *Watershed management and recovery*—Including control of erosion and the transport of top soil, through reforestation with native species; control of nonpoint pollutant sources; river regeneration and recovery by management of river banks and re-aeration; construction of pre-impoundments on rivers to solve problems of sediment transport and increasing the area of artificial wetlands.
- *Reservoir water quality management*—Management of the reservoir littoral zones; controlling and maintaining macrophyte stands; controlling the water retention time.
- *Modelling*—Control of water quality and eutrophication is of paramount importance in the Tietê River reservoirs. The introduction of ecological and mathematical modelling is an important tool for obtaining a prognosis of the system development. Eutrophication modelling, and estimation of reservoir water quality deterioration in the cascade, also is important since the six reservoirs downstream are much less eutrophic than the upstream Barra Bonita and Bariri Reservoirs.
- *Perspectives*—Navigation, tourism, recreation and aquaculture will be the biggest development activities for the Tietê River reservoirs in the next ten years. An estimate of the economic investment in these activities is approximately US $10 billion over the next ten years. This will not only increase the complexity of the processes related to multiple uses of the reservoirs, but also requires an investment in the management and controlling

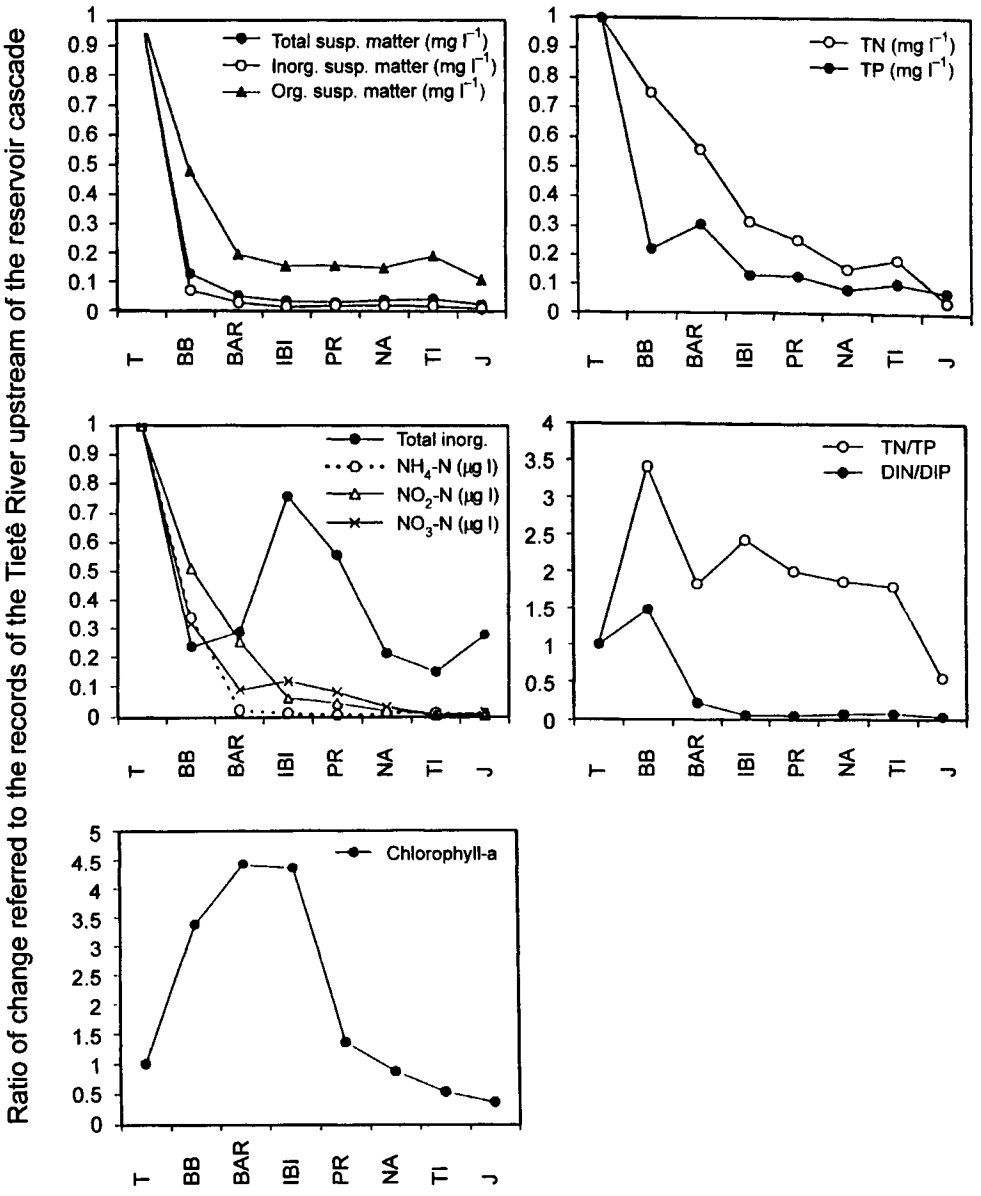



Fig. 9.24. Longitudinal profiles along Tietê River reservoir cascade. Sampling points are in vicinity of each dam. The values are relative to the situation in the river inflow to the cascade (from Barbosa et al., 1999).

of eutrophication and pollution of the reservoirs. During the year 2000 and beyond, it is expected that the Polluter-Pays-Principle will be introduced to water use, with industries contributing to water quality improvement not only within the context of their wastewater, but also to the overall management of the water quality in the upstream portion of the Tietê River drainage basin. This is a multidisciplinary task that will require involvement of many disciplines and specialists with good field knowledge of these reservoir systems.

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*Chapter 10*

# SYNOPSIS

The goal of this final chapter is to stress the most important conclusions arising from the discussions of the previous chapters. Most dealt with natural and man-made lakes (reservoirs) simultaneously, and when the term "lakes" was used without further specification, it referred to both natural lakes and reservoirs. In contrast, Chapter 6 dealt explicitly with the unique characteristics of reservoirs, as well as the differences between them and natural lakes. These features and differences must be considered in attempting to manage reservoir water quality.

## 10.1 LAKES AND RESERVOIRS AS WATER RESOURCES

By the year 2000, the human population will require about 5500 $km^3$ of water each year, with a great portion of this volume to be taken from lakes and reservoirs. However, lakes are not only a source of water for human needs. In fact, humans have many contacts with lakes, be it for recreational purposes or even just living close to them. In a deteriorating world environment, we have concerns about water not only contributing to worsening human health, but also recognize that visual factors also have a role. Lakes and reservoirs are valuable, picturesque landscape features. The willingness of people to pay a premium for good water quality continues to rise, with the prices of lakefront properties being about 10–15% greater than of non-lakefront properties. An additional premium is paid for being able to live next to clear water lakes.

Because the quantitative and health safety concerns about water are still dominant in a large part of the world, the authors identify approaches that are considered useful for overall water quality management. Climate change occurring around the world obviously makes the water situation even more critical, not only because of an overall decrease of humidity, but also because of highly-increased weather variability, resulting in higher floods and more severe droughts, with negative consequences for water quality. Thus, the range of concerns comprising water quality management are broadening. These authors attempt to cope with this tendency by stressing both the immediate, everyday concerns of water quality management, as well as the more complex issues. The approach discussed here focuses on moving from classical technology to ecotechnology, the use of nature-friendly technologies with a broader focus, from correction to prevention, from "end-of-pipe" to "start-of-pipe", from a local to global perspective.

The economic value of water is not fully recognized by economists. Some attempts to estimate the value of "ecosystem services" that rivers, lakes and reservoirs provide free-of-charge to humanity have resulted in an astonishing average global value of US $1.7 trillion

per year, which is about 10% of the annual world gross national product. This estimate includes both the direct costs (sale of water for human consumption, services provided by self-purification and production of food items, transportation, etc.) and the indirect costs (e.g., hydroelectric power generation). It does not, however, cover the cost of environmental deterioration resulting from changes in water flows, habitat modifications resulting from dam construction, and decreasing quality of outflowing rivers resulting from increased recreational and aesthetic use of water, etc. New approaches to the economic valuation of water, including keeping it clean in the first place, are being developed. The Polluter-Pays-Principle, marketable pollution permits, green taxes and green auditing, and environmental impact assessment for new water developments, are used in developed countries. However, such measures are only just beginning to be used in the developing world. The role of non-governmental organizations (NGOs) also can be very positive in protecting water supplies in general, and lakes in particular, as well as in mobilizing water authorities to protect and conserve water resources.

*Distribution of Lakes and Reservoirs*

It is estimated that there are about 3–6 million natural lakes in the world, covering an area of about 1.6–3 million $km^2$. There is no similar estimate for reservoirs. The geographical distribution of lakes and reservoirs is extremely uneven, both with respect to latitude and altitude, as well as continentality. There is generally more water in the vicinity of sea coasts. However, coastal lakes are subject to salinity problems. Dry regions are subject to general water shortages from multi-year drought cycles, many of which also are irregular in occurrence. Geography also can exert major effects on lake physics, chemistry and biology, since lake mixing types are driven by climatic variables. Knowledge of macro- and particularly mesogeographic features is important in managing water quality, since many lake water quality features can be deduced from their geographic location. The geographic differences also lead to the conclusion that the application of experiences from temperate and hydrologically-balanced regions is not necessarily directly applicable to waterbodies in tropical or arid regions. Rather, the specifics of lake behavior in different regions also must be taken into account. As an example, one basic characteristic of lakes in tropical regions is that their daily biological and chemical cycles are usually most significant, while the annual water quality cycles are of more significance for temperate-region lakes. However, even more detailed geographic distinction than temperate versus tropical location also must be considered in the management of lake water quantity and quality.

*Uses and Abuses of Lakes and Reservoirs*

Individual uses of lakes and reservoirs (Box 1) are treated separately by the authors. However, the reality is that most lakes and reservoirs are used for multiple purposes. This results in the creation of conflicts among various user groups, and the creation of complex water quality management problems. Thus, the positive features of water use are treated simultaneously with their negative impacts. As discussed further below, therefore, wise lake

Box 1. Uses and functions of lakes and reservoirs

- Drinking water
- Irrigation
- Flood control
- Production of fish and other useful organisms
- Mining
- Fire- and ice-ponds
- Urban reservoirs
- Energy
- Industry
- Low-energy water purifiers
- Traffic on lakes and along lakeshores and tributaries
- Recreation
- Conservation and biodiversity
- Training and education

management must be directed to accentuating the positive features, while also decreasing the negative impacts.

*Water supply* from natural and man-made lakes is increasing in proportion. Examples are Lake Biwa in Japan supplying water for 13 million people, the European city of London being supplied by a system of embankment reservoirs, and the great world megacity São Paulo getting its water from a system of valley reservoirs.

*Irrigation* with lake water is growing, creating increasing water quality concerns. One facet of the problem is related to water losses resulting from evaporation and evapotranspiration, and another to salinization. The extensive environmental damage resulting from diversion of the tributary flows to the Aral Sea provides a dramatic example of the need to re-evaluate irrigation projects from both a water quantity and water quality perspective.

*Flood control* is exerted naturally by some lakes, with water levels decreasing during dry periods. Many reservoirs are specifically designed for flood control. Unresolved management problems are related primarily to the insufficient natural water retention of present man-made landscapes. Deforestation, destroyed wetlands, artificially straightened and canalized rivers, and large percentages of the land surface made impermeable in urban regions, significantly decrease the water-holding capacity of the land surface.

*Aquaculture* activities in fresh and brackish water is rising, with more species of fish and wild aquatic animals, including alligators, capybaras and other higher animals, a number of domestic birds and various invertebrates, being exploited and cultivated. High organic pollution is caused by intensive aquaculture. The widespread invasion of exotic species into waterbodies is often driven by positive intentions, but unfortunately its consequences are predominantly negative. In a new environment, the introduced species are free from the balanced competition and natural controls existing in their original habitats. As a result, they can proliferate rapidly in the new environment, often destroying native species. The results are most often negative. Thus, these authors conclude that the introduction of nonnative fish and other organisms must be strictly controlled, and in most cases avoided

completely. See exception, Lake Kariba (Chapter 9). Our knowledge of species and habitat interactions is still insufficient to be able to predict in advance all the complex interrelations between native and nonnative organisms that can lead to success or catastrophic consequences, upon the introduction of the latter to a new environment.

*Hydroenergy generation* is considered the least-polluting means of generating energy. However, it also is not without negative environmental effects, related primarily to large hydroelectric projects. Another use of lakes in regard to energy production is their use as sources of cooling water for power plants, with the negative impact being caused by high water losses due to evaporation, and by pollution resulting from the concentration of materials in the water as a result of the evaporation.

*Recreation and transport.* There is a general tendency for an increase from the current approximately 13% of all economic spending devoted to travel and tourism. The appreciation of lakes and reservoirs as sites for recreational activities is increasing. A particular requirement for recreational needs is to protect the lake shores and lake fringes. Otherwise, the recreational value of lakes and reservoirs can rapidly decrease.

*Transportation*, in both an aquatic form (by boats) and a terrestrial form (roads crossing lakes) can create pollution, eutrophication and the unintentional introduction of exotic species to lakes and reservoirs.

*Urban lakes* receive particular attention because of their major management problems, including protection from pollution from the surrounding city, as well as from various lake-related activities, multiple uses of urban lakes, and conflicts among the various users.

*Conservation and biodiversity.* More than 100 lakes and wetlands are among the Biosphere Reserves of UNESCO within the Project Aqua. Efficient legislation should be the main management concern for these waterbodies.

## 10.2 POOR LAKE MANAGEMENT

This section discusses major causes of bad lake management (Box 2), focusing mainly on topographic requirements that have not been achieved, such as the location of new construction outside risk zones along the shoreline or close to the mouth of the inflowing river(s), protection of at least 30% of the area of nearshore zones for self purification, creation of a 50–100 m protected shore zone, the avoidance of conflicting activities in the same location, and proper management of submerged vegetation and avoidance of adverse methods.

Box 2. Major causes of poor lake management

- Rapidly-growing population in sensitive drainage basins
- Rapidly-growing tourism and recreation on lake shorelines, political pressures and profit motivation for actions, poor financial capacity, and lack of awareness of serious problems because of poor education and training
- Complex, multinational problems requiring regional cooperation (pollution in multinational drainage basins, acidification, climatic change, etc.)

*Monitoring Lake Water Quality*

A typical feature of lake, and water quality monitoring in general, is the "data rich, but information poor syndrome". Monitoring often consists of extensive sampling and determination programs. However, there often is limited utilization of the results because they are not translated into useful information. To avoid this situation, monitoring is treated as a system of activities, with water sample collection being only one. Emphasis is on obtaining maximum information for management purposes with a minimum effort. This can be achieved by distinguishing and analyzing the various monitoring components and their interactions (Box 3).

Monitoring goals often vary and, therefore, must be specified a priori. The monitoring goals are designed for a number of reasons, including:

- Finding the sources of pollution or water quality deterioration,
- Detecting trends of water quality changes resulting from industrial production and land use development, and
- Serving as a background for determining pollution fines, or for estimating the capacity of a receiving waterbody for a given type of pollution.

In each case, the monitoring system components have to be carefully selected to contribute to this goal. A generalized framework for monitoring, based on the type of water use, is suggested in Box 4.

Sound evaluation of each step is essential. A very sensitive, time-consuming method used in a highly variable environment is typically a waste of time. A cheaper method that enables one to sample more stations and detect major pollutant sources is recommended in this case.

*Indicators and classifications.* The systems of water quality indicators used for flowing water usually are not useful for lakes. Thus, water quality indicators are different for rivers and lakes. For lakes and reservoirs, indicators are based more on chemical composition, biological trophic state indicators, and nutrient concentrations responsible for the resulting trophic state.

Box 3. Components to be considered in developing lake monitoring systems

- Determining the specific monitoring goals
- Identifying the relevant indicators of the state of the environment (i.e., the variables to be measured, and the methods for their determination) that will provide information useful for improving the situation
- Developing a sampling schedule (timing, localities, depths, etc.), consistent with reasonable expectations, that leave some time for preliminary evaluation, and yet enable one to derive sound conclusions
- Determining the type of sampling (manual, use of electronic probes, automatic recording, etc.) appropriate for the monitoring goals
- Sample preservation, transport and elaboration
- Information extraction

Box 4. Suggested framework for monitoring

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

Approaches to water quality evaluation are as follows:

- To evaluate individual criteria,
- To use combined, complex, multi-metric, multi-criteria systems.

For most lake and reservoir problems, the first approach is recommended, since it provides detailed information for management decisions. Indices are appropriate for broad comparisons of the state of different waterbodies. However, they do not allow for recognizing pollutant sources, or for performing appropriate corrective management actions. The use of indices may require the same sampling and analytical effort as the evaluation of individual criteria, but the information is lost when the data is condensed into one expression or index number.

Most of the more commonly-used indices are geographically-conditioned, with restricted generality. Thus, it often is impossible or useless to use a classification system for a lake located outside of the area for which the index was developed and/or the goal for which it was developed. Trophic indicators are examples of a classification scheme that would be inappropriate for evaluating waterbodies characterized by naturally-high phytoplankton primary production and chlorophyll levels (e.g., tropical and shallow waterbodies). These indicators would classify almost all such waterbodies in the highest trophic category, thereby not enabling any specific management decisions. Another example is the inappropriate application of an indicator system derived from natural lakes to evaluate

reservoirs. Because of the greater turbidity, larger nutrient loads and the effects of water retention time on reservoir water quality, as well as the effects of reservoir operation, systems derived from natural lakes are of limited value for reservoirs.

In addition to classical water quality indicators, newly-developed indicators of ecological quality, which also address conditions for aquatic organisms (particularly fish), are emphasized. Ecotoxicological indicators also are presented. In addition to classic toxicity tests, indicators are based on:

- The use of semipermeable membranes,
- Methods for detecting longer-term sublethal effects,
- Genotoxicity and use of biomarkers,
- Microcosm and mesocosm studies,
- Monitoring of long-term consequences of contaminants in nature, based on species diversity,
- Predictions of toxicity based on molecular structure,
- Tests for changes of morphologic or morphometric characteristics of organisms,
- Detection of pathological individuals.

Thus, a critical evaluation of different approaches to aquatic toxicity is possible.

*Automatic monitoring* systems are of two main types:

- Depth recording (i.e., monitoring by lowering an electronic probe into different water depths),
- Recording over time.

Depth-profiling instruments are of two types. The first is of slow reaction, lowering an electronic probe to a certain water depth and manually starting the record. The second is a free-falling probe, capable of automatically recording the water quality profile with a very high depth resolution (to millimeters). It is stressed that electronic probes can become unreliable if not frequently calibrated. The reaction time of the probe also must be considered, otherwise erroneous results may be obtained.

*Remote sensing* is the use of satellite and aerial imaginary to help water quality management decisions, by providing information on:

- Variables such as the temperature distribution at the lake surface,
- Horizontal distribution of chlorophyll-a,
- Areas of low and high phytoplankton concentrations,
- Horizontal distribution of Cyanobacteria blooms,
- Areas of high and low turbidity,
- The horizontal displacement of river plumes carrying pollutants and suspended material into a waterbody,
- Areas of concentrations of suspended inorganic and organic matter,
- Location of mass fish kills in a lake.

With careful analysis and correlation with other variables (e.g., chlorophyll-a, temperature, transparency or adsorption coefficients), the distribution of total phosphorus and nitrogen also can be derived for specific waterbodies. The detection of mean or elevated

macrophyte concentrations, areas of severe plant stress, and delineation of groundwater flows into lakes also can be determined in specific situations.

Evaluation of monitored water quality data can be achieved with several data elaboration methods (Box 5).

Conclusions to be derived from monitoring can be divided into three groups (Box 6), each of which was considered in more detail in Chapter 3.

The most important water quality changes within the lake, compared to the inflowing water, include the processes given in Box 7. Methods for quantification of these processes are given and some general comparisons are made possible.

*Management Approaches*

Approaches to water quality management are based on the methodological perspectives of three basic management approaches:

Box 5. Data elaboration methods

- Characterization of the type of statistical distribution of individual water quality variables
- Determination of water quality trends
- Correlation between variables
- Multiple regression
- Spectral characteristics of the data by means of spectral analysis (i.e., time series analysis)
- Cross-spectral analysis of several variables, and detection of time lags between signals
- Creation of statistical input–output models
- Use of data for dynamic models

Box 6. Types of conclusions from monitoring

- Conclusions concerning management of the drainage basin
- Conclusions concerning management of the waterbody
- Conclusions concerning water treatment plants

Box 7. Processes causing water quality changes in lakes

- Mineralization of organic compounds, indicated by a decrease in biochemical oxygen demand (BOD), chemical oxygen demand (COD) and water color
- Improvement of water quality, based on decreased phosphorus, organic substances, and non-dissolved (particulate) matter, due to sedimentation of the particulate matter, and because of dissolved matter becoming particulate in the waterbody (e.g., as algal cells)
- Deterioration of water quality due to intensive decomposition of organic compounds, and the related processes of either releasing or binding certain substances (e.g., release of iron, manganese, ammonia, hydrogen sulfide and nitrites, or the binding of nitrates, especially under anoxic conditions)
- Deterioration of water quality, due to an excessive production of organic matter in the form of phytoplankton (eutrophication)

- Preventive,
- Corrective,
- Sustainable.

The use of preventive ("at-the-source") approaches over the corrective ("end-of-pipe") approaches is emphasized. Preventive methods are typically cheaper, have no negative side effects, are long lasting, and are much closer to the final goal of sustainable water use. The present commonly-used approach of first diluting materials (organic matter, nutrients, metals) in water (thereby creating pollution problems and high costs), and subsequently removing it again prior to water use at even higher cost, is considered completely irrational.

The location for application of management actions is categorized into two main groupings:

- Management options used in the drainage basin,
- Management methods applied in the waterbody (i.e., in-lake approaches).

Although individual options typically are performed either in the drainage basin or in the waterbody, management activities should ideally be integrated (i.e., covering both environments). The "at-the-source" methods belong largely in the *drainage basin* category. The in-lake methods predominantly comprise *corrective* methods. There are only a few in-lake methods belonging to the preventive category (e.g., epilimnetic mixing) and these methods should receive more consideration. Although the "corrective approach" typically is used within the sphere of aquatic resources, water quality managers have an obligation to consider any realistic means for preventing water pollution.

*Approaches for Drainage Basin Management*

The drainage basin-based management approaches considered are listed in Box 8. New developments include life cycle analysis and cleaner technology, approaches used by the industry to increase profits, by facilitating decreased use of water, energy and resources, and by decreasing pollution. These approaches must be strongly fostered by water managers, since they can significantly contribute to saving water supplies and decreasing pollution. The critical causative factors for dominant groups of problems and corresponding approaches also are mentioned.

Wetlands are treated separately as the most universally-applicable approach, as evidenced from their frequency in Box 8. Wetlands appear in two basic forms in Box 9:

- *Natural* (including regenerated or created) *wetlands*—representing ecotones (the natural transitions between lakes and their terrestrial surroundings),
- *Constructed wetlands*—representing purification plants that utilize the cleansing and pollution-retention capacity of vegetation.

Prevention of water pollution can take many forms, some being technical and others being educational in nature. It is stressed that water managers must be active in the educational incentives aimed at convincing both town and country people, farmers, industrialists and politicians about the importance of their daily activities and their decisions in regard to keeping water supplies safe and clean (Box 10).

Box 8. Water quality problems in reservoir drainage basins, and ecotechnological methods applicable for their management

- *Organic pollution*—Clean production, diversion of effluents, purification plants, wetlands
- *Excess nutrients and eutrophication*—Diversion of wastes, tertiary treatment plants, progressive agricultural practices, meadow and riparian forest zones on vegetated banks, natural and constructed wetlands, pre-impoundments at the inflows, Wahnbach phosphorus-reduction plant
- *Eutrophication and oxygen depletion of rivers*—River restoration, re-oxygenation
- *Siltation*—Erosion control, rehabilitation of river banks, reforestation, groundwater recharge, pre-impoundment of inflows
- *Heavy metal contamination*—Clean production, life cycle analysis, reduction of polluted effluents, wetlands
- *Acidification*—Life cycle analysis, clean production, liming, addition of organic matter
- *Salinization*—Improved irrigation practices, decreased fertilizer applications, decreased road salting
- *Decreased biodiversity due to reservoir construction*—Prohibiting introduction of foreign species, reintroduction of native species, maintenance of wetlands as nursery grounds, maintenance of preserved areas for native species

Box 9. Use of natural and constructed wetlands

- Maintenance and preservation of the transition zone between terrestrial ecosystems and the lake ecosystem functions as a buffer zone for pollution, and preserves the species diversity in both ecosystems
- Building and maintaining constructed wetlands is much less costly than building and maintaining classical purification plants
- Wetlands, both natural and constructed, have a high adsorption capacity for many pollutants (e.g., nutrients, heavy metals, toxic organic compounds)
- Wetlands are able to denitrify up to several tons of nitrate-nitrogen per hectare and year
- Recovery of natural wetlands, and creation of constructed wetlands, are crucial abatement methods for diffuse pollutants originating from agriculture
- Constructed wetlands represent an attractive wastewater treatment method for recreational areas adjacent to lakes, for areas with low population densities, and as post-treatment of purified effluents from cities
- The application of quantitatively-based management of all types of wetlands, including the use of models for these ecosystems, is recommended

Box 10. Desirable educational activities of water managers

- Water quality managers must stress the usefulness of cleaner production techniques, and the product life cycle evaluation procedure, and strive for such evaluations. Local water management councils can be very helpful in this direction
- Saving energy has positive consequences for water quality, since energy generation can be the cause of environmental degradation, including water pollution
- Saving water in households improves water quality, since existing wastewater treatment plants function better with less incoming wastes, thereby reducing the need for upgrading, renovation and new construction

Box 11. Methods of agricultural pollution prevention

- Agriculture is one of the major sources of diffuse pollution in many places of the world. Diffuse pollution from agriculture should be minimized. Among other concerns, it is economically beneficial to ensure that expensive fertilizers do not end up unutilized by crops
- Application of fertilizers in the most efficient manner for uptake by crops usually is also most favorable for enhancing good water quality, since it creates the least water pollution. Water quality difficulties are often caused by unutilized (but nevertheless expensive) and lost fertilizer. Agricultural best management practices are the most useful practices for addressing water quality problems
- Losses of fertile soils as a result of improper agricultural practices and other land uses are related to major water pollution problems. It is to the benefit of both agriculture and water quality to minimize erosion by contour plowing, terracing, and leaving soil bare of vegetation for the minimum time
- Forests generally enhance good water quality, while deforestation typically results in water quality deterioration
- The use of plant protection agents must be strictly controlled, because some are highly toxic to aquatic life and humans
- Buffer zones, riparian forests and vegetation strips along stream and lake shorelines, together with wetlands, are efficient in reducing the loads of nutrients and some other chemical compounds to lakes and reservoirs

Agricultural activities represent one of the most severe nonpoint (diffuse) pollution sources. Methods for preventing this type of pollution are reviewed (Box 11). Natural forests generally have positive water quality impacts, although the effects of intensive cultivation of introduced tree species (e.g., *Eucalyptus*), and improper forestry practices and forest fertilization, may be significantly negative.

### *In-Lake Management Methods*

There are a multitude of in-lake management methods (Box 12). Examples for each method are provided, conditions and limitations of their use are enumerated, positive and negative effects mentioned, and some indication of costs given.

Artificial mixing is used for:

- Oxidation of a deoxygenated hypolimnion (*destratification, propeller mixing, hypolimnetic aeration* and *hypolimnetic oxygenation*),
- Improvement of the water quality of specific water layers in a stratified waterbody (*layer aeration*),
- Inhibition of phytoplankton growth (*epilimnetic mixing*).

Epilimnetic mixing is based on decreasing phytoplankton primary production, and increasing its respiration in the water column, by mixing surface water layers to a depth where respiration compensates for production on a daily basis. This method, therefore, belongs in the very efficient *preventive* category. It was successfully used to protect the drinking water supply of London and some major towns in The Netherlands. Another way

Box 12. In-lake management methods

Methods based on artificial mixing and oxygenation:
- Destratification by total mixing of the water column
- Mixing and re-aeration of the hypolimnion
- Epilimnetic mixing
- Layer aeration
- Hypolimnetic oxygenation
- Metalimnetic mixing
- Propeller mixing

Methods based on treating sediments:
- Sediment removal
- Sediment capping
- Sediment oxygenation by the RIPLOX procedure
- Phosphorus precipitation with sediment sealing

Other methods:
- Hydraulic regulation (selective water withdrawal)
- Diversion of polluted inflow water
- Extraction of hypolimnetic water (hypolimnetic siphoning)
- Biomanipulation
- Macrophyte removal

to decrease algal production is by decreasing light penetration into the water column. This has only been performed in a few instances by shading, and by the application of surface covering, soot, and color. The cheapest in-lake option is *hydraulic regulation*, or selecting the water layers with the best water quality for drinking water treatment by:
- Releasing poor-quality water layers near the lake bottom (also achievable with bottom water siphoning),
- Decreasing phytoplankton biomass by releasing surface water layers,
- Using density currents to get rid of by short spikes of water pollution.

Hydraulic regulation can be performed by means of selective offtakes, or their substitution by plastic curtains.

*Biomanipulation* is a procedure based on enhancing the suppression of phytoplankton by zooplankton, and control of fish populations in a manner so as to achieve efficient phytoplankton reduction by zooplankton filtration. This can be achieved by manipulating the fish populations in a lake. However, this method is applicable only when the in-lake phosphorus concentrations are below a certain level. For very eutrophic waterbodies, it is necessary to combine biomanipulation and nutrient reduction by other methods. Successful application of this method is generally more pronounced in shallow and smaller waterbodies, mainly because controlling fish populations is easier under such conditions. The procedure cannot be considered a routine method, however, since it should only be performed with the participation of skilled limnologists. Further, the present experience with biomanipulation is

based primarily on conditions in the northern part of the temperate region, which is characterized by a relatively uniform, well-known species composition of fish and zooplankton populations.

*Macrophyte harvesting* by means of mechanical collection, and by use of plant-eating fish and other vertebrates, is recommended. Successful examples of eradication of tropical floating plants by some species of aquatic insects are available.

Managers may note that little attention was given in this book to the commonly-used corrective method of killing algae with copper sulfate or other algicides. This is because the negative side effects may easily exceed the positive benefits, particularly because the toxic materials used may accumulate in the sediment and organisms. As has been seen in a few cases, a lake may become useless for the desired water uses, and the fish become inedible, after many years of application.

A combination of these approaches is recommended, since one method may not necessarily achieve complete success in all situations or regions.

To address lake *acidification*, two options are considered useful; namely, the addition of lime or organic matter to waterbodies impacted by acid rain. These materials work to neutralize the acidic water.

### *Modelling Water Quality*

Although modelling is an activity best done by specialists, it is important for managers, and for those concerned with water quality in general, to know about their existence, possible uses, capabilities and limitations, and possibly about how to evaluate and apply the modelling results. From this perspective, it is important to understand possible model uses (Box 13).

Box 13. The use of models in water quality management

*General*—To estimate pollution sources in a drainage basin by means of simple calculation models

For lakes and existing reservoirs:

- To predict future water quality conditions resulting from human activities in a drainage basin
- To provide estimates for decisions between different water quality options in long-term planning efforts, and to support short-term operational management decisions
- To optimize sampling schedule investigations and water quality controls, and to assess the environmental risks associated with the discharge of micro-pollutants

Prior to reservoir construction:

- To estimate the loads of major water quality components of rivers entering a reservoir, those existing in the main body of a reservoir, and those in the reservoir outflows
- To provide information for evaluating alternative construction sites, dam heights, and outflow and outlet structures
- To predict the conditions in future reservoirs, and the consequences of different management options for water quality within a drainage basin

Models available for helping management decisions are categorized on the basis of differing criteria, depending on form, scope, and the water quality problems to be solved. The differentiation of models according to their forms is important, as shown in Box 14.

The most advanced modelling approach is represented by the Decision Support Systems (DSS), representing specifically-constructed interactive guides for management decisions. The word *support* in this descriptive name stresses the notion that no decision is made by the model. Rather, quantified information is supplied to a manager, in order to facilitate basing the management decisions on the basis of the maximum background information. In fact, the approach combines all (or at least most) of the other model types named in the list. Extensive databases and connection to monitored information sources are components of such systems. Each DSS has a specific goal, and there is a rapid development of this type of system recently for water quality management.

The specific models directed to different water quality problems are classified into the categories identified in Box 15. Further, the formalization of Environmental Risk Assessments is treated in Chapter 5.

The group of models identified last in Box 15 represents complex lake and reservoir water quality models that have been most recently used in water quality management activities. The relevant DSS for such purposes are enumerated in Chapter 5.

Box 14. Classification of models useful for water quality management

- Simple static calculation models, consisting of algebraic equations or graphs
- Complex dynamic models for analyzing the timing aspects of water quality
- Geographical information systems for analysis of problems requiring spatial resolution
- Prescriptive models for calculating water quality conditions without indicating appropriate management options for a given situation
- Optimization models for incorporating a selection procedure to choose the most suitable option on the basis of a set of criteria
- Expert Systems, using qualitative and quantitative expressions to guide the user toward relevant answers to complex water quality questions
- Decision Support Systems (DSS), representing the most advanced and most helpful decision aids, incorporating various computer software relevant for a specific water quality decision problem facing managers—a computer is typically used with these systems

Box 15. Models applicable to different management problems

- Eutrophication models
- Toxic substance models
- Acidification models
- Wetland models
- Fisheries models
- Biomanipulation models
- Commonly-applied water quality models

*Reservoir Water Quality Management*

Reservoirs are artificial waterbodies constructed for many diverse purposes. Thus, although there are many types, they are often treated from both a theoretical and management perspective, in the same manner as natural lakes. This is not justified, however, because reservoirs are operated for the purposes for which they were constructed, a condition not possible for natural lakes. Human operation of reservoirs has considerable influences on their water quality. Thus, understanding their specific characteristics enables a considerable extension of their management possibilities.

Considering deep river valley reservoirs and deep lakes, basic differences between natural and artificial lakes are specified in Box 16. On the other hand, shallow, unstratified reservoirs, particularly riverine ones, are not much different from shallow lakes. Reservoirs can be categorized into impounded lakes, valley or dam reservoirs, off-river reservoirs and embankment reservoirs.

Stratification, mixing and other variables of deep valley reservoirs depend strongly on the water retention time ($RT$ = volume/flow), in addition to variables that determine lake stratification (Box 17). Quantification of some relations is developed in Chapter 6.

Box 16. Differences between deep reservoirs and deep lakes

- For similar situations, pollution of reservoirs can be higher than for natural lakes, because reservoirs are constructed to have greater drainage basin/waterbody area ratios than lakes, in order to fulfill their intended purposes. Thus, reservoir water is generally more throughflowing than for lakes, and reservoirs have shorter theoretical water retention times
- Longitudinal differentiation of the water quality of deep reservoirs is due to the continuous slope of the original river bottom from the inflow to the dam. Four regions are distinguished:
  *The inflow region* is usually shallow and narrow, characterized by rapid unidirectional flows, with flow rates distributed similarly to that of the river. This is usually the most polluted region
  *The transition region* is where the water flow decreases as the reservoir gets deeper and broader. The river water plunges into deeper water strata. Maximum concentrations of suspended sediments, nutrients and algae occur in this zone, as well as maximum sedimentation
  *The lake region* closest to the dam usually has the best water quality, with vertical stratification characterized according to the rules mentioned above
  *Bays* with the longitudinal pattern observed in the main inflow sometimes repeated to a certain degree, with modifications due to density conditions and inflow intensity and composition
- Water level fluctuations in reservoirs are higher than in natural lakes. The degree of the fluctuations is related to the natural inflow variability and operation of the reservoir
- The phosphorus retention capacity (both particulate and dissolved) of deep reservoirs is very high in both temperate and tropical reservoirs (average around 80%, but can reach more than 90%), and increases with the areal phosphorus load. Some other elements also are retained in the reservoir basin, and the water leaving a reservoir is cleaner than its inflows
- The beginning of a reservoirs existence (period of aging) is characterized by poor water quality for several years. Two causes are shown to determine aging, including external (physical, chemical) and internal (biological)

Box 17. The effects of increasing the water retention time on different variables in deep reservoirs

- Decreasing hydraulic loading and pollution loading
- Increasing degree of stratification up to a water retention time of about one year, with additional dependency on geographic location
- Increasing surface temperature up to a water retention time of about one year, being unaffected thereafter
- Decreasing bottom temperature
- Increasing longitudinal differentiation up to some water retention time, decreasing thereafter
- Decreasing quantity of sediments (at comparable inflow concentrations)
- Asymptotically-increasing nutrient retention, most rapidly at a water retention time less than 200 days
- Increasing phytoplankton biomass up to some water retention time, decreasing thereafter
- Increasing trophic state, followed by a decreasing state
- Increasing occurrence of Cyanobacteria under comparable inflow nutrient concentrations
- Decreasing bottom fauna, but highly dependent on near-bottom oxygen conditions
- Increasing hypolimnetic anoxia increases up to some water retention time, decreasing thereafter
- Increasing zooplankton increases up to some water retention time, decreasing thereafter
- Increasing fish biomass
- Increasing length of period of aging

Management techniques specific to reservoirs include *selective withdrawal* (= hydraulic regulation), which is realized by selective water offtakes or, in the absence of selective offtakes, by plastic curtains. Construction of *pre-impoundments* at the reservoir inflow site decreases pollution of the main reservoir by utilizing the phosphorus retention capacity of the pre-impoundment. Conditions for their efficient functioning are quantified in Chapter 6.

*Level and flow operation* enables water quality improvements. Several approaches exist for the water quality management of outflows. Special attention should be devoted to management of reservoir systems.

*Construction of new reservoirs* typically has both positive and negative consequences for the local population and the outflowing river. Thus, careful consideration of the positive and negative sides of new reservoir construction must be performed by means of Environmental Impact Assessments (EIA).

Based on comparative studies of reservoirs, guidelines for the construction of new reservoirs have been developed, particularly for drinking water reservoirs that are most sensitive to water quality concerns.

### *Integrated Management*

Integrated management is the management of a system of functions under a single general control in a way that seeks a compromise to simultaneously maximize the combined benefits from the individual functions. Guidelines for integrated management are defined in Box 18, and elaborated in Chapter 7.

Ways to facilitate achievement of these general guidelines are presented in Box 19.

Indeed, it is readily evident from the above that a major change in thinking and attitudes is necessary on the part of both water quality specialists and water managers. The rationale for this statement is that traditional water management methods are no longer sufficient in many cases (Box 20).

Possible solutions to promote sustainable water resources are summarized in Box 21.

Box 18. Guidelines for integrated water resource management

- Develop methodologies for integrated river basin management that include land-use management and water planning
- Improve the volume and accuracy of national and global assessments of water resources
- Develop, promulgate, and implement new, innovative approaches to water supply and sewage treatment
- Develop, promulgate, and implement waste minimization and recovery techniques
- Develop low-tech, low-cost treatment options
- Increase application of use-related receiving waste standards
- Develop and apply economic evaluation tools to both environmental costs and benefits
- Inform, educate, and train water professionals and public

Box 19. Realization of integrated water resource management rules

- Identify drainage basin-wide issues
- Prioritize subcatchments for action
- Identify specific areas and variables of concern
- Prioritize pollution sources according to the magnitude and the possibility of management
- Screen types of controls, and propose strategies
- Screen practices, rank and develop the action plan
- Implement actions
- Monitor results and revise strategy
- Audit action plans

Box 20. Reasons for inadequacies of traditional water management methods

- They are very expensive
- They concentrate almost exclusively on "end-of-pipe" solutions
- They are often unable to provide sustainable solutions to problems
- They cannot cope with changing environmental conditions
- They often simply move environmental degradation problems from one place to another, producing new environmental deterioration in the other areas and creating new environmental problems

Box 21. Suggested solutions for achieving sustainable water resources

- Decentralized decisions and management
- Reduced water use
- Recycling water
- Sanitation procedures that do not require water
- Recycling organic matter
- Introducing new wastewater treatment technologies
- Introducing low-tech, nature-friendly methods of ecological engineering and ecotechnology
- Placing more emphasis to the prevention of pollution and deterioration before utilizing purification and other corrective measures
- Emphasizing "start-of-pipe" rather than "end-of-pipe" methods
- Giving more emphasis to imission rather than emission criteria
- Giving more attention to methods for abatement of diffuse pollution sources
- Increasing the use and elaboration of mathematical models for different specific problems and evaluation of various approaches
- Emphasizing the adaptation of approaches to local social, material and other conditions
- Elaborating, introducing and evaluating methods of monitoring "global changes" from the hydrodynamic, chemical and biological perspective
- Increasing the availability of information to managers, and to the local rural population, about the consequences of their activities on water quantity and quality
- Promoting closer collaboration between managers, engineers, scientists and decision-makers
- Increasing community participation
- Encouraging water quality specialists to be more aggressive in stressing the need of "clean technology" and pollution prevention, rather than water purification
- Encouraging application of the "life cycle evaluation" of products, which consists of considering the environmental consequences of a product throughout its entire life cycle (creation to disposal). Considerable financial savings enhance the economic competitiveness of products, accompanied by considerable savings in water and energy resources, and reduced pollution
- Simultaneously evaluating possible alternative options. The capabilities of nontraditional ecotechnological approaches also must be stressed
- In evaluating management options, considering the global, rather than just the local, "cost" (damage) to the environment
- Emphasizing the goal of the long-term horizon sustainability of clean water resources
- Intensifying environmental education

## 10.3 SELECTION OF BEST MANAGEMENT STRATEGIES

A detailed outline to the development of appropriate management strategies is given in Chapter 8, to enable higher-level managers that do not need the technical details given in Chapters 1–7 to utilize this book. Only the most important recommendations concerning management approaches are summarized here (Box 22).

## 10.4 CASE STUDIES

An overview of situation and management problems of a few selected lakes is given in Chapter 9. The case studies (Box 23) cover a broad geographical range, lakes and reser-

Box 22. Recommendations for lake/reservoir water quality management

- Proper environmental strategy usually requires a range of approaches and techniques
- It ideally requires the application of a combination of "end-of-pipe" technologies (environmental technology), ecotechnology, cleaner technology, and environmental legislation to be most successful
- The correct timing of steps is very important. A complete environmental management plan should be established at an early stage, as a means of utilizing the available resources in the most optimal manner
- From an economic perspective, it is very important to consider *prevention* over *correction*, since lake restoration efforts can be extremely expensive
- Because of the complexity of aquatic systems and their problems, sound ecological knowledge is a prerequisite for an ecologically-sound environmental management programme. This is the only appropriate means of avoiding unexpected results from treating the ecosystems of concern

Box 23. Case studies presented

- *Neusiedlersee (Austria and Hungary)*—Very shallow lake overgrown by vegetation; the recreational interest is in conflict with nature protection because the lake has been declared a Biosphere Reserve by UNESCO
- *Great Lakes of North America*—Huge, densely-populated territory where intensive analysis, modelling and control of point and nonpoint source (agriculture, urban areas) pollution, conducted by the U.S.–Canada Pollution from Land Use Activities Reference Group (PLUARG) has resulted in considerable eutrophication abatement
- *Lake Fure (Denmark)*—Lake located in a highly urbanized area, an example of major water quality improvement based on community activities
- *Lake Ichkeul (Tunisia)*—Also a UNESCO Biosphere Reserve, a shallow lake endangered by increasing salinity due to the damming of some of its inflows
- *Biesbosch reservoirs (The Netherlands)*—A system of three embankment reservoirs, supplying drinking water to major towns. Problems with water pumped from lower reach of an European River are successfully solved with a combination of engineering and biological methods
- *Rímov Reservoir (Czech Republic)*—A small, stratified drinking water supply reservoir with multiple outlets, with application of hydraulic operation and biomanipulation management methods.
- *Lake Kariba (Zimbabwe and Zambia)*—One of the largest reservoirs exposed to an invasion of macrophytes over many years. The reservoir also is a rare example of a successful fish introduction
- *The River Tietê Reservoir system (Brazil, São Paulo State)*—Six reservoirs with 50 million people in the headwaters of the river; an example of management problems in an economically rapidly-developing region

voirs, developed and developing countries, temperate and tropical regions, and shallow and deep waterbodies, used for different purposes. The situations and management solutions used are given for each example.

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