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

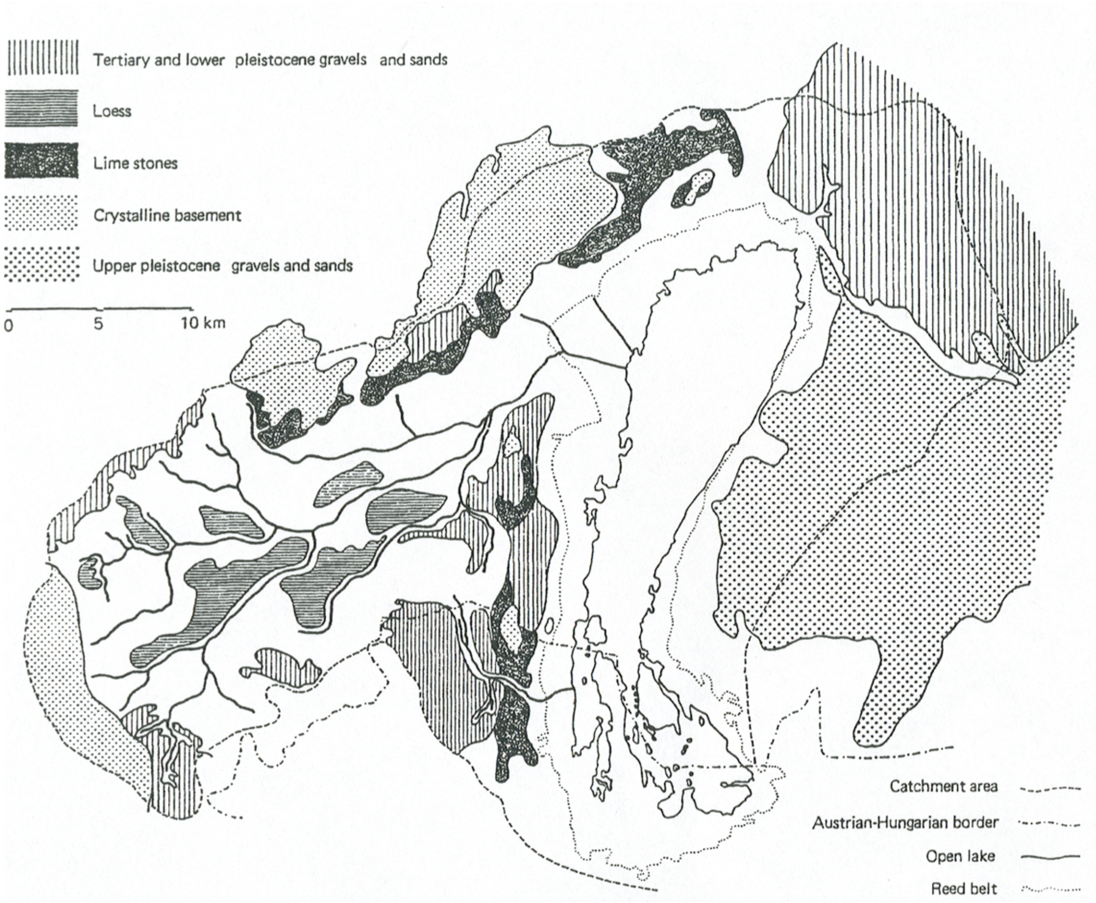

Fig. 9.1. Map of Neusiedlersee, showing extent of macrophyte beds until 1845 and since 1870. The Wulka River flows into the lake from the upper end. (Orig.)

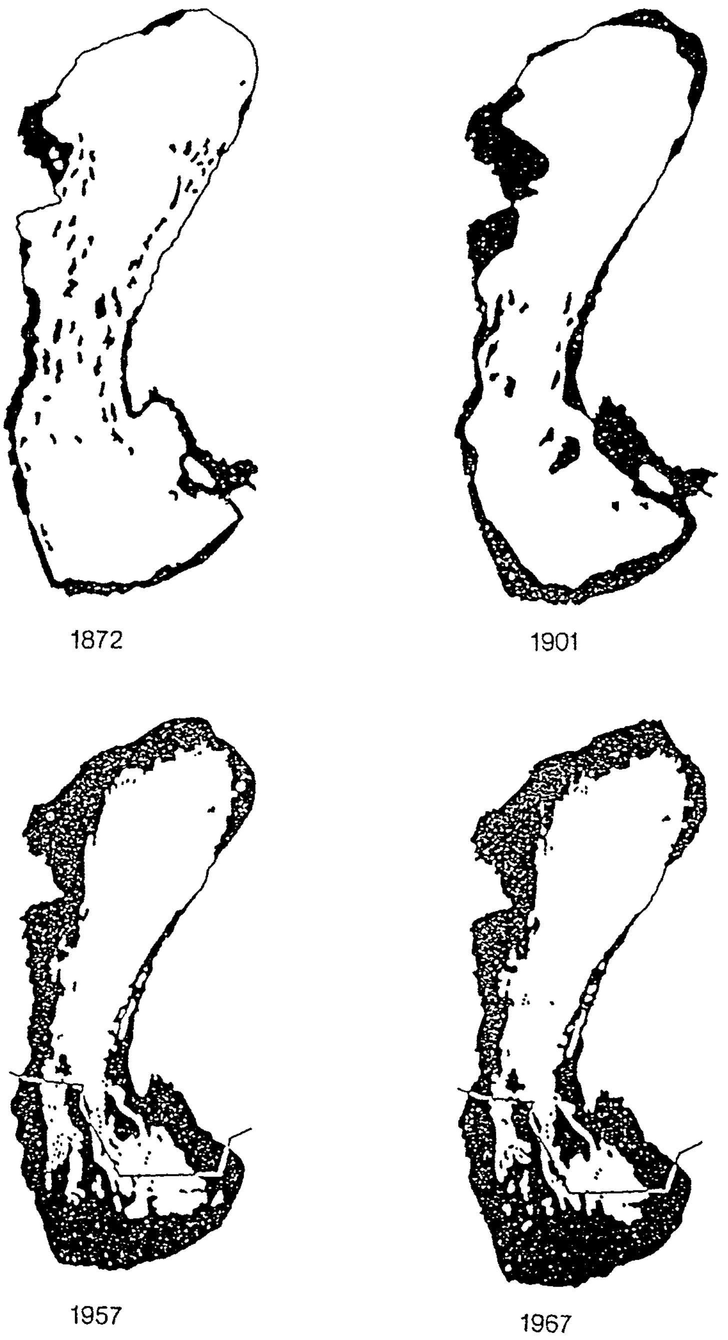

Fig. 9.2. Development of the Neusiedlersee reed belt from 1872 to 1967. (Orig.)

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

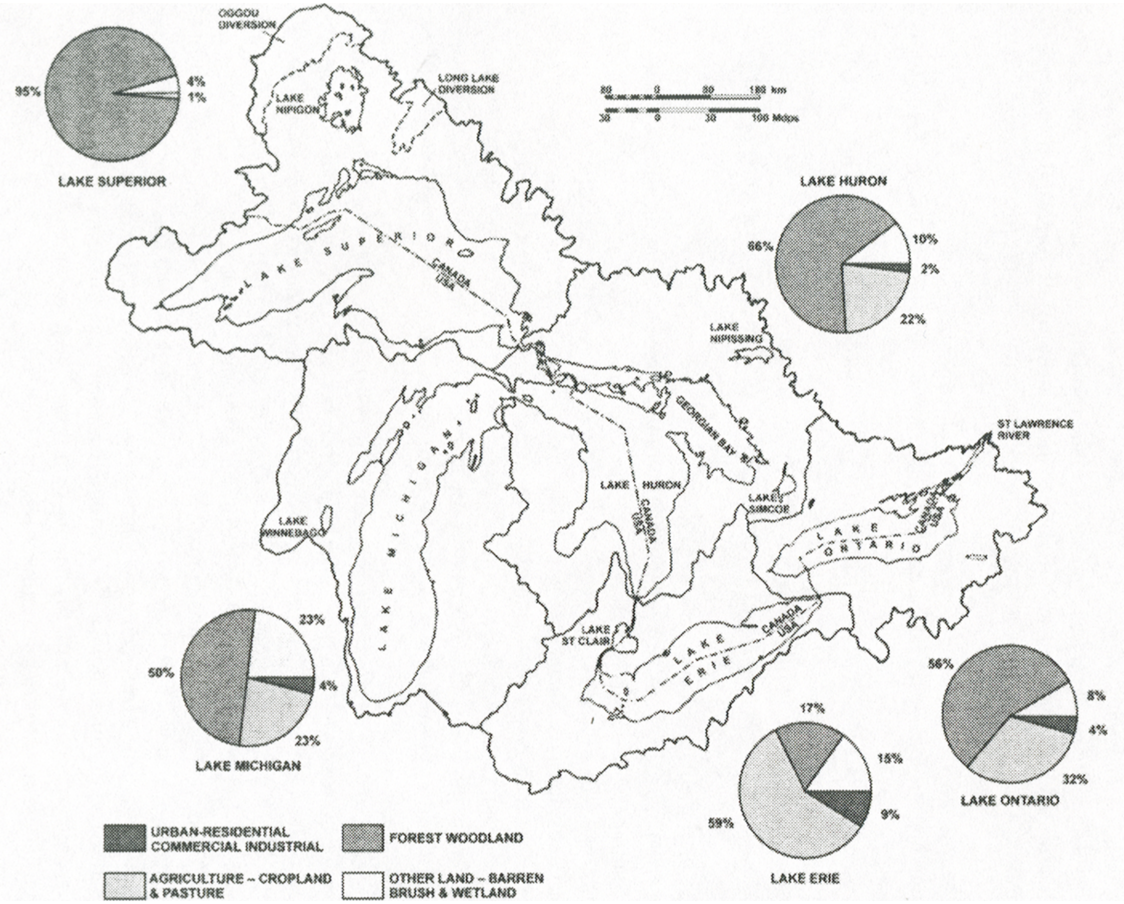

Fig. 9.3. Great Lakes drainage basin and land use percentages for each lake (PLUARG, 1978).

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.

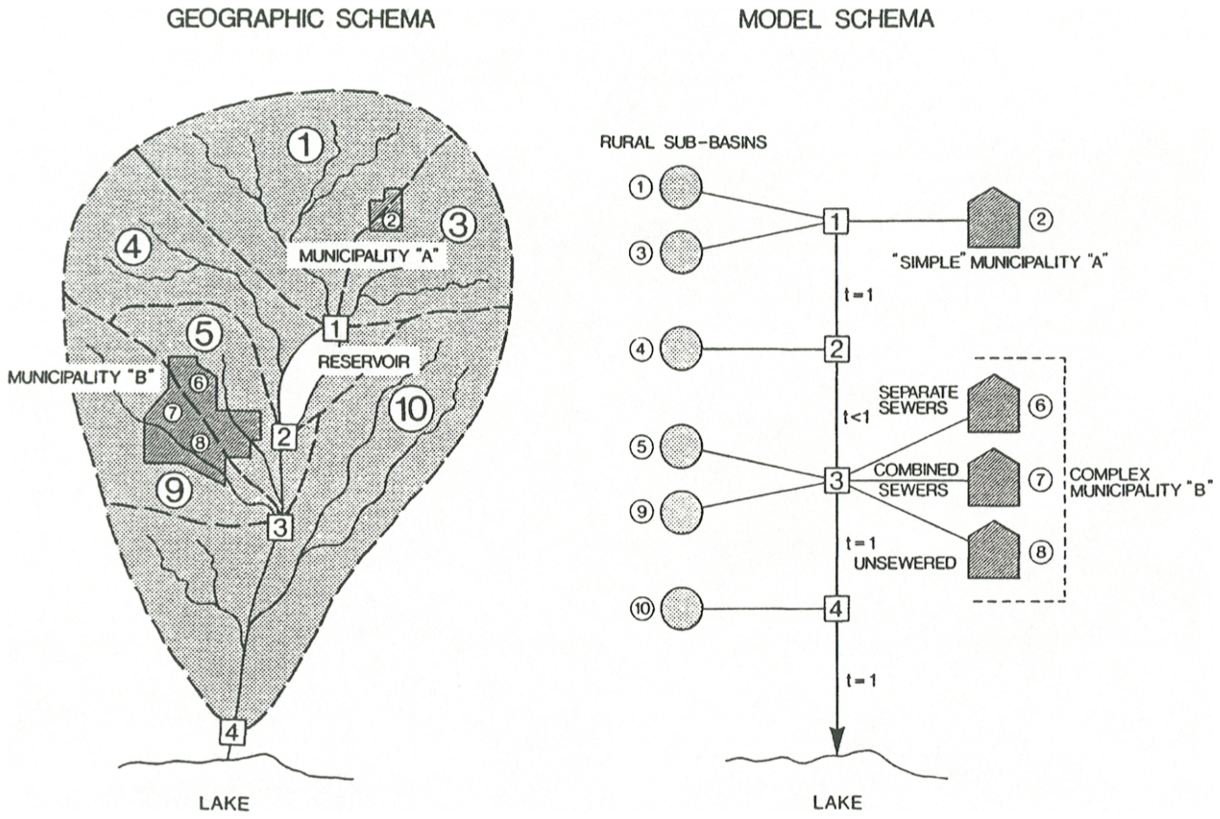

Fig. 9.4. Watershed model for Great Lakes used during the PLUARG study (PLUARG, 1978).

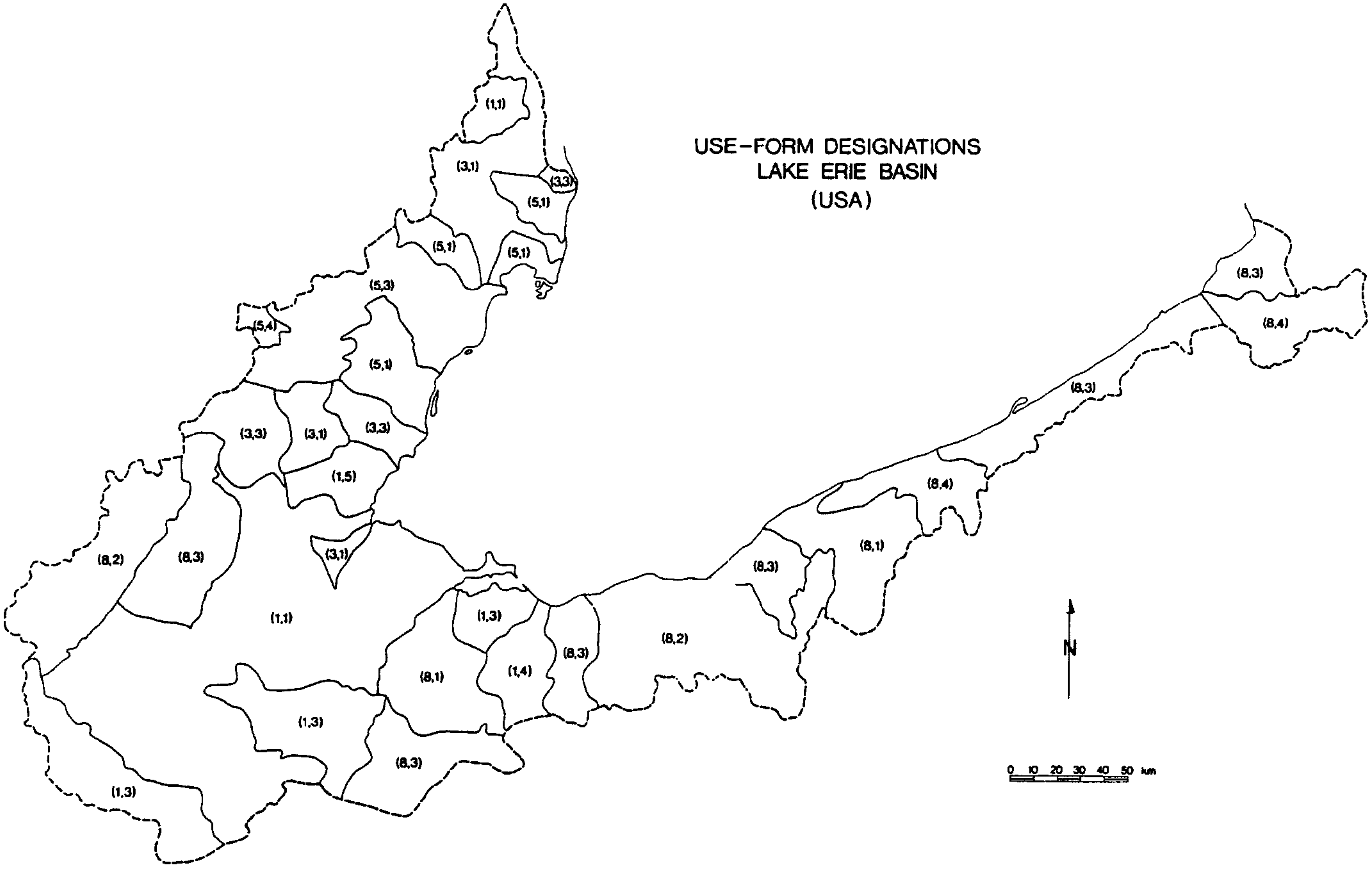

Fig. 9.5. Land use—land form designation for the U.S. portion of the Lake Erie drainage basin (Johnson et al., 1978).

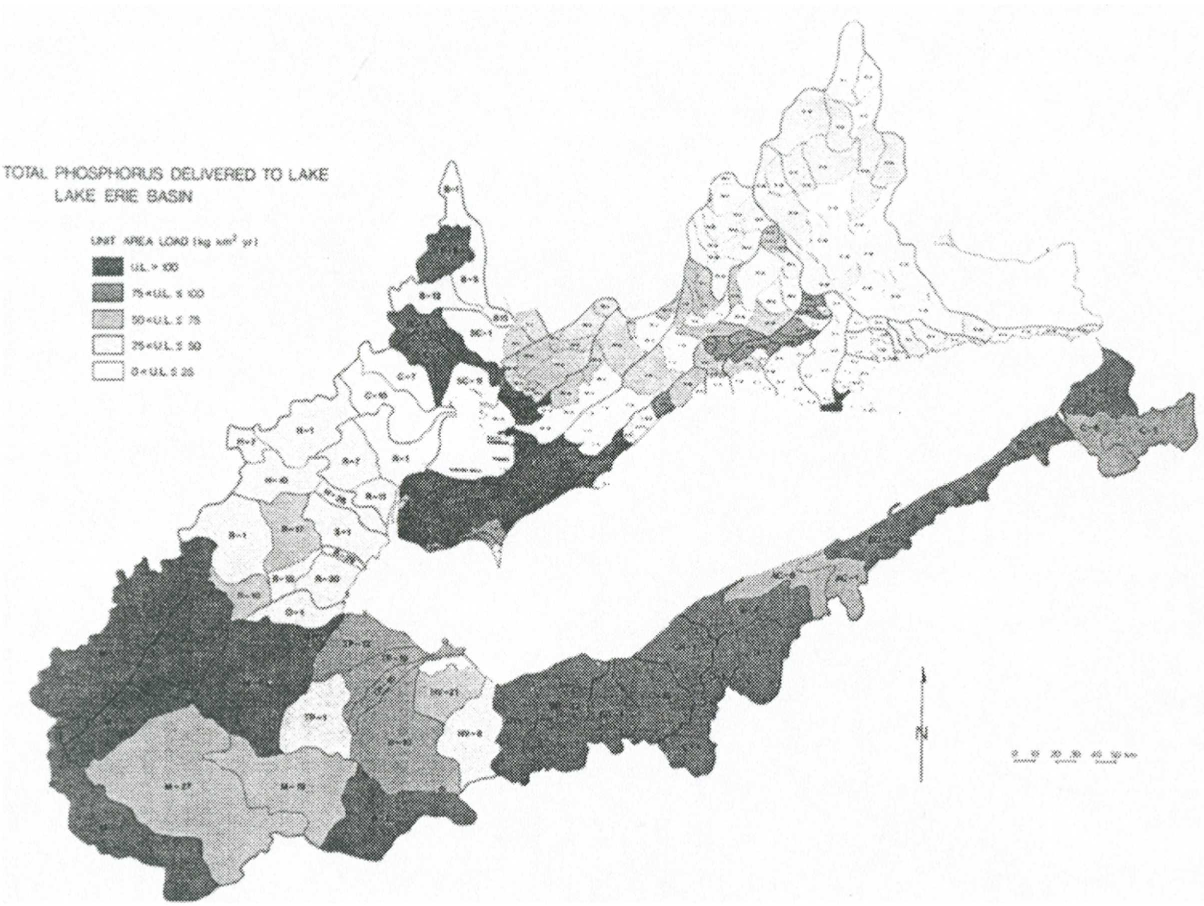

Fig. 9.6. Total phosphorus unit area loads for Lake Erie drainage basin (Johnson et al., 1978).

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

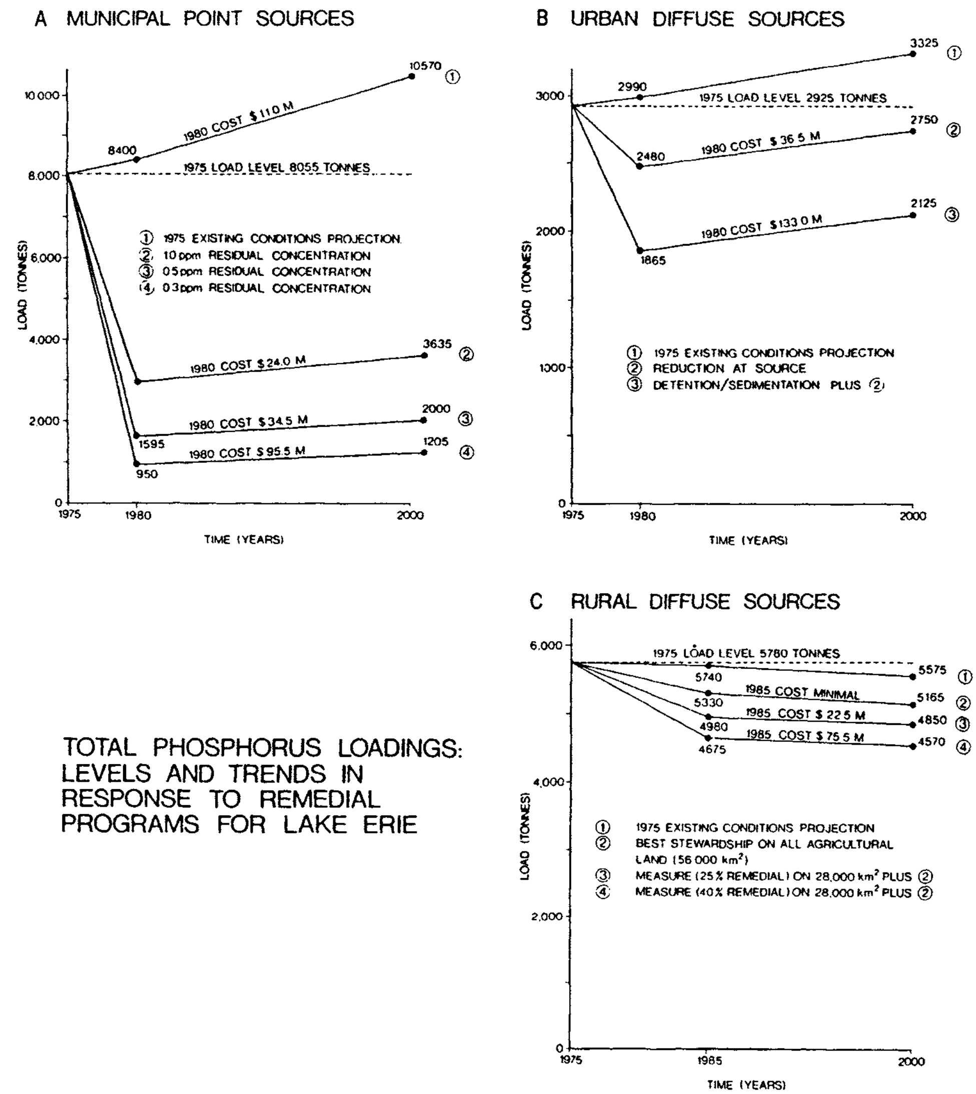

Fig. 9.7. Total phosphorus loading levels and trends in response to remedial programs for Lake Erie (Johnson et al., 1978).

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 $\mu g\ l^{-1}$ 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)
	−32 g l^{-1} (winter 1997)
	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%) (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)

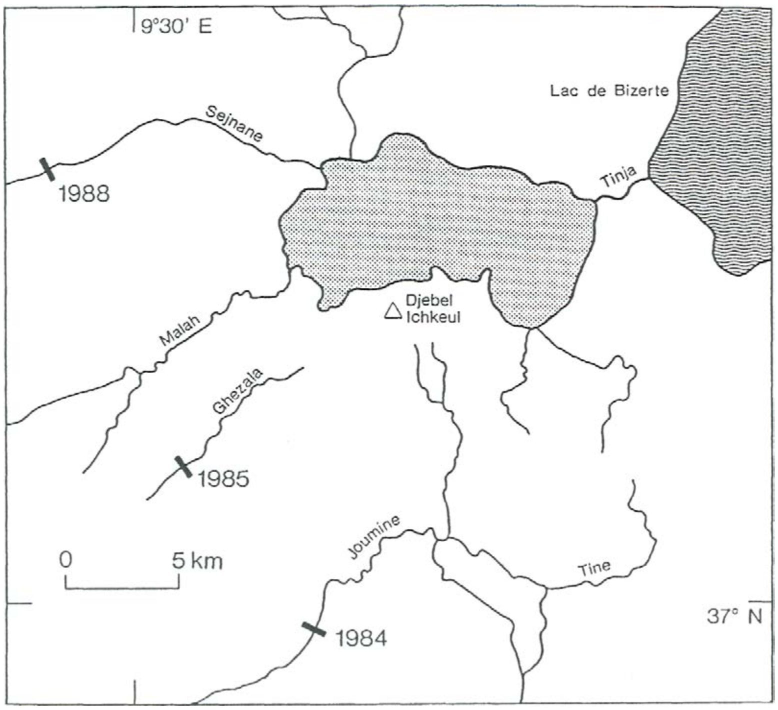

Fig. 9.8. Map of Lake Ichkeul.

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.

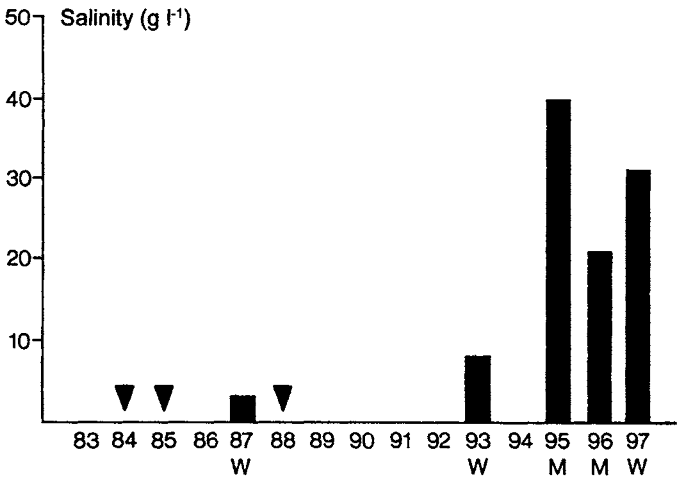

Fig. 9.9. Lake Ichkeul. Arrows indicate years when dams on tributaries were constructed; W—winter, M—March.

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

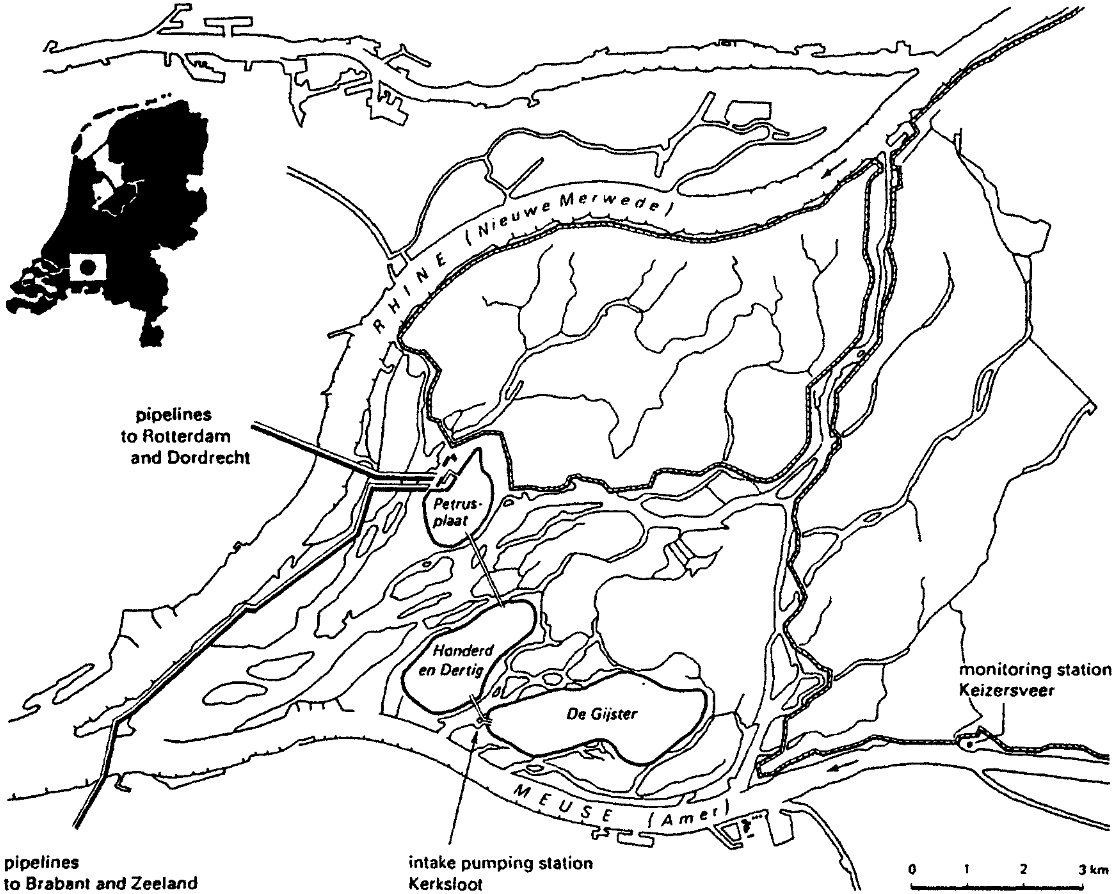

Fig. 9.10. Location of Biesbosch reservoirs in The Netherlands.

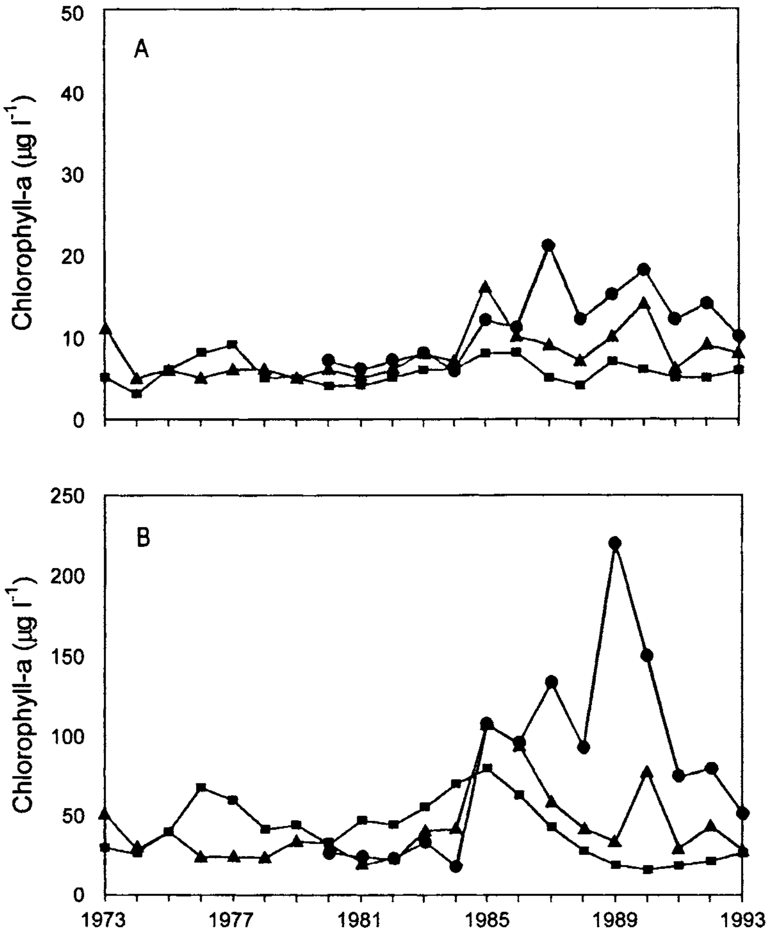

Fig. 9.11. Annual mean (A) and maximum (B) chlorophyll-a concentrations in Biesbosch reservoirs outlets, 1973–1993. ●, De Gijster; ▲, Honderd en Dertig; ■, Petrusplaat (from van Breemen and Ketelaars, 1995).

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

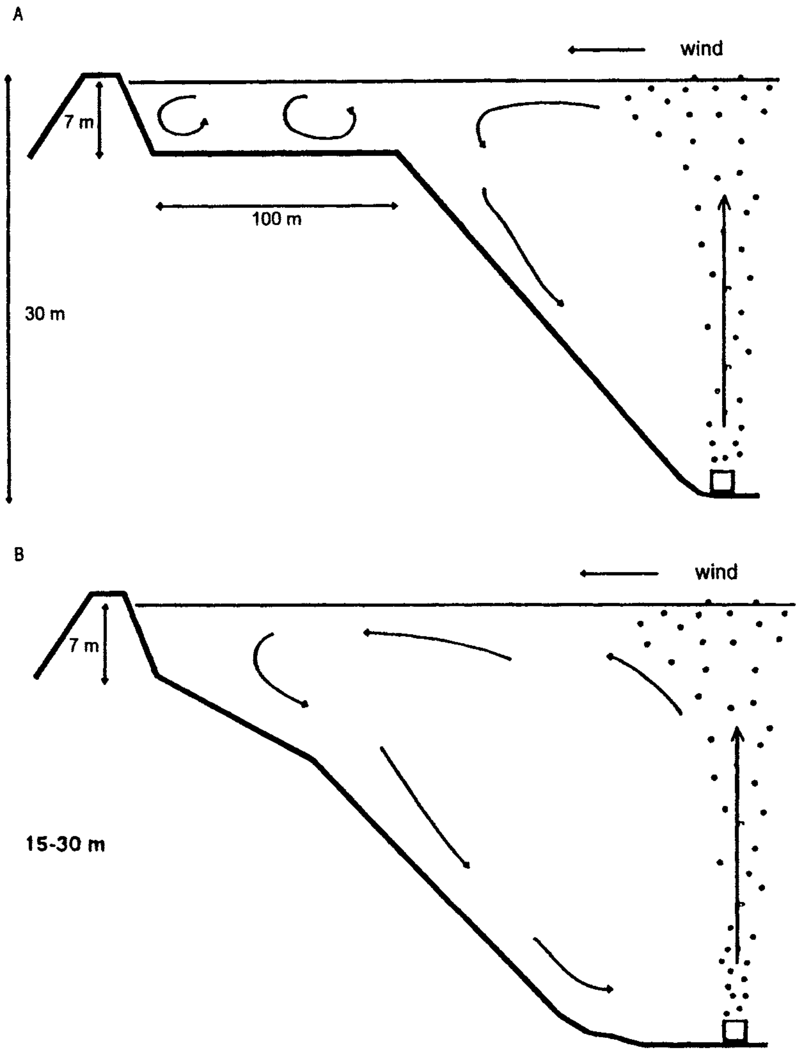

Fig. 9.12. Schematic cross-section of the littoral zone of Biesbosch reservoirs. A—De Gijster Reservoir; B—Honderd en Dertig and Petrusplaat Reservoirs. (Orig.)

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

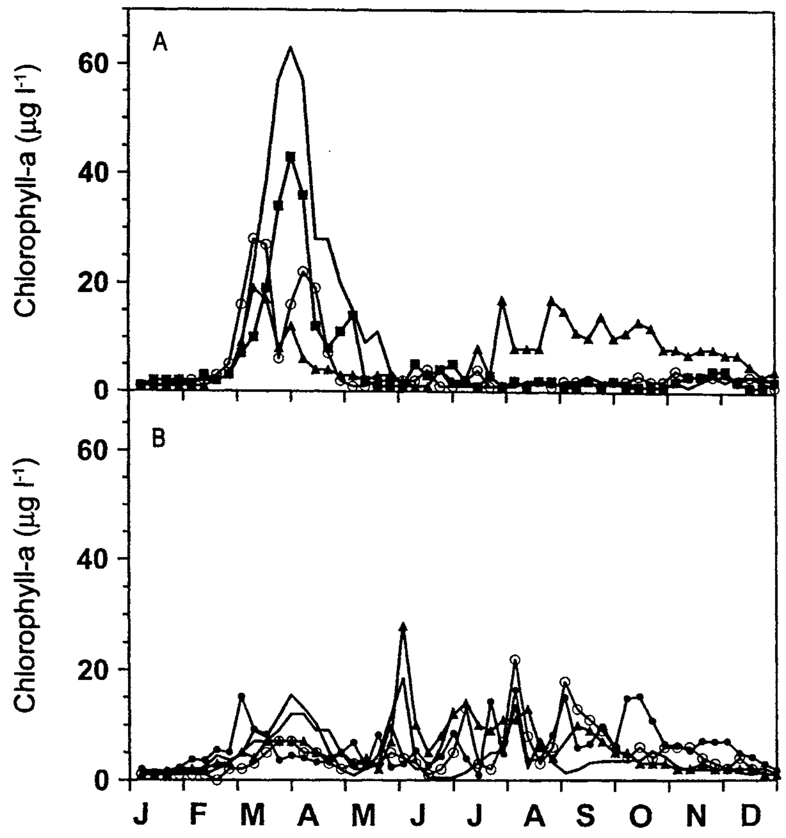

Fig. 9.13. Seasonal variation of chlorophyll-a concentrations in the outlets of Biesbosch reservoirs. A—Petrusplaat Reservoir 1986–1989. B—Petrusplaat Reservoir 1990–1994. 1986; 1987; 1988; 1989; 1990; 1991; 1992; 1993; 1994 (from van Breemen and Ketelaars, 1995).

(*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

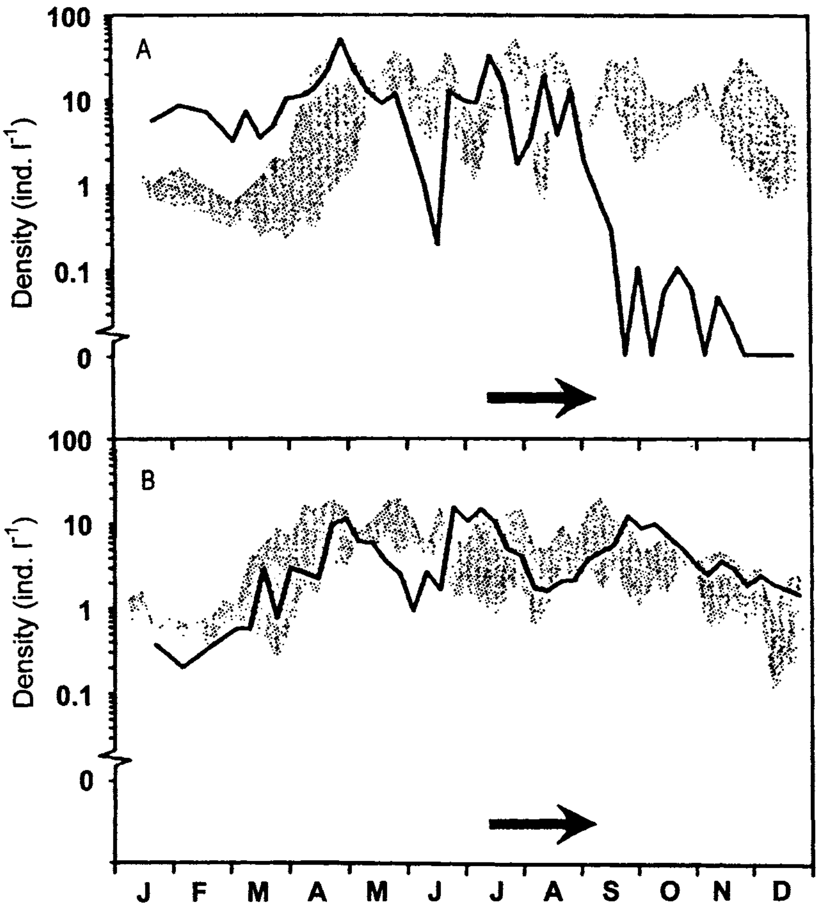

Fig. 9.14. Filtering zooplankton densities in the Honderd en Dertig Reservoir. A—Densities of *Daphnia* spp. and *Bosmina* spp. in 1998 (solid line) compared to those in 1995–1997 (dotted area); B—The same for adult copepoda. Horizontal arrow indicates the start of the period that *Hemimysis anomala* 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

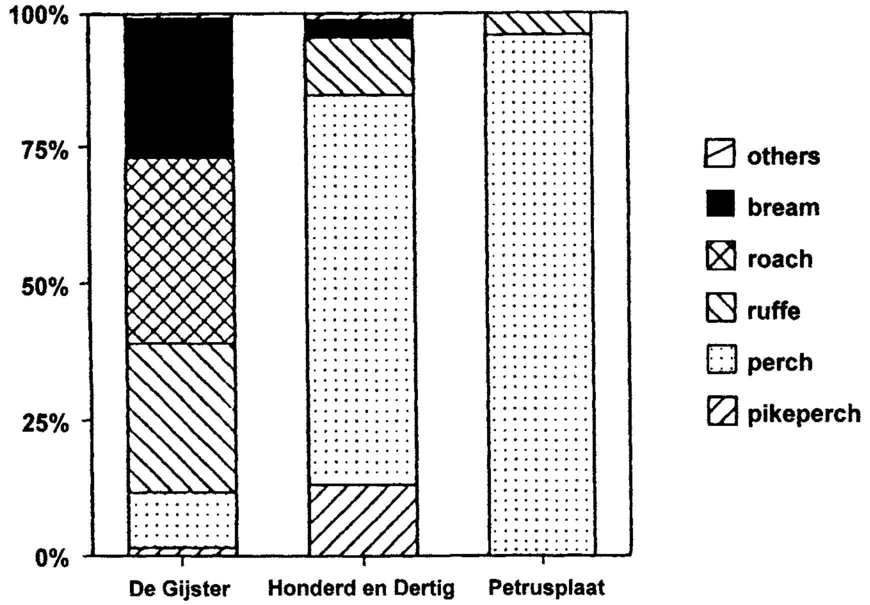

Fig. 9.15. Relative abundance of fish caught in gill-nets in the Biesbosch reservoirs: De Gijster (1991), Honderd en Dertig (1992) and Petrusplaat (1993) (from van Breemen and Ketelaars, 1995).

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 $\mu S\ cm^{-1}$

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

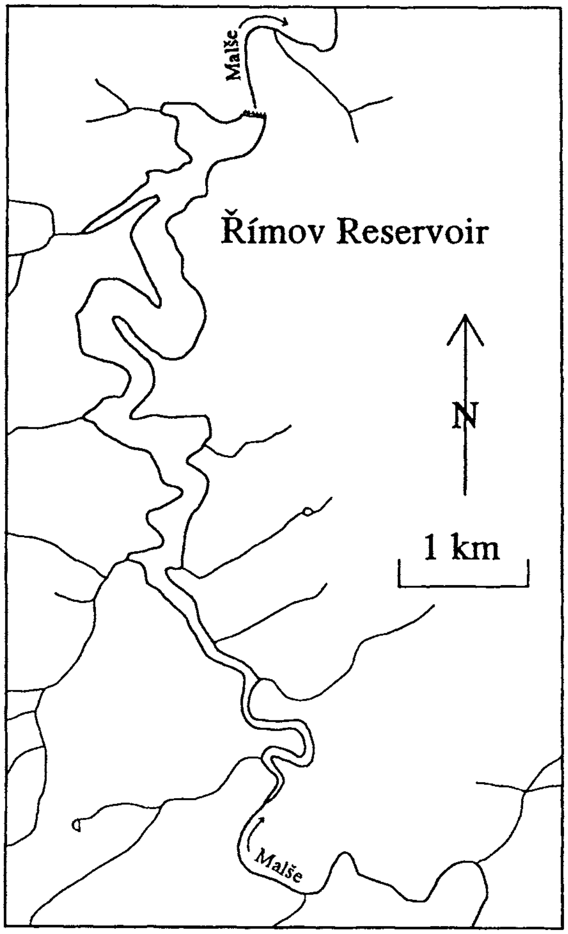

Fig. 9.16. Map of Římov Reservoir showing the riverine character, main inflow—Malše River, and tributary brooks with their bays (from Komárková and Hejzlar, 1996).

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.

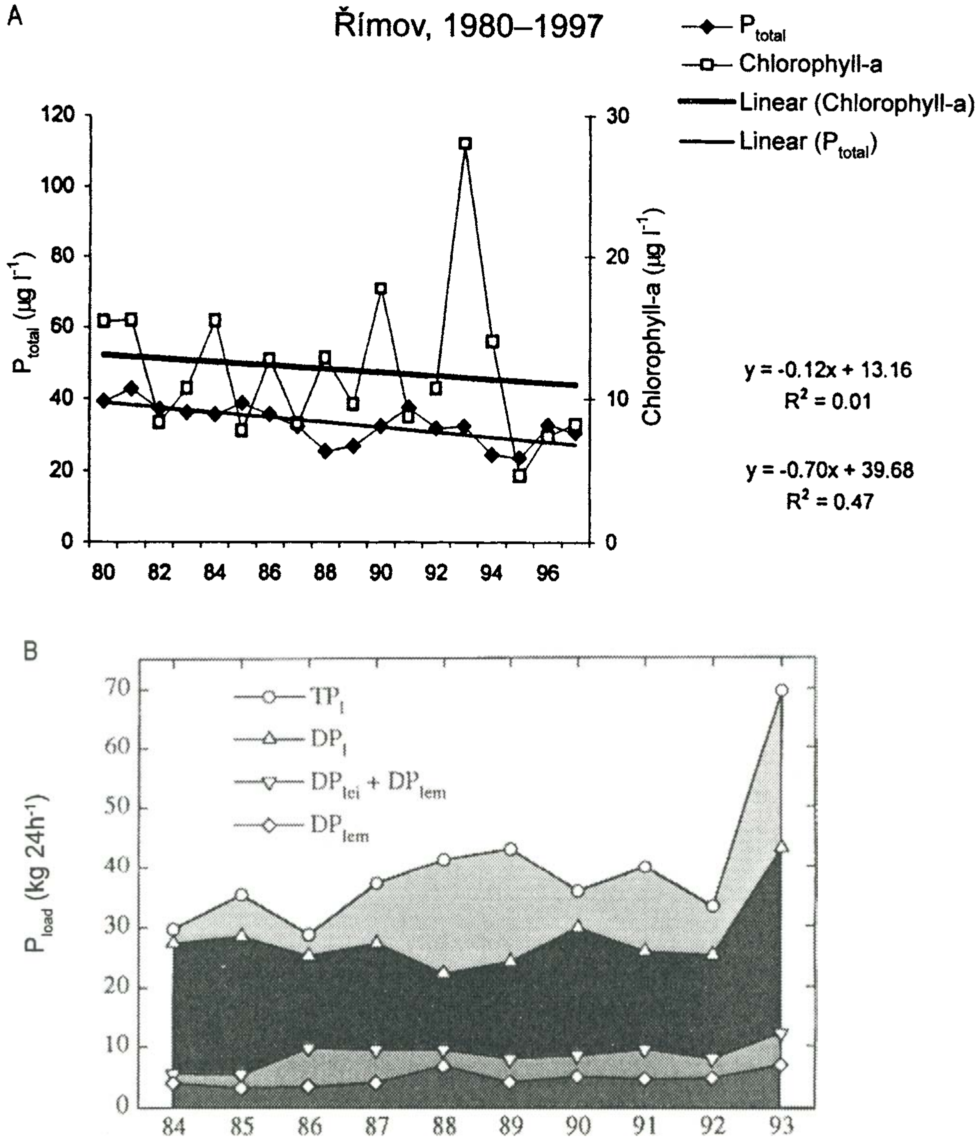

Fig. 9.17. Long-term eutrophication trends in Římov Reservoir. A—Trends of TP and CHA concentrations in the reservoir are decreasing since the reservoir filling, but the amplitude of CHA oscillations among years are increasing (from 38th Annual Report, Hydrobiological Institute, Academy Sciences Czech Republic); B—Phosphorus load of the reservoir, particularly the TP load, shows an increasing tendency (from Komárková and Hejzlar, 1996).

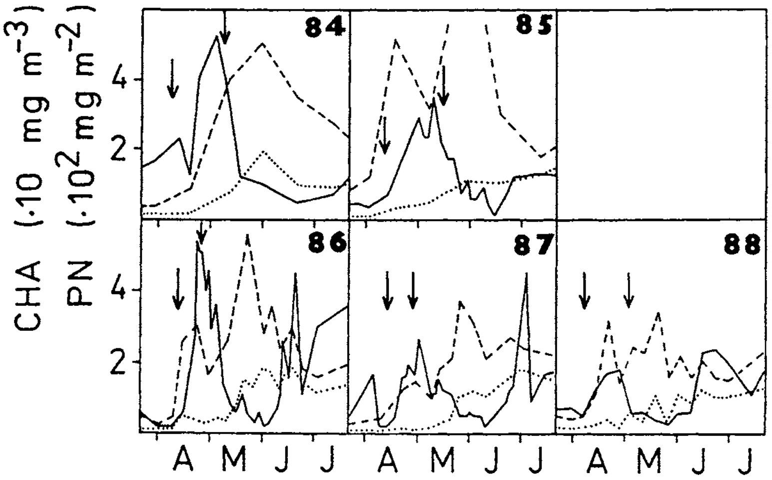

Fig. 9.18. The "clear water phase" in the reservoir, exhibiting low phytoplankton when its control by zooplankton is high, is a regular phenomenon occurring after the onset of summer stratification (from Vyhnálek et al., 1991).

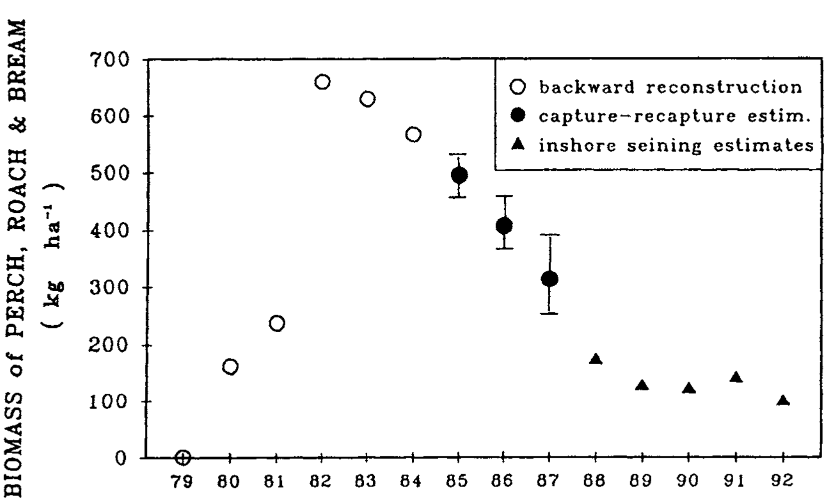

Fig. 9.19. Long-term changes in the annual biomass of three main planktivorous species, perch (*Perca fluviatilis*), roach (*Rutilus rutilus*), and bream (*Blicca bjoerkna*). Only fish older than one year are included (from Sed'a and Kubečka, 1997).

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

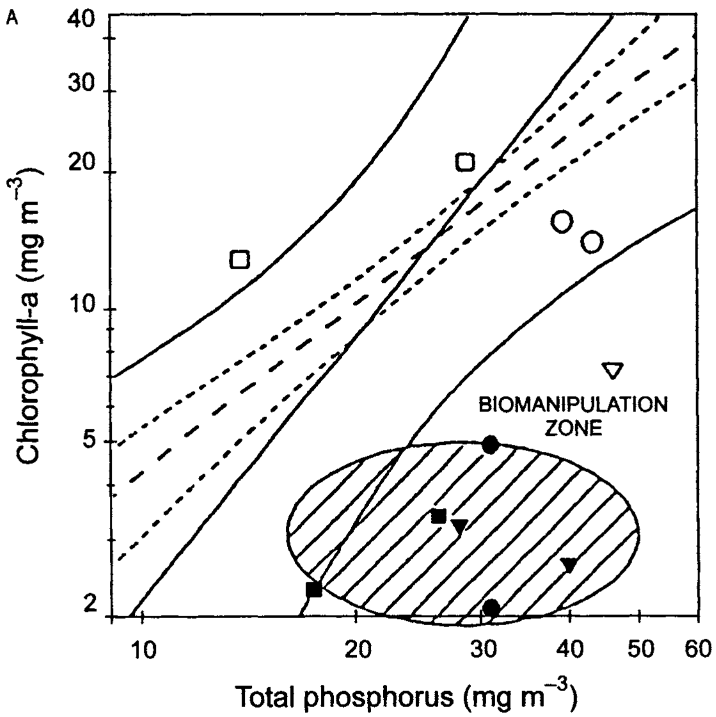

Fig. 9.20. Biomanipulation in lakes and reservoirs. A—Comparison of chlorophyll–phosphorus relationships from unmanipulated reservoirs, and the zone where a significant chlorophyll-a decrease due to biomanipulation was observed in three sets of biomanipulations (Hubenov Reservoir, Round Lake and Haugatjem 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

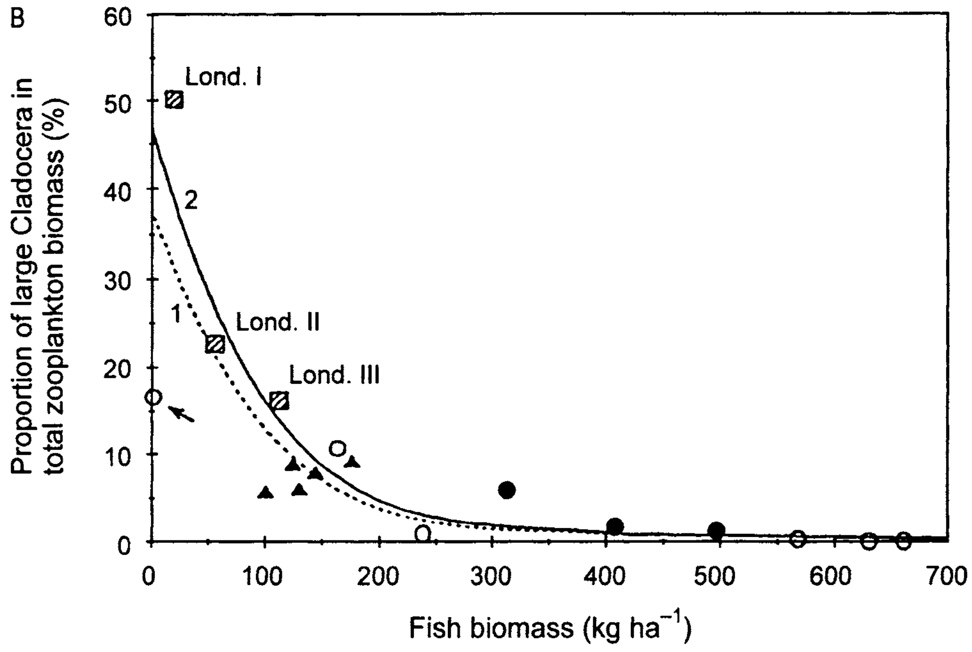

Fig. 9.20 (*continued*). B—Biomanipulation is related to the control of larger zooplankton filtrators by planktivorous fish. Only larger zooplankton appear to be capable of efficiently controlling phytoplankton. The relation is derived from observations in Římov Reservoir and three water supply embankment 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).

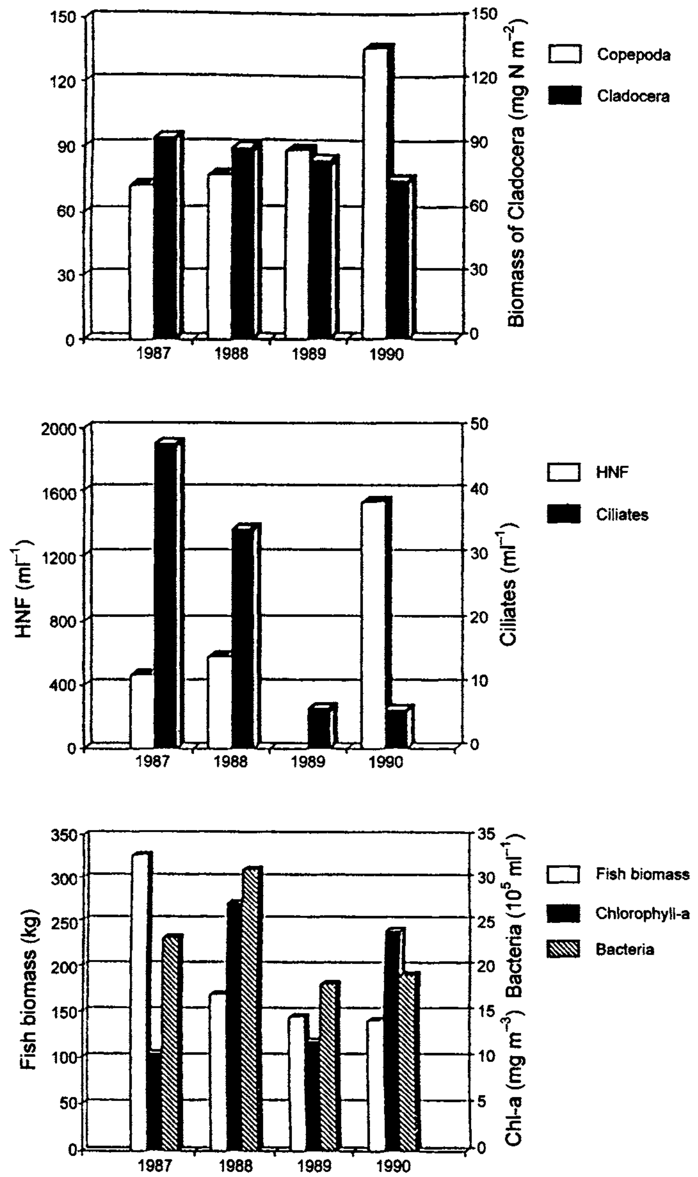

Fig. 9.21. Interannual changes in the annual mean values of microbial and classical food chain in Římov Reservoir. HNF—heterotrophic nannoflagellates. The Copepoda and Cladocera biomass is expressed in nitrogen (from Straškrabová et al., 1994).

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,

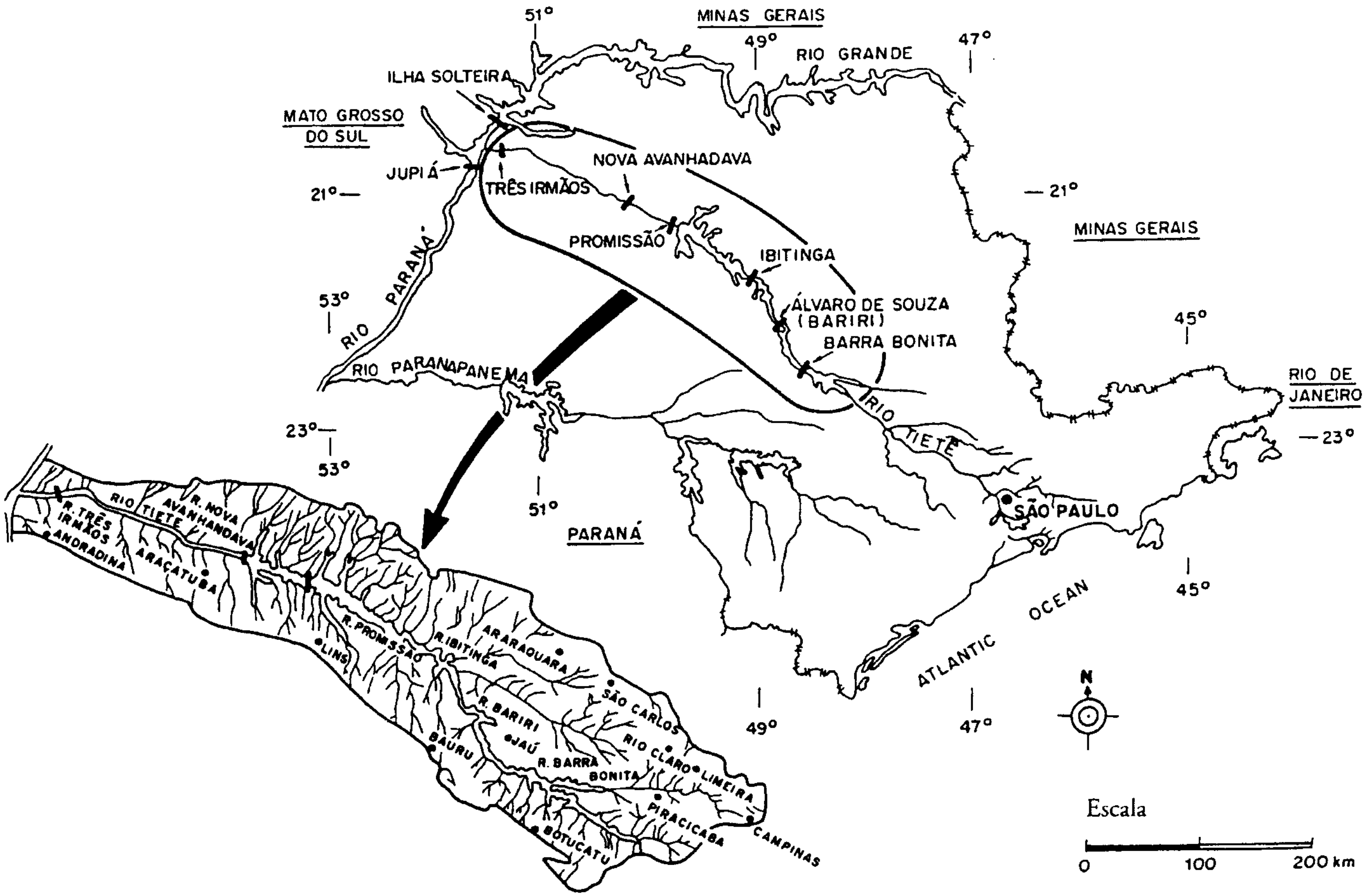

Fig. 9.22. Map of reservoir system on the Tietê River and its location in the São Paulo State, Brazil (from Tundisi et al., 1999a).

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,

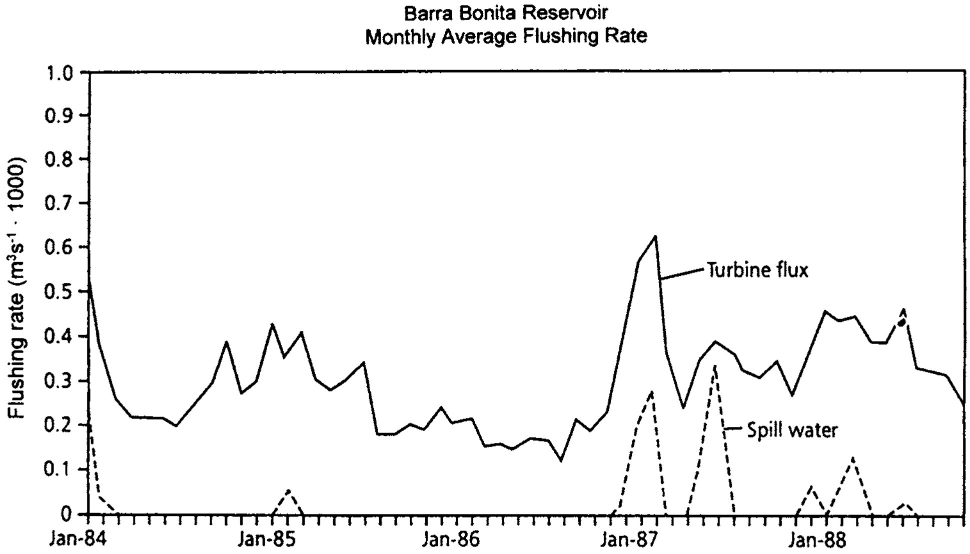

Fig. 9.23. Variation of outflow rates in Barra Bonita Reservoir as an example of the seasonal differences in functioning (from Tundisi et al., 1999b).

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

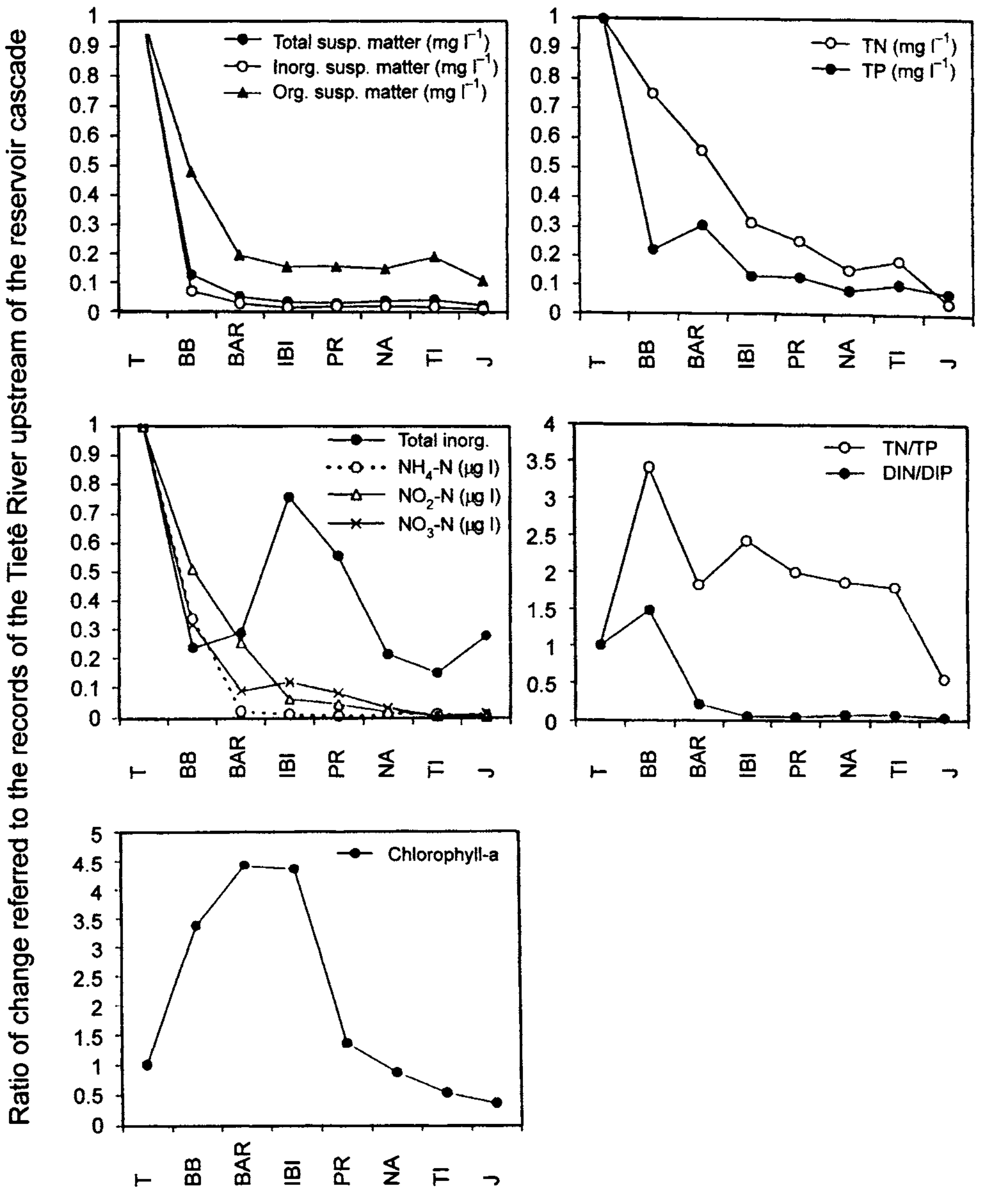

Fig. 9.24. Longitudinal profiles along Tietê River reservoir cascade. Sampling points are in vicinity of each dam. The values are relative to the situation in the river inflow to the cascade (from Barbosa et al., 1999).

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.

REFERENCES

Allan, R.J. and Ball, A.J., 1990. Toxic contaminants in water and sediments of the Great Lakes. Pub. Canadian Assoc. on Water Poll Res. and Control, Monograph No.2, Canada Centre for Inland Waters, Burlington, Ontario, Canada.

Anonymous, 1993. Protection des ressources en eau. Ministere de l'environment (Republique Tunisienne).

Anonymous, 1997. Biodiversite et rarefaction biologique en Tunisie. Ministere de l'environment (Republique Tunisienne).

Anonymous, 1997. Tunisia and its wetlands. Ramsar Newsletter.

Barbosa, F.A.R., Padisák, J., Espíndola, E.L.G., Borics, G. and Rocha, O., 1999. The Cascading Reservoir Continuum Concept (CRCC) and its application to the River Tietê-basin, São Paulo State, Brazil. In: J.G. Tundisi and M. Straškraba (Eds), Theoretical Reservoir Ecology and Its Applications, International Institute of Ecology, Brazilian Academy of Sciences and Backhuys Publishers, Leyden, The Netherlands, pp. 425–439.

Botts, L. and Krushelnicki, B., 1997. The Great Lakes: An Environmental Atlas and Resource Book. Environment Canada, A.S. Environmental Protection Agency, Brock University and Northwestern University, Toronto.

Cotterill, N.G. and Thornton, J.A., 1985. Hydroclimate development following impoundment in a tropical African man-made lake (Lake Robertson, Zimbabwe). J. Limnol. Soc. Sth. Afr., 11(2): 54–61.

Cronberg, G., Andersson, G., Erikson, M., Kautsky, N., Leonardson, L., Lindmark, G., Magadza, C., Machena, C., Skarpe, C. and Stenson, J., 1988. The Ecology of Lake Kariba, Zimbabwe. Report Inst. Ecology. Limnology University Lund, Sweden.

Dick, G., Dvorak, M., Grüll, A., Kohler, B. and Rauer, G., 1994. Vogelparadiese mit Zukunft. Ramsar Bericht, 3. Neusiedlersee-Seewinkel, Umweltbundesamt, Wien.

Dumont, H.J., Laureys, P. and Peusaert, J., 1979. Anostraca, Conchostraca, Cladocera and Copepoda from Tunisia. Hydrobiol., 66: 259–274.

Hejzlar, J., 1989. Dissolved amino sugars in the Římov Reservoir (Czechoslovakia). Arch. Hydrobiol. Beih. Ergebn. Limnol., 33: 291–302.

Hejzlar, J. and Straškraba, M., 1989. On the horizontal distribution of limnological variables in Římov and other stratified Czechoslovak reservoirs. In: B. Henderson-Sellers and V. Straškrabová (Eds), Proceedings of the International Conference of Reservoir Limnology and Water Quality, Vol. 1. Physical Limnology, Mathematical Models and Microbial Decomposition. Archiv für Hydrobiologie, Beiheft. Ergebnisse der Limnologie, 33: 41–55.

Hollis, G.E., Agnew, C.T. and Stevenson, A.C., 1983. Wildfowl dammed. The Geographic Magazine.

IJC (International Joint Commission), 1970. Pollution of Lakes Erie, Ontario and the International Section of the St. Lawrence River. Report of the International Joint Commission to the Governments of the United States and Canada. U.S. Department of State, Washington, DC.

Johnson, M.G., Conneau, J.C., Heidtke, T.M., Sonzogni, W.C. and Stahlbaum, B.W., 1978. Management information base and overview modelling. Technical Report, PLUARG, IJC, Great Lakes Regional Office, Windsor, Ontario, Canada, 90 pp.

Kautsky, N., Berg, H., Folke, C., Larsson, J. and Troell, M., 1997. Ecological footprint for assessment of resource use and development limitations in shrimp and Tilapia aquaculture. Aquaculture Research, 28(10): 753–766.

Ketelaars, H.A.M. and Van Breemen, L.W.C.A., 1993. The invasion of the predatory cladoceran Bythotrephes longimanus Leydig and its influence on the plankton communities in the Biesbosch reservoirs. Verh. Int. Verein. Limnol., 25: 1168–1175.

Ketelaars, H.A.M., Lambregts-van de Clundert, F.E., Carpentier, C.J., Wagenvoort, A.J. and Hoogenboezem, W., 1999. Ecological effects of the mass occurrence of the Ponto–Caspian invader, Hemimysis anomala G.O. Sars, 1907 (Crustacea: Mysidacea), in a freshwater storage reservoir in The Netherlands, with notes on its autecology and new records. Hydrobiol., 394: 233–248.

Komárková, J. and Hejzlar, J., 1996. Summer maxima of phytoplankton in the Římov Reservoir in relation to hydrologic parameters and phosphorus loading. Arch. Hydrobiol., 136(2): 217–236.

Kusel, E. and Löffler, H., 1997. University Student Excursion Report. Unpubl. manuscript.

Lemoalle, J., 1987. Coastal North Africa. In: M.J. Burgis and J.S. Symoens (Eds), African Wetlands and Shallow Water Bodies. ORSTOM, pp. 35–78.

Löffler, H. (Ed.), 1979. Neusiedlersee. Limnology of a Shallow Lake in Central Europe. Monographiae Biologicae, 37. Junk, The Hague, The Netherlands, 543 pp.

Löffler, H., 1981. Neusiedlersee: A puzzle for scientists, a challenge for planners. Nature Resour., 17: 10–14.

Löffler, H., 1990. Paleolimnology of Neusiedlersee, Austria. Hydrobiol., 214: 229–238.

Löffler, H. and Gunatilaka, A., 1994. The shallow lake and reed (Phragmites australis) wetland of Neusiedlersee. In: B.C. Patten (Ed.), Wetlands and Shallow Continental Water Bodies, Vol. 2. Case Studies. Academic Publ., The Hague, The Netherlands, pp. 183–202.

Lund, J.W.G., 1978. Changes in the phytoplankton of an English lake. Hydrobiol. J., 14: 6–21.

Machena, C. and Kautsky, N., 1988. A quantitative diving survey of benthic vegetation and fauna in Lake Kariba, a tropical man-made lake. Freshw. Biol., 19: 1–14.

Magadza, C.H.D., 1988. The Status of Lake Kariba. Expert Group Management on river/lake basin approach to environmentally sound management of water resources. Otsu and Nagoya, Japan. ILEC, UNEP, UNCED.

Marshall, B.E., 1982. The benthic fauna of Lake McIlwaine, and the fish of Lake McIlwaine. In: J.A. Thornton (Ed.), Lake McIlwaine: The Eutrophication and Recovery of a Tropical African Man-Made Lake. Monographiae Biologicae, 49. Junk, The Hague, The Netherlands, pp. 144–188.

Marshall, B.E. and Junor, F.J.R., 1981. The decline of Salvinia molesta on Lake Kariba. Hydrobiol., 83: 477–484.

Marshall, B.E., Junor, F.J.R. and Langerman, J.D., 1982. Fisheries and fish production on the Zimbabwean side of Lake Kariba. Kariba Studies, 10: 175–231.

Marshall Macklin Monaghan Ltd, 1977. Evaluation of remedial measures to control nonpoint sources of water pollution in the Great Lakes Basin. Task A Report, PLUARG, IJC, Great Lakes Regional Office, Windsor, Ontario, Canada.

Munawar, M., Edsall, T. and Leach, J. (Eds), 1995. The Lake Huron ecosystem: Ecology, Fisheries and Management. Backhuyus Publishers, Leiden, The Netherlands.

Munawar, M., Edsall, T. and Munawar, I.F. (Eds), 1999. State of Lake Erie (SOLE). Backhuyus Publishers, Leiden.

Oskam, G., 1978. Light and zooplankton as algae regulating factors in eutrophic Biesbosch reservoirs. Verh. Int. Verein. Limnol., 20: 1612–1618.

Oskam, G., 1982. Quality aspects of the Biesbosch reservoirs. Aqua, 6: 497–504.

Oskam, G. and van Breemen, L.W.C.A., 1992. Management of Biesbosch reservoirs for quality control with special reference to eutrophication. In: D.W. Sutcliffe and J.G. Jones (Eds), Eutrophication: Research and Application to Water Supply. Freshwater Biological Association, Ambleside, pp. 197–213.

PLUARG, 1977. Land use and land use practices in the Great Lakes Basin, Joint Summary Report—Task B (United States and Canada). Task B Report, PLUARG, IJC, Great Lakes Regional Office, Windsor, Ontario, Canada.

PLUARG, 1978. Environmental Management Strategy for the Great Lakes System. Final Report, PLUARG, IJC, Great Lakes Regional Office, Windsor, Ontario, Canada.

PLUARG, 1979. Annotated bibliography of PLUARG reports. PLUARG, IJC, Great Lakes Regional Office, Windsor, Ontario, Canada.

Ramberg, L., Björk-Ramberg, S., Kautsky, N. and Machena, C., 1987. Development and biological status of Lake Kariba—A man-made tropical lake. Ambio, 16: 314–321.

Rast, W. and Gregor, D., 1979. Report on differences in Great Lakes phosphorus load estimates. Task D Report, PLUARG, IJC, Great Lakes Regional Office, Windsor, Ontario, Canada.

Sanyanga, R.A., 1998. Food composition and selectivity of Synodontis zambezensis (Pisces: Mochokidae) in Lake Kariba, and the ecological implications. Hydrobiol., 361: 89–99.

Sed'a, J. and Kubečka, J., 1997. Long-term biomanipulation of Římov Reservoir (Czech Republic). Hydrobiol., 345: 95–108.

Skimin, W.E., Powers, E.C. and Jarecki, E.A., 1978. An Evaluation of Alternatives and Costs for Nonpoint Source Controls in the United States Great Lakes Basin. Task A Report, PLUARG, IJC, Great Lakes Regional Office, Windsor, Ontario, Canada.

Skinner, J., Redfish, M. and Wood, J.B., 1987. Conservation reports 13. In: G.E. Hollis (Ed.). University College, London.

Stipek, H., Kühnelt, W., Mazek-Fialla, K. and Wimmer, C., 1935. In: Heimat und Schule, 3. Deutscher Verlag für Jugend und Volk, Wien und Leipzig.

Straškraba, M., Dostálková, I., Hejzlar, P. and Vyhnálek, V., 1995. The effect of reservoirs on phosphorus concentration. Internationale Revue der gesamten Hydrobiologie, 80(1): 403–413.

Straškrabová, V. and Procházková, L., 1992. Zadržování fosforu v nádrži Římov (Retention of phosphorus in reservoir). In: Pitná voda z údolních nádrží 1992. Sborník 2 celostátní konference, May 25–28, Tábor. Watwer and Environmental Technology Team České Budějovice, pp. 401–405.

Straškrabová, V. and Šimek, K., 1993. Microbial loop in lakes and reservoirs related to trophy and metazooplankton development. Verh. Int. Verein. Limnol., 25: 1183–1186.

Straškrabová, V., Komárková, J., Macek, M., Sed'a, J., Šimek, K., Vrba, J. and Vyhnálek, V., 1994. Microbial–algal–crustacean interactions in a reservoir under different fishstock. Arch. Hydobiol. Beih., 40: 209–221.

Talling, J.F. and Talling, I.B., 1965. The chemical composition of African lake waters. Intern. Rev. Ges. Hydrobiol., 50: 421–463.

Thesiger, W., 1964. The Marsh Arabs. Longmans, Green & Co Ltd.

Tundisi, J.G., Matsumura-Tundisi, T. and Rocha, O., 1999a. Ecossistemas de áquas interiores. In: C.A. da Rebouças, B. Braga and J.G. Tundisi (Eds), Áquas Doces no Brasil. São Paulo, Escrituras Editora, pp. 153–192.

Tundisi, J.G., Matsumura-Tundisi, T. and Rocha, O., 1999b. Theoretical basis for reservoir management. In: J.G. Tundisi and M. Straškraba (Eds), Theoretical Reservoir Ecology and Its Applications. International Institute of Ecology, Brazilian Academy of Sciences.

U.S. Department of State, 1978. Great Lakes Water Quality Agreement of 1978. Agreement, with Annexes and Terms of Reference, between the United States and Canada, signed at Ottawa, April 15, 1972. U.S. Department of State, Washington, DC.

Van Breemen, L.W.C.A., Ketelaars, H.A.M., Visser, P. and Ebbeng, J.H., 1991. A new method to control growth of geosmin-producing benthic cyanobacteria. Verh. Int. Verein. Limnol., 24: 2168–2173.

Van Breemen, L.W.C.A. and Ketelaars, H.A.M., 1995. The influence of artificial mixing and other factors on algal biomass in the Biesbosch reservoirs. Aqua, 44(1): 65–71.

Visser, P.M., Ketelaars, H.A.M., Van Breemen, L.W.C.A. and Mur, L.R., 1996. Diurnal changes of Microcystis in an artificially mixed storage reservoir. Hydrobiol., 33: 131–141.

Vollenweider, R.A., 1976. Advances in defining critical loading levels for phosphorus in lake eutrophication. Mem. Ist. Ital. Idrobiol., 33: 53–83.

Vyhnálek, V., Komárková, J., Sed'a, J., Brandl, Z., Šimek, K. and Johanisová, N., 1991. Clear-water phase in the Římov Reservoir (South Bohemia): Controlling factors. Verh. Int. Verein. Limnol., 24: 1336–1339.

Winkler, A.E., 1923. Die Zisterzienser am Neusiedlersee und Geschichte dieses Sees. Missionsdruckerei St. Gabriel, 248 pp.

Zaouali, J., 1995. Limnology in Tunisia. In: R.G. Wetzel and B. Gopal (Eds), Limnology in Developing Countries, Vol. 1. Int. Ass. Limnol., pp. 41–61.