Restoration of Lake Ecosystems

a holistic approach

A training handbook edited by Martina Eiseltová

With contributions from
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With financial support from

World Wide Fund For Nature
UK Government's Know How Fund
Joint Nature Conservation Committee, United Kingdom

IWRB Publication 32 • 1994

Published by the International Waterfowl and Wetlands Research Bureau,
Slimbridge, Gloucester, GL2 7BX, UK

ISBN 0 9505731 5 9

This publication should be cited as follows: Eiseltová, M. (ed.), 1994. Restoration of lake ecosystems – a holistic approach. IWRB Publ. 32. 182 pp.

Cover photograph: Vajgar fish pond in Jindřichův Hradec, Czech Republic – before and after restoration. Courtesy of Jan Pokorný.

Designed and produced by the Nature Conservation Bureau Limited, 36 Kingfisher Court, Hambridge Road, Newbury, Berkshire, RG14 5SJ.

Printed by Information Press, Oxford.

Dedicated to the memory of Jiří Janda
who devoted his life and work to the conservation
of his beloved Czech countryside

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Preface

Despite the increasing appreciation of wetland values and functions, and the acknowledgement of their benefits for humankind, wetlands continue to be destroyed and degraded at an alarming rate. At the same time as this process of wetland degradation continues, the effort to promote nature conservation, including conservation of wetlands, is rapidly escalating. There is more money being spent and more effort directed towards the conservation and wise use of wetlands than ever before. Increasing numbers of scientists are becoming involved in wetland ecology research, and individual and organisational support for conservation, restoration and sustainable use of wetlands is growing. Wetland conservation has become an important issue in political debates which, in several countries, has resulted in the development of national wetland policies. The increasing international support for wetland conservation is reflected by the number of countries (81 by July 1994) which have become Contracting Parties to the Ramsar Convention, an international treaty which obliges the Contracting Parties to protect and make wise use of wetlands in their territories.

In many cases, however, wetland degradation is caused by non-sustainable, destructive management of landscapes as a whole. Amongst the most serious types of damage is the deterioration of the water cycle due to excessive drainage and destruction of natural vegetation cover, which is coupled with irreversible matter losses from catchment areas to the sea. It is now known that only by considering all the processes in a catchment area can the function of wetlands be understood and their proper functioning secured. This requires that a new – holistic – approach to landscape management be adopted.

Radical developments in the wetland ecology science and in approaches to wetland management over the last two decades have provided an impetus for the promotion of training and re-training of personnel involved in wetland research, management and conservation. Among other international organisations – including the World Conservation Union (IUCN) and World Wide Fund For Nature (WWF) – the International Waterfowl and Wetlands Research Bureau (IWRB) has also become involved in the field of training. In October 1991, IWRB with support from WWF-UK embarked upon a coordinated wetland management training programme, focused primarily on Central and Eastern Europe. The main objective of the programme is to develop and implement a strategy for training which will support the conservation and sustainable use of wetlands in this part of Europe.

As many wetlands in Central and Eastern Europe have suffered serious deterioration, wetland restoration was identified as one of the key areas where training would be beneficial. The restoration of eutrophic lakes was the theme of two training courses, held in former Czechoslovakia (1992) and Estonia (1993), resulting in the development of this handbook. It is aimed at a fairly broad, advanced audience of ecologists, engineers and planners, who are involved in planning, designing, and implementing restoration projects. Close collaboration between these sectors in the design and implementation of restoration activities is a prerequisite for the long term success of such projects. It is also aimed at higher educational and vocational training institutions, as a resource for curriculum development.

One has to keep in mind that wetland restoration is not a simple task, and requires a sound scientific understanding of general wetland ecology, hydrology, water chemistry as well as relevant knowledge of zoology and botany. Through this handbook, we intend to provide a comprehensive insight into wetland restoration but do not want to repeat basic ecological information dealt with in every textbook. The necessary scientific background is presented in a new way which should contribute to the understanding of how biological processes, the water cycle and energy are interrelated.

The handbook is primarily developed to address the restoration of wetlands in Central and Eastern Europe and, therefore, encompasses mostly case studies from the temperate zone. Nevertheless, the principle of a holistic approach to wetland restoration and integrated landscape management is applicable to other regions, too. Indeed, the non-sustainable management of water resources leads to the degradation of catchments in many parts of the world which, in more southern countries, can escalate even more rapidly due to the higher solar energy input.

Although the handbook is focused on the restoration/redevelopment of lake ecosystems, it is emphasised that these systems function in a larger ecological entity – the landscape. To be truly sustainable, the management of wetlands has to be incorporated into a system of integrated land and water use and, indeed, into the socio-economic system of each country. Thus, we suggest some possible solutions for ecologically sound landscape development considering both environmental and socio-economic interests of society.

IWRB intends this publication to be used widely in the development of wetland restoration strategies, in the design and implementation of restoration projects, and as a resource for training professionals. It is necessary that we adopt the concept of integrated landscape conservation and effectively protect one of the most important resources of life – water. Water of good quality and sufficient quantity must be retained in the landscape. This can be achieved only by preserving, and where necessary re-creating, areas of properly functioning natural water bodies and wetlands which are vital for our planet and its biosphere. We hope this publication contributes to the theoretical knowledge and practice of a holistic approach to restoration ecology.

Martina Eiseltová
July 1994

Acknowledgements

The authors of the handbook must be the first to be acknowledged. Sven Björk, Willy Ripl, Jan Pokorný and Bo Verner were around the table at Třeboň, where the first discussions over the preparation of the handbook started. A few months later, in March 1993, this team, accompanied by Klaus-Dieter Wolter, met again to finalise the contents. Sven Björk had arranged a nice, quiet place at Tranemåla where three days passed in lively discussions over the details of contributions by individual authors – with the aim of producing a comprehensive handbook. Many years of their experience in designing, planning and conducting restoration activities, as well as their extensive teaching experience, helped tremendously in identifying a useful level of theoretical information and illustrative case studies to be included. Jaroslav Hrbáček, Josef Matěna, Vojtěch Vyhnálek and Karel Šimek were then invited to contribute with their many years of experience in limnological research, especially with regard to food webs. Help on the preparation of a few of the manuscripts was also received from Steve Ridgill and Václav Hauser. Anna-Lisa Björk accompanied many chapters of the text by illustrative, scientifically precise, drawings for which she receives my sincere thanks. Close collaboration with all these people was a great pleasure and a considerable support in the effort to prepare a valuable publication. All authors kindly agreed to include their addresses (Appendix B) and will be happy to respond to any readers' enquiries if time and knowledge allow them.

Throughout editing the manuscripts, I was consulting with Steve Ridgill over the many peculiarities of English grammar. He deserves my grateful thanks for his generous editorial help. Thanks are also due to my colleagues at IWRB, Mike Moser and Crawford Prentice, who assisted with the improvement of a few chapters, Simon Nash for dealing with the designers, and Sharon Favell who continued patiently to type and retype the manuscripts.

The handbook was made possible thanks to WWF-UK and the National Westminster Bank, UK, who have been supporting the IWRB wetland management training programme for three years. Contact was through Gaynor Whyles and Chris Tydeman who provided valuable advice throughout. Two training courses on the theme of lake restoration were convened prior to the preparation of this publication. My thanks must go to the Institute of Botany at Třeboň, Academy of Sciences of the Czech Republic for hosting the training course in October 1992, and the Limnological Station at lake Võrtsjärv, Institute of Zoology and Botany of the Estonian Academy of Sciences for hosting the training course in May 1993. I would also like to thank many people who contributed to the smooth organisation of these training courses, in particular to Jan Pokorný, Jana Šuláková, Jana Knoppová, Ingmar Ott and Arvo Tuvikene. Financial support was also received from the Czech Ministry for the Environment and the UK Government's Know How Fund. The Joint Nature Conservation Committee, UK, have provided generous financial support allowing extra copies of this handbook to be printed.

Final thanks must go to three colleagues and friends who had a great influence in my career. Max Finlayson introduced me to the job at IWRB and provided valuable advice and encouragement for developing the training programme in its beginning. For my education in wetland ecology I am especially grateful to Jan Květ and Jan Pokorný, and to my parents who made my education possible.

Martina Eiseltová
July 1994

1. Overview

Sven Björk

Man-induced degradation of inland waters

The drainage of lakes and wetlands, and the pollution of inland and coastal water bodies have had detrimental effects in various forms: large-scale losses of surface waters and their biota, rapid terrestrialisation of accumulation basins, regional lowering of the ground water level, and degradation of water quality. Many years of observations have led to the conclusion that severe pollution sometimes results in more or less irreversibly poor conditions, i.e. conditions that remain even after the elimination of sewage and wastewater discharge. Similarly, studies on shallow lakes in which the water level had been lowered have shown that it is often impossible to revert such lakes to their previous character simply by raising the water level. The overall reason for this course of events is that the exploitation of the individual water body was focused on just one goal, as when wetlands were utilised as wastewater receivers or drained for obtaining arable land. Most often the goal was simply short-term economic profit. Because ecological prognoses were lacking, the negative effects successively appearing in both the water body and its catchment area were not considered.

The shoreline of a lake is not a demarcation line between terrestrial and aquatic ecosystems. On the contrary, the littoral is a transitional zone for the transport of water, nutrients and other substances from land to water. The characteristics of individual inland water bodies reflect the conditions in their catchment areas as the hydrological cycle closely connects terrestrial and aquatic ecosystems, with water as the carrying medium. Therefore, water management has to be based on a holistic view in both space – taking into account the interdependence between the lakes and the surrounding land as well as possible effects appearing downstream – and time – not least because there is often a considerable time delay between an influence and the demonstration of its full-scale harmful effects in nature. In the beginning, the changes are often difficult to observe and, therefore, not easy to demonstrate in the short-time perspective of a human generation. Many man-induced changes in aquatic ecosystems, such as overgrowing and terrestrialisation, are not linear processes but start slowly and proceed towards a rapid escalation.

Approach towards wetland redevelopment – the need for guidelines

With the lakes nearest to towns and cities destroyed by pollution, with the agricultural landscape devoid of wetlands, with running waters straightened or hidden in pipes and the forests drained, the negative boomerang effects successively become more obvious, creating a public desire to restore what has been lost. In the 1960s, the first well-documented restoration projects started (e.g. Lake Trummen, see Case study, Chapter 8) in order to demonstrate the possibilities of ecosystem redevelopment. The growing interest in lake and wetland redevelopment/restoration techniques, projects and results, is reflected in the number of congresses and symposia on the topic as, for example, in Blacksburg 1975 (Cairns *et al.* 1977), Rome 1985 (Vismara *et al.* 1985), Budapest 1987 (AMBIO 1988) and Hohenheim 1994 (Böcker & Kohler 1994). The subject has been discussed in new textbooks on limnology and water protection (Besch *et al.* 1984) and is part of modern planning for water resources (*cf.* Björk 1988, the Stockholm planning model). Efforts to incorporate basic ecological thinking in order to secure sustainable environmental conditions – replacing the primitive concept of continuous economic growth without profound ecological consideration – were made by The World Commission on Environment and Development (1987).

At the same time as the degradation of inland water ecosystems is getting worse in some parts of the world, an increasing amount of experience on restoration techniques and results from re-created ecosystems is being collected elsewhere. The desire to bridge this gap between the growing needs for ecosystem redevelopment on the one hand and available expertise on the other, led to the inception of this handbook. The aim is to disseminate a holistic view in space and time on lake ecosystems, their development, and their changeable structure and function, and to illustrate how to govern systems in order to reach defined goals. The explanation of the long-term development of lakes, their transformation into wetlands and their final terrestrialisation, seems to be an appropriate way to inoculate the basic understanding of, and respect for, the so often forgotten time-factor when dealing with dynamic and changeable ecosystems. The primary aim of restoration projects is to create sustainable systems – as measured on a human time-scale. In some cases, continuous management may be necessary in spite of being costly. Cosmetic measures, however, may have an ephemeral revitalising effect but in the course of time only result in aggravated problems.

Redevelopment and restoration

In limnology, which is the science of inland water ecosystems, research, aiming to actively adjust and govern the structure and function of degraded lake and wetland ecosystems started in the 1960s. From the very beginning, the term *restoration* was used only for activities carried out in the ecosystem itself. According to this definition, measures taken to reduce or divert sewage supply do not belong to the act of lake restoration *sensu stricto* but to preparatory stages of making the restoration of damaged ecosystems efficient. (*Compare the fact that marble and limestone sculptures, from European classical antiquity and the Renaissance, damaged by recent atmospheric pollution will not regain their features just through cleaning up the air!*) Looking only at nutrients, preparatory activities like sewage diversion, the building of sewage treatment plants with phosphorus removal, and efforts to avoid diffuse pollution, are necessary steps taken to normalise or reduce external loading. Restoration, on the other hand, means, in this connection, the normalisation or considerable reduction of internal nutrient loading.

In order to correct damage in water bodies where the water level has been lowered, direct operations are also necessary before any raising of the water level. Precipitation of phosphate in hypertrophic lakes, hypolimnetic (deep-water) aeration, artificial circulation through bubbling and lasting regulation of pH-conditions can also be designated as true restoration activities. Removal and neutralisation of toxic substances in contaminated lakes belong to this category too.

Redevelopment is the superior term for both restoration *sensu stricto* and all preparatory work preceding restoration projects. Redevelopment comprises all activities aimed at an upgrading of the environment according to given goals, for the use and utilisation of ecosystems. Improvements appear immediately or gradually.

It should be stressed that restoration does not imply the permanent reinstatement of a water body to the conditions prevailing during a specific phase in its development. In practical environmental protection and management, the meaning of the term restoration is the re-creating of the conditions in such a way that acceptable environmental conditions are re-established (Björk 1968). As a rule, the restorative measures imply the re-establishment of a lake or wetland – according to local or regional interests – for those purposes for which it used to be suitable before its degradation. It should always be a goal to design the ecosystem in such a way that lasting results are obtained, thereby minimising management costs. The ecosystem design includes the determination of the concentration level of nutrients, the level of productivity, balanced metabolism and – if considered realistic – management of food webs.

Subjects covered in this handbook

Inland water ecosystems are not static systems – in the long term a lake develops from a juvenile to an ageing state. The speed of this development varies with this size of the lake, with the latitude, the geology, the climate, etc. Every state in the natural development of every lake has its specific ecosystem characteristics, which can be revealed through palaeoecological studies of the sediment and peat deposits, the eco-historical archives of ecosystems. Knowledge about ecosystem dynamics, and about speed and direction in the development of the system, must constitute the general background when dealing with redevelopment and restoration projects which always demand a holistic approach in both time and space. Typical features in the evolution of lakes and wetlands are outlined in Chapter 2.

The impoverishment of the landscape with respect to surface water bodies, the lowering of the groundwater table and the qualitative degradation of lakes and other still remaining wetlands, call for a holistic approach to water resource management as stressed by Ripl *et al.* in Chapter 3. Arguments that there is an urgent need for re-creation of surface water bodies in the landscape as well as for programmes for management, preservation and optimum wetland utilisation are brought forward.

General descriptions of the structure and function of inland aquatic ecosystems are dealt with in every textbook on limnology and it is not necessary to repeat it here. However, the specific problem concerning vegetation succession and changes in primary productivity over time, especially in shallow lakes, needs to be elucidated (Pokorný, Chapter 4).

More than 30 years ago, Hrbáček demonstrated the relationships in the food web in ponds with and without fish. In Chapter 5, he summarises and synthesises his profound knowledge collected over several decades within this field of research. Based on Hrbáček's studies, the possibilities and restrictions of governing the food web are described by Matěna, Vyhnálek and Šimek (Chapters 7 and 8, Food web management). The naïve application and misuse of the principles of food web relationships as a means to 'restore' heavily polluted, natural lakes necessitates scientific clarification. The term food web management should not be replaced by the infelicitous 'biomanipulation' because the literal interpretation of this word makes one think of unfair change or falsification of life.

Methods to revitalise and restore inland water ecosystems have been elaborated and have already been applied on a small scale. Experience clearly indicates that pre-project ecological investigations are prerequisites for obtaining basic information about past and present relationships between the water body and its catchment area, about the history of the ecosystem and its prevailing structure and function. This knowledge is necessary for making reliable limnological diagnoses as well as for the technical planning of restoration projects (Chapter 6). The authors of this handbook aim to present experience from such projects as a means for widening and intensifying activity within this field of applied ecology.

Restoration methods and techniques are described in Chapter 7. Although defined methods are available, it must be remembered that every lake and wetland ecosystem has its individual characteristics. The treatment of them must, therefore, be given an individual design and the local possibilities should be fully utilised wherever possible.

Pre-project investigations, technical planning and accomplishment, post-project studies and results of restorative efforts are treated in each of the case studies reported in Chapter 8. A spectrum of degraded ecosystems, each demanding tailor-made restoration methods are described as examples which, hopefully, will serve as guidelines for future projects in a variety of degraded lakes and wetlands.

Because the landscape has been so impoverished with respect to wetlands, all opportunities to enrich it once again with these types of biotopes must be utilised. For a long time, gravel and peat deposits have been exploited in a technically primitive manner without any consideration for ecological planning. Before the start of mining, the groundwater table has often been lowered, an operation which has effects over a large surrounding area. This makes it difficult, or often impossible, to re-create aquatic biotopes. However, old pits and mined-out mires sometimes offer opportunities to be restored as wetlands. Before any new mineral extraction project of this sort is allowed to start, plans for the design of the wetland to be created after the mining has finished, should be presented. As a guideline, the general rule should be not to lower the groundwater table. Although this type of activity is most important for the preservation and creation of wetlands in the landscape, neither methods nor case studies are included in this book, because conditions can vary tremendously from case to case. We hope, however, that the information given here about the design and mode of reaction of wetland ecosystems should also inspire the creation and management of this category of aquatic biotopes.

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2. The evolution of lakes and wetlands

Sven Björk

Postglacial evolution of lakes and wetlands

The fact that inland water ecosystems are not static units but subject to continuous evolution has already been emphasised (p. 3). The speed of these changes is high in shallow, productive accumulation basins and extremely slow in deep, large, oligotrophic lakes. Palaeolimnological studies on ecological succession clearly demonstrate how important it is to take the *time-factor* into account when dealing with the restoration of systems which have been degraded and which now possess all the characteristics of rapidly ageing wetlands.

With the perspective of a geological time-scale, shallow lakes and inland wetlands are short-lived ecological units which become filled up with inorganic and organic material. During, and immediately after, the latest deglaciation of northern Europe, layers of minerogenic (i.e. of mineral origin) matter were deposited in depressions. Coarse particle fractions settled close to the shore beside the mouths of feeder streams, while clay and other fine particles settled further lakewards.

The landscape of northern Europe is characterised, originally, by its richness in lake and wetland basins produced by glacial action. In comparison, the region south of the glaciated area is poor in lakes. Former northern European lake and wetland basins were cleaned of sediment and peat by glaciers and, after deglaciation, the development of lakes in shallow moraine depressions also started, as it were from scratch, with a minerogenic bottom. South of the glaciated area, tundra conditions prevailed during the ice age. As a consequence of frost erosion – cryoturbation – the topsoil layers resembled the moraine of the deglaciated region. This is because they were also largely unleached, and the processes in these soils, including the *leaching of nutrients* and the influence of the successively *developing vegetation*, corresponded to that found in the moraine-covered, formerly glaciated part of Europe. The postglacial (Holocene) relations between catchment areas and lakes are, therefore, comparable in both regions. Because the developmental history of northern European lakes is so well known, the following description refers to these systems (*cf*. Digerfeldt 1972, Digerfeldt & Håkansson 1993).

Northern European lakes

After deglaciation, the fresh moraine was generally rich in nutrients. These were subsequently leached and transported by water from the drainage areas to the lakes, where a high production of algae was supported. In the sediment, the transition from minerogenic to successively

more organogenic (i.e. of organic origin) strata reflects this course of events. In several lakes, the lowest and oldest organogenic sediment consists of algal gyttja deposited during a short period of high primary productivity in the lake but during low supply of organic matter from the surrounding areas. Even in regions which now have a typically oligotrophic character, the oldest layers can consist of algal gyttja and lake marl, sediment types in sharp contrast with the present leached and unproductive character of the lake surroundings.

After this initial phase of high productivity, the lake productivity drops as the supply of nutrients from the catchment area reduces. This reduction is dependent partly on the decreased leaching and partly on the development of terrestrial vegetation accumulating and recirculating nutrients. However, changes in temperature, precipitation and vegetation cover during the last 10,000 years (Holocene) have caused variations in the productivity of lakes. Along with the climatic changes, the water level of lakes has also changed, and this, in turn, meant lakeward expansion or landward retreat and a reduction of the littoral macrophytic vegetation (Digerfeldt 1972). At the same time, the sedimentation limit, i.e. the highest level up to which organic particles settle above the shoreline, has been dislocated downwards (due to erosion) during periods of low water and upwards (because of deposition) during high water periods.

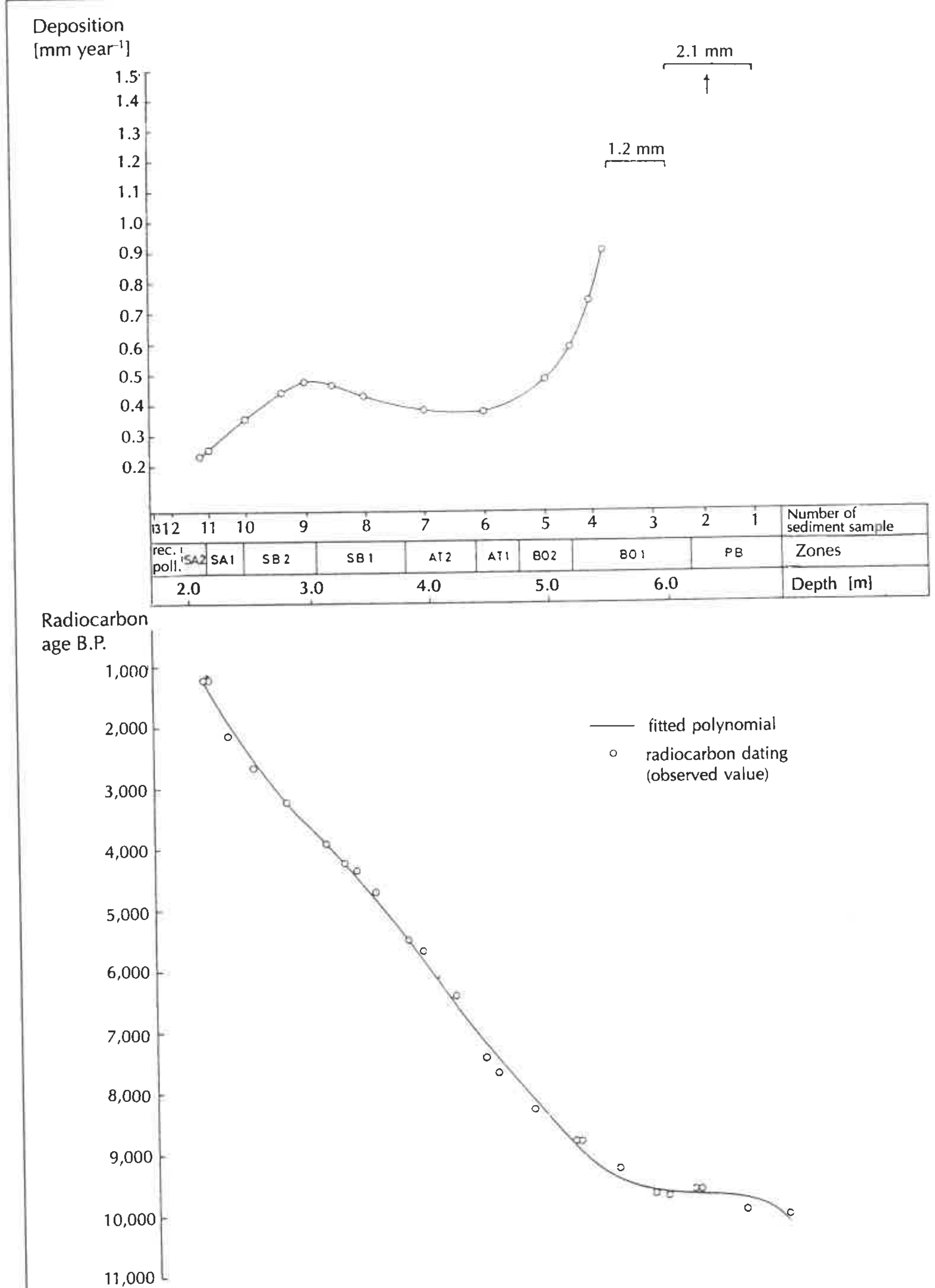
In an individual lake, the location of the actual sedimentation limit is lower along shores exposed to winds than along wind-protected shores. As prevailing winds are westerly in large parts of Europe during the ice-free season, the littoral zone is washed clean of organic particles down to a greater depth along the eastern, wind-exposed shores than along the wind-protected, western shores where particles can settle in more shallow, calmer water.

Growth rate of the sediment

The rate of increase in the sediment thickness is dependent on the productivity of the lake itself, on the supply of matter from the catchment area, and the efficiency of the mineralisation in the lake ecosystem. Intact lakes, within previously glaciated regions, had the highest productivity and sediment growth rate during the first phase of their evolution (*cf*. Digerfeldt 1972, Digerfeldt & Håkansson 1993). After that, these properties became reduced at the same time as the leaching and supply of nutrients from the surroundings decreased (Figure 1). The thickness of sediment varies from lake to lake as well as within the individual lake depending on the topography of its minerogenic bottom. The deposition of organic matter flattens the bottom such that the thickest sediment is found in the deepest depressions of the minerogenic bottom.

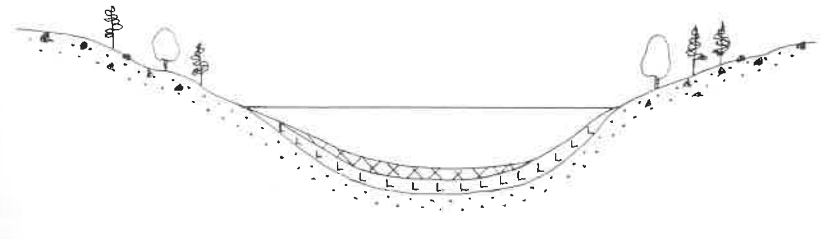
The normal, average thickness of organogenic sediment in, for example, a south Scandinavian lake with an age of 12,000–13,000 years is about 5 metres. In this region, the current sediment growth rate is about 0.2mm or less per year in a more or less intact, shallow oligotrophic lake, and about 0.5–1.0 mm per year in a shallow eutrophic lake. If a lake becomes polluted by the discharge of nutrient-rich sewage, the balance between production and mineralisation is disturbed. Organic matter accumulates and the sediment growth rate

Figure 1.
Lake Trummen, Sweden. Upper Diagram: Rate of sediment deposition. (The indicated rates during the Preboreal (PB) and Early Boreal (BO 1) periods are mean values.) Lower diagram: Radiocarbon dating of the sediment. From Digerfeldt 1972.

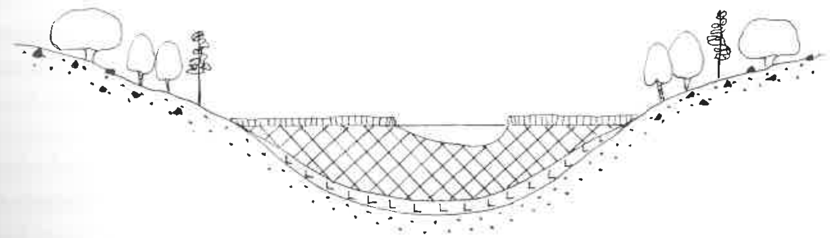


Key
Rec. poll. = recent pollution, SA = Sub-Atlantic period, SB = Sub-Boreal period, AT = Atlantic period, BO = Boreal period, PB = Pre-Boreal period, B.P. = before present.

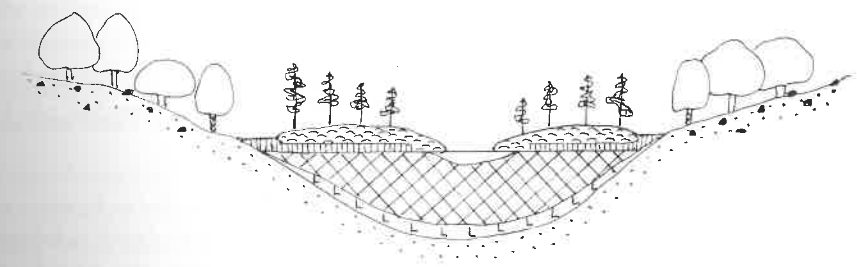
Figure 2.
The transformation of a lake from an open water ecosystem to a raised bog.



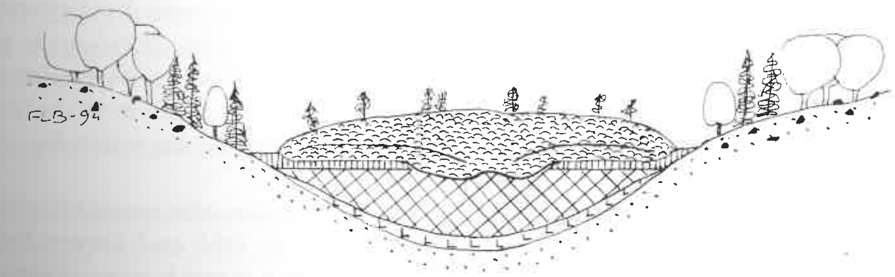
1. Lake with deposition of organogenic sediment (gyttja) on the minerogenic bottom.



2. The lake filled up with gyttja overgrown by peat-forming emergent macrophytes.



3. The macrophyte peat covered by *Sphagnum* peat. During a period with dry climate the bog is colonised by *Pinus*.



4. The former lake completely transformed to a raised bog with fast-growing *Sphagnum* during a wet climate period. The bog surrounded by a fen (lagg), i.e. the zone influenced by water from the mineral soil. The raised bog totally dependent on precipitation and airborn matter (ombrogenous bog).

Key
A B C D E F
A = moraine, B= sand-silt, C = silt-clay, D=gyttja,
E = peat of emergent macrophytes, F = *Sphagnum* peat.

can increase to about 10 mm per year. Thereby, the speed of ageing of a shallow lake is multiplied – the result of the addition of an internal nutrient supply from the sediment to that of the increased external loading (i.e. from the catchment area).

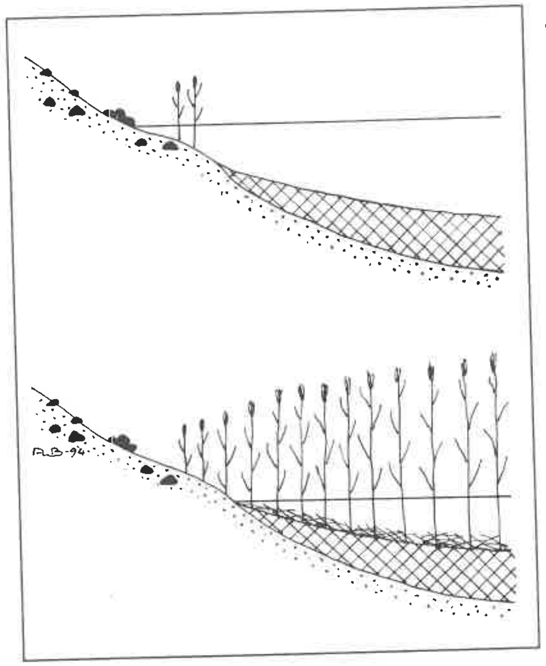
Primary productivity and vegetation succession in lakes and wetlands

In a naturally ageing lake, there is a successive decrease in the primary productivity based on plankton. However, a very dramatic change in the productivity takes place when the lake has become so shallow that it is possible for peat-forming macrophytes, such as the common reed (*Phragmites*), bulrush (*Schoenoplectus*) and cattail (*Typha*), to colonise the organogenic bottom (Figure 2). The plankton is replaced by communities of microorganisms (periphyton) developing in the water on stems and leaves of emergent, floating-leaved and submerged macrophytes.

The most productive phase in the whole evolutionary history of a lake is the period when the shallow lake has just been transformed to a wetland overgrown by emergent vegetation. The reasons for this sudden increase in productivity are as follows:

- there has been a continuous supply of nutrients from the catchment area to the lake which has acted as a trap for these elements;
- there is never water shortage in a wetland; and
- the perennial, highly productive emergent macrophytes are adapted to an efficient utilisation of the environmental conditions in all three media: sediment, water and air.

Figure 3.
The effect of water level lowering in a shallow lake on the quantitative development of macrophytic vegetation.



The macrophytes rooted in the sediment can thus make use of the accumulated, nutrient-rich resources which until now have not been available for deeply rooted plants during the previous stages in ecosystem development. Wetlands overgrown by plants such as *Phragmites* and other perennial, emergent macrophytes constitute the most productive ecosystems of any at the same latitude.

In northern latitudes, the preservation of organic matter in cold and oxygen-free sediment and peat is much more efficient than in waters of warmer latitudes where the mineralisation processes take place at higher rates and over a longer period of the year. This makes it especially troublesome to preserve northern, highly productive wetlands characterised by a high growth rate

in the accumulation of coarse detritus produced by emergent macrophytes. The general ecological succession and terrestrialisation process includes stages where *Typha*, *Schoenoplectus* and *Phragmites* are replaced by *Carex*, and the formation of peat above the sediment prepares for the invasion of *Salix* and *Betula*. Lowering of the water level in shallow lakes means a tremendous speeding up of the ageing process (Figure 3).

Palaeolimnological studies

The sediment composition reflects the communities which have inhabited a lake over the millenia. Provided the sediment growth rate is known, it is possible, by means of chemical analysis to determine the supply of different elements during the various phases of development. By means of palaeolimnological investigations, using a long series of analytical methods, the developmental history of both the lake and its catchment area can be reconstructed. The presence of frustules of diatoms, cysts and other remnants of algae, remains of insects and cladocerans, seeds and other diaspores, roots, etc., allows for an ecological interpretation of the conditions prevailing when the different layers of sediment and peat were deposited. Pollen analysis, dating of sediments by determination of the content of radioisotopes of carbon (^{14}C), caesium (^{137}Cs) and lead (^{210}Pb), and palaeomagnetic investigations, are used to determine the age of sediment layers. When the age is known, the sediment growth rate and the accumulation of different elements per unit of time can be determined (*cf.* Digerfeldt 1972). This means that both natural and man-induced leaching and transport processes from the catchment area to the lake can be revealed through studies of the sediment archives (Figure 4).

Palaeolimnological investigations are often an essential prerequisite for the design of restoration plans because the aim is usually to try to re-create ecosystem qualities which have been lost. Past and present relationships between the catchment area and the lake have to be elucidated and used as guidelines in restoration planning. With a collapsed ecosystem as the patient, and with a holistic approach in space and time as a necessity, the pre-project investigations need to be organised as team-work for penetrating different sectors of the system and for providing analytical data for solid, ecological-diagnosing, syntheses.

Important physical processes

The influence of man on lakes and wetlands often eliminates important natural agents such as ice movements and seasonal water level fluctuations. Temperature changes in the Holocene climate has involved variations in the development of ice cover on lakes. A general feature is that ice can have an influence on the shores both when it covers the whole lake and at the ice break-up. In cold winter periods, especially at night when the ice cools down and contracts, tension cracks appear in the ice cover. When cracking, a sound like thunder is heard, the 'bellowing' of Scandinavian lakes. The cracks are immediately filled with water which then freezes. In warmer periods, as on sunny days, the volume and area of the ice cover increases and a very strong pressure is exerted against the shorelines, which are subject

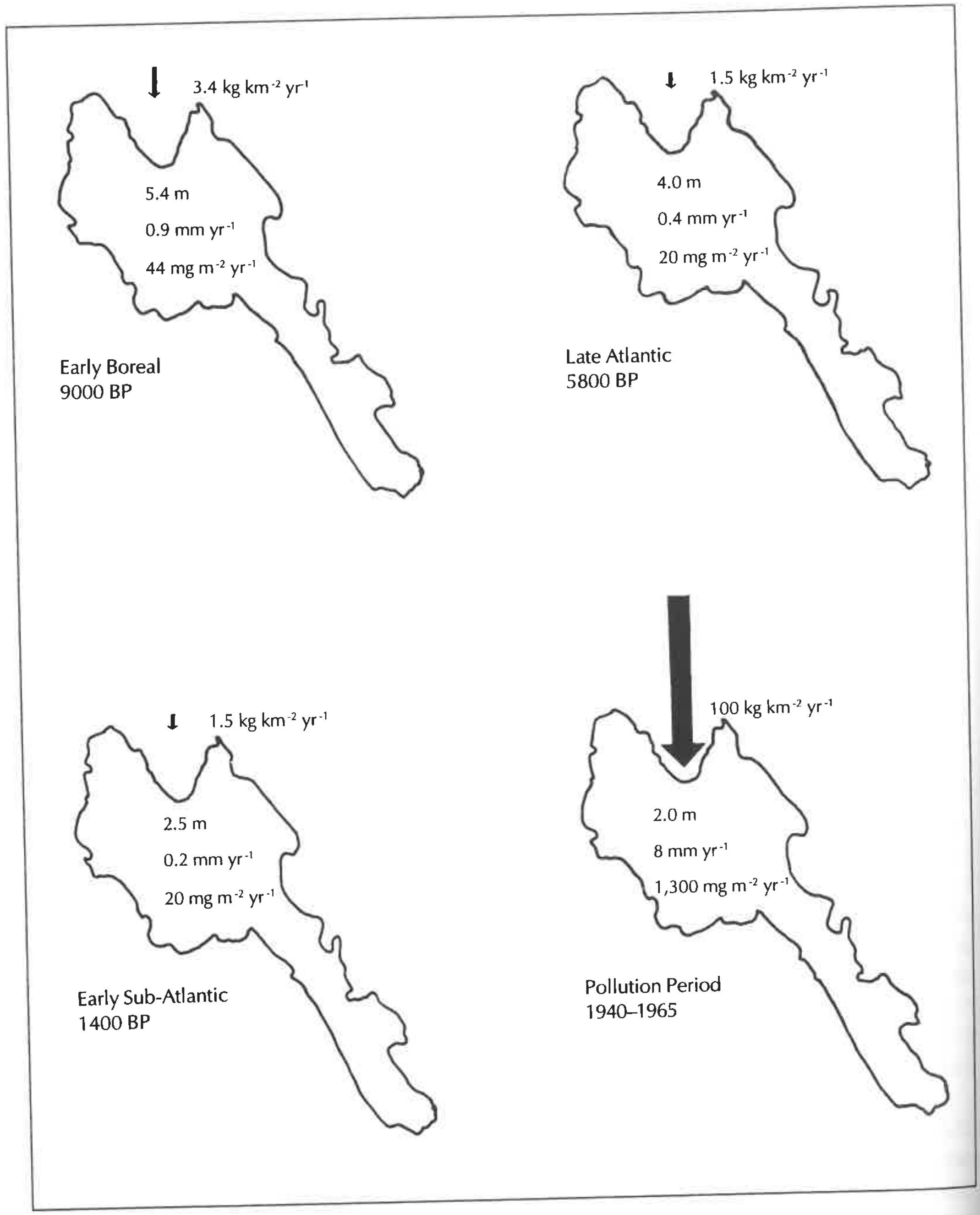


Figure 4.
Lake Trummen, Sweden. The transport of phosphorus (indicated by arrows) from the catchment area for deposition in the sediment during four periods. The figures within the lake denote: lake depth, rate of deposition, and the annual deposition of phosphorus. Compiled on the basis of data from Digerfeldt (1972). From Björk *et al.* 1972.

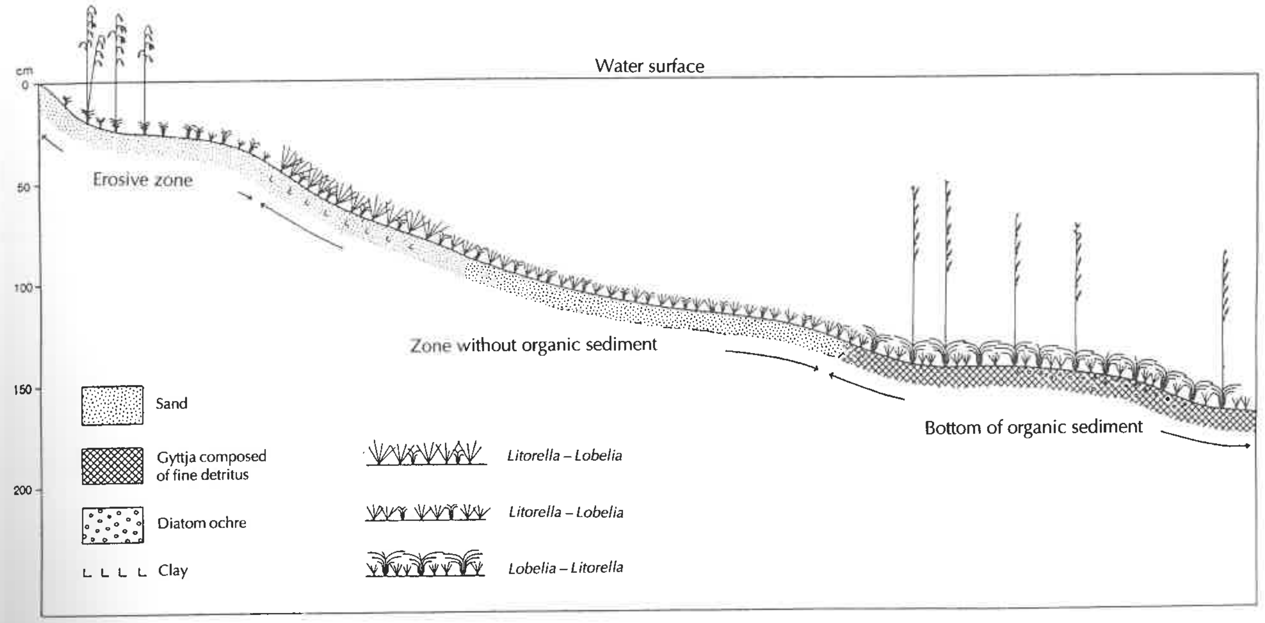


Figure 5.
Schematic distribution of vegetation in different littoral zones in a *Lobelia* lake (Lake Fiolen, Sweden). The upper portion of the minerogenic zone is under the erosive influence of ice movements. The minerogenic zone is lakewards followed by the organogenic bottom. From Thunmark 1931.

to ice-pressing. With the changes in the winter temperature, this process is repeated continually and means a successive increase in ice pressure. Over the vertical range of the shore, the erosive zone (Figure 5) which is influenced by the ice, can be cleaned of vegetation, bottom material (including big boulders) can be forced landwards, and building constructions demolished. In regions covered by coarse moraine deposits, barricades of boulders along the shores have been built up by ice-pressing (Figure 6). These barricades were probably mainly formed in the early Subatlantic period (about 2,000 years before present), when the ice-pressing is considered to have been stronger than at present.

In a very large number of Scandinavian lakes, the water level nowadays reaches far below the barricade, indicating that the lake has been lowered. In intact lakes, however, the water level is still found at the base of the barricade (Figure 3).

Besides ice-pressing, another type of ice movement occurs, which is also important in the physical sculpturing of the shore. This is ice-pushing, brought about by ice-floes in connection with the break-up of the ice. When a sudden break-up of the ice is caused by strong winds, the ice-floes are pushed against the shore and piled up along the wind-exposed side of the lake. In this way, the shore is also cleaned of vegetation and trees are de-barked on the lakeward side.

Whenever possible, in the restoration and management of lakes and wetlands, the conditions for the formation of qualitatively good ice in order to make use of ice movements, as well as of water level fluctuations, should be re-created. The utilisation of ice movements is a cheap way to protect lakes from overgrowth and heavy detritus production by macrophytes, and to preserve open shores suitable for wading birds, etc.

Figure 6. Lake Fiolen, Sweden (summer, above; winter, below). Littoral barricade of boulders pushed together by the ice. Photo: Einar Naumann 1917.

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3. A holistic approach to the structure and function of wetlands, and their degradation

Wilhelm Ripl, Jan Pokorný, Martina Eiseltová and Steve Ridgill

A holistic approach to wetland management

Only by considering all the processes in a catchment area, over space and time, can the function of wetlands within the catchment, and their possible degradation, be understood.

This is known as a holistic approach. By relating living processes, the water cycle and energy use together, these processes are seen in a new light. An understanding of how the energy-driven processes, the water cycle and the living processes are mutually interrelated and inter-dependent allows new insights into how man's interference is de-stabilising ecosystems. A holistic approach to wetland management is fundamental to the task of putting right the imbalance and re-stabilising the system.

The condition and life-expectancy of catchments and natural water systems is determined by their efficiency in minimising non-regenerative, dissolved matter losses from the catchment. Man is de-stabilising these ecosystems by his random distribution of energy in time and space, by decreasing system efficiency (and hence stability) and by greatly increasing irreversible matter losses. In the last hundred years, irreversible charge flows (matter loss, mainly base cations) via surface waters to the sea have multiplied, often more than a hundredfold, such that matter losses from agricultural areas are often more than one tonne per hectare per year. Use of fertilisers will hardly compensate for this loss, nitrogen and phosphorus together only accounting for 1 to 2% of this matter loss.

Man's influence on the landscape has affected the water cycle and its coupling with energy flow and the transport of matter. The recreation of vegetation and a water-saturated soil is a prerequisite for the minimisation of water and matter losses. Wetlands, as efficient dissipators of energy in time and space, help to ameliorate the climate, shorten water cycles, maintain high groundwater levels, retain high nutrient and mineral loads, and minimise matter losses. All these aspects must be taken into account for the redevelopment of catchments and their natural water bodies.

Historical perspective of landscape development

Palaeolimnological studies of south Swedish lakes with respect to landscape development, conducted in the early 1970s by Digerfeldt, revealed the conditions of the catchment areas

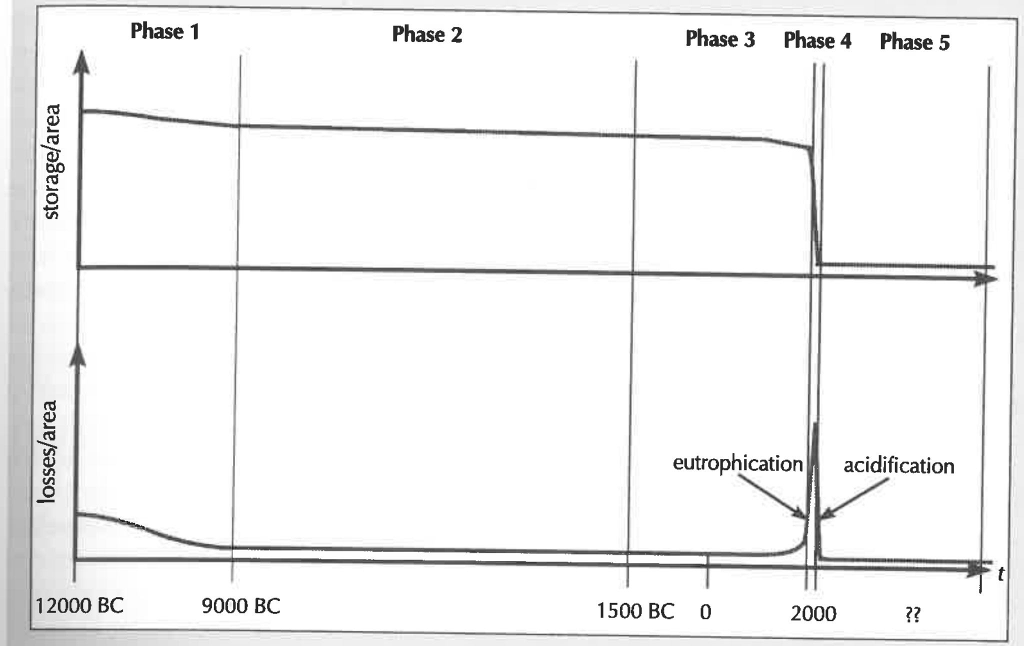


Figure 1. Area matter losses with water transport in postglacial North-European catchments.

Phase/period of time	Development of reactivity and concentration of charges in recipient waters		Loss factor of bases	Loss factor of nutrients
	pH	conductivity µS cm⁻¹		
Phase 1: 12000 to 9000 BC Pioneer vegetation	6.5 – 8	20 – 200	2 – 3	2 – 10
Phase 2: 9000 to 1500 BC Climax condition	6.5 – 8	10 – 40	1	1
Phase 3: 1500 BC to 1850 Anthropogenic development	4 – 9	30 – 300	2-5	5-100
Phase 4: 1850 to 2000 Desertification	3 – 11	50 – 1000	5 – 100	10 – 1000
Phase 5: after 2000 Restitution before next glaciation?	4 – 10	decreasing	low	decreasing

since the last glacial period. Initially, directly after glaciation had ceased, matter deposition rates in lakes were relatively high (Figure 1). Over the next 3000 to 4000 years the rates of sedimentation for various deposited compounds gradually reduced by a factor of between 4 to 10. From then on, the low deposition rates of approximately 0.1 to 0.2 mm per year remained rather constant until the second half of the 19th century. Maximum deposition rates then occurred, as sewage from cities and urban areas was diverted to lakes and deposition rates increased nearly a hundredfold to almost 8 to 10 mm per year. Examination of the different parameters of these lake ecosystems revealed a high degree of auto-correlation. This means that organic matter, nutrients such as phosphorus and nitrogen and even base cations such as calcium and manganese showed the same relative deposition pattern (Digerfeldt 1972).

Reflecting on these facts leads to several conclusions:

- deposition rates within the lake are directly proportional to the process density in space and time within the catchment;
- processes in the catchment are coupled and their energy and potentials are channelled by the vegetation development;
- the amount of transported matter to the lake is inversely proportional to the increased organisation of biocoenosis in the landscape;
- disturbances result in highly increased matter flow rates from the landscape to aquatic systems;
- the amounts of increased matter flow not deposited in sediments are irreversibly transported to the sea;
- there exists a relationship between the deposited amount (in sediment) and the irreversible matter flow (to the sea), which is given by the landscape's hydrograph, the morphometry of the lake basin and the metabolic processes in the lake;
- for a given morphometry, a higher metabolism will result in a relatively lower retention of matter in the lake. However, these relationships are not linear ones but are related to the spatial and temporal distribution of the coenosis. For example, the short-circuited bio-film structures on macrophyte surfaces (periphyton) show a much higher retention of matter than planktonic assemblages;
- the former viewpoints show that water protection has to be carried out by controlling processes and flow patterns in the landscape rather than just the management of the aquatic ecosystem;
- the main causes for the deterioration of aquatic systems occurred together with the large-scale interference with vegetation and manipulation of the water regime which degraded ordered structure in the landscape and replaced it with increased randomness.

The above-mentioned points show that a holistic approach to the integrated landscape and water transport system is necessary for the redevelopment of the environment. The

development of high loss rates and system deterioration is connected with industrially-caused interference (i.e. vegetation destruction and extended areas of sealed soil surfaces) with the cooling capability of vegetation cover, the latter being a consequence of the deterioration of the dissipative properties (the distribution of energy, potentials and processes in the landscape).

Water cycle and matter losses

The central point in managing landscape and water is to examine the role of energy and its dynamic agent, water. In nature, water is almost exclusively the medium with which transport takes place and chemical reactions are performed. Quantity and quality of water in the landscape, and associated aquatic systems, are almost entirely determined by energy and its temporal and spatial distribution in the landscape, and the interactions with the prevailing coenosis. The coenosis partitions energy into a water evaporation process, a chemical dissolution process and a biological production process. All these three processes are cooling processes which have as their cyclic counterparts heating processes: condensation, precipitation and respiration. Thus, the water cycle can be considered as an energy processor that channels energy into dissipative cyclic processes according to its inherent structure under given environmental conditions.

Energy-Transport-Reaction (ETR) model

The interrelation of energy, water transport, physical, chemical and biological processes are described in a new ecological model, called the *Energy-Transport-Reaction (ETR) Model*, proposed by the first author (more detail can be found in Ripl 1992, 1994). Ecosystems are optimised in their structure as dissipative energy processors, through the actions of the water cycle, by being in phase with the cyclical additions of energy. Man interferes with the regular, phased, energy input, and its regular dissipation by a selected, optimum biocoenosis, through the introduction of random elements into the system. Man's input of fossil energy coupled with the transport of material has resulted in excessive energy- and matter flow densities. The degradation of biocoenotic structures and the distortion of the water cycle has resulted in an increased lack of cooling properties. The out-of-phase oscillations thus created, result in highly irregular perturbations in natural systems, which can be manifested as the increased likelihood of floods and droughts, extremes of temperature, abnormal upper atmospheric conditions, mass die-off of plant and animal communities, etc., i.e. randomly distributed processes in both space and time.

Development of vegetation cover

The water cycle and the development of vegetation cover (coenosis), and the control of matter losses, are intrinsically linked together. To enable a better understanding of the consequences of such links, and man's interference, the development of the landscape since the end of

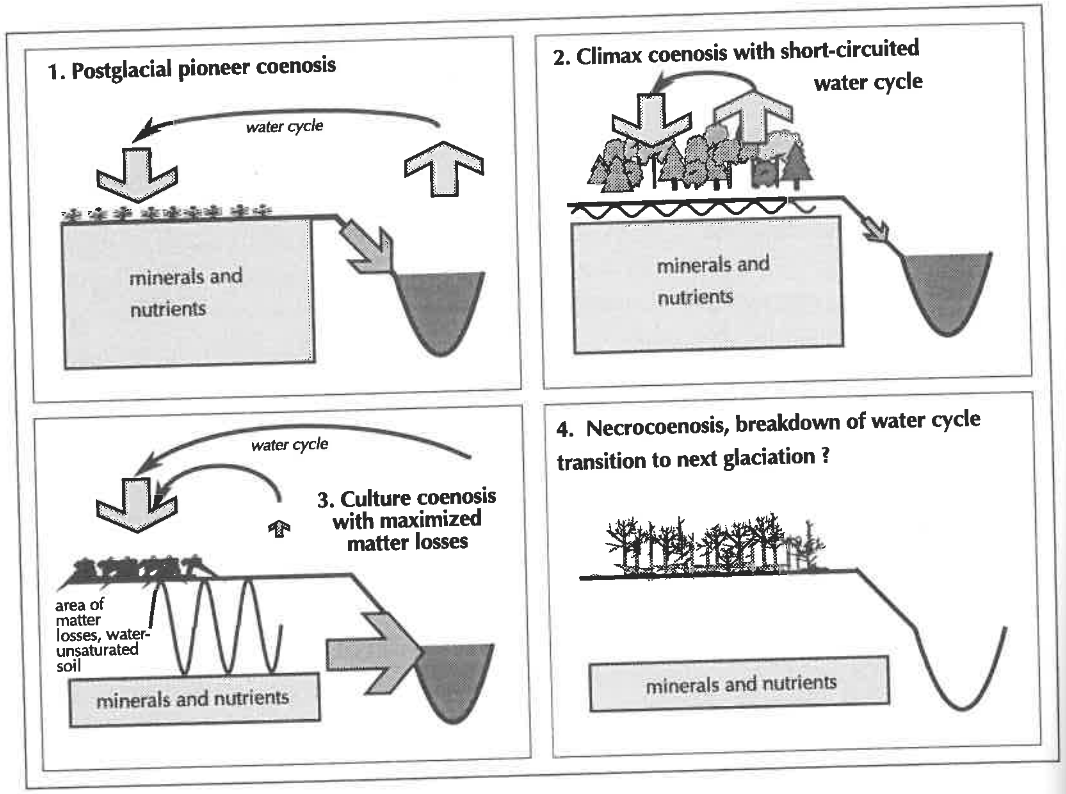


Figure 2. Water cycle and matter losses in various stages of landscape development since the last glaciation.

the last glaciation is considered below in more detail. Figure 1 shows areal matter losses with water transport in postglacial, north European catchments in five time phases which correspond to (1) vegetation expansion, (2) climax vegetation, (3) man's gradual interference with agriculture, (4) the dramatic consequences of the industrial age, and (5) the predicted collapse. The development of vegetation cover and areal matter losses is schematically illustrated in Figure 2, where four stages correspond similarly to the above: (1) pioneer and (2) climax coenosis, (3) man's interference and (4) the breakdown of the water cycle.

Two stages or phases (Figure 2, stage 1 and 2) in the development of landscape can be recognised:

1. *an establishing phase* with organisms expanding, structures evolving and matter losses diminishing, and
2. *a coupling phase* which increases the system stability and where losses are at a minimum.

1. Postglacial pioneer coenosis
The development of vegetation and soil begun after the glaciers had retreated and streams had sorted the soluble and suspended material to form the landscape. Waters showed increased biological activity and metabolic processes, caused by increased nutrient influxes due to both chemical surface erosion and soil formation. As the vegetation expanded, ions were taken up, a certain amount of energy was used, the pH was lowered and the dissolved

ions were susceptible to rain erosion. Therefore, in this period, matter losses were initially high (Figure 2, stage 1) but loss rates continually reduced as an interlaced vegetation cover developed in the whole catchment.

2. Climax coenosis
The rate of matter loss became reduced as the coenosis, and the cyclic processes carried out by the coenosis, developed and space became limitating. The growing spatial-temporal interlacing of organisms in the catchment led to an increase in system efficiency, a reduction of matter losses, and the interlacing of processes, which even affected the development of climate (Figure 2, stage 2). Soil was the interface between vegetation, the substratum and the water cycle, where plants recycled matter and nutrients, controlled the water cycle and primary production and energy level by evapotranspiration, and decomposition of detritus was controlled by the water cycle. The system became closed as losses became minimal, and only net production (harvest) caused siseable losses. New growing biomass would take ions from the mineral ground (easy dissolvable minerals).

Degradation of vegetation cover

The next two stages (Figure 2, stage 3 and 4) show the result of man's interference in the landscape:

3. *a phase of system disturbance* with an increased mineralisation of organic matter and accelerated matter losses from the catchment, and
4. *a transitional phase to the next glaciation* due to the water cycle breakdown and the coenosis die-off.

3. Man's interference and the degradation of the biocoenosis
The climax coenosis lasted in northern Europe until about 1500 B.C., when man began cultivating these systems with increasing intensity. However, compared to present day activities, cultivation was orientated almost always to human needs, with management being in accordance with spatial and temporal phases and less randomly in relation to the loss-determining dynamics of water. Interferences in the structure of the system led already to a reduction of its thermodynamic efficiency (minimised charge losses), which could, however, still be well buffered by the matter storage in the topsoils. The control mechanisms within the system still allowed sustainable development at the low population densities of man.

The transition to stationary agriculture and the exploitation of the forests, meant that matter losses in the catchment rose considerably. Clearance of land for agricultural purposes often involved destruction of forests by fire with consequential loss of much nutrient-laden ash. The increased flow of nutrients, however, was used in the further spread of littoral vegetation of rivers and lakes. This helped to retain the extra nutrients in the plant biomass, and the rate of sedimentation in most lakes and rivers, as well as the nutrient level in the water, was still very low.

As agriculture developed, the first step of net production (harvest) was on account of the mineralisation of the existing organic matter. However, the increased clearance of forests, exposure of bare land, drainage of agricultural land, and the increasing populations along rivers, which became used for transport and the first manufacturing processes, accelerated the matter losses from the catchment. The lowering of water levels by man increased the rate of mineralisation and resulted in matter-loss from the water-unsaturated soil zone (Figure 2, stage 3). In the water bodies, the increased inputs of nutrients can be documented by higher deposition rates of sediment – the beginning of eutrophication.

Before the interference of man, the electrical conductivity (a measure of ionic concentration) in run-off water was *c.* 10–30 $\mu S\ cm^{-1}$ (rain water as surface run-off from areas with an intact, moisture-keeping reticulum of humic substances). For phosphorus, the efflux of nutritive substances from an intact vegetation lies at *c.* 10 $\mu g\ P\ l^{-1}$ or less, and for nitrogen *c.* 50–300 $\mu g\ N\ l^{-1}$. Considering northern European catchments, electrical conductivity was gradually raised to *c.* 150–250 $\mu S\ cm^{-1}$ by increased percolation through the soil. The man-made increase in conductivity was accompanied by a fivefold increase in phosphorus and nitrogen concentrations. Mineralisation was only able to supply the needs of primary production until available nutrients were depleted and fertilisation became necessary for the continuation of intensive agriculture.

The growth of cities in the industrial era brought dramatic leaps in matter losses. Water, in many cases, was extracted from the ground and discharged into surface waters, along with putrescible sewage. At the turn of the 20th century, conductivity rose to almost more than 300 $\mu S\ cm^{-1}$. Biocoenoses, loaded far beyond their range of control, almost lost their properties for the retention of nutrients in water systems. By the natural control span of the biocoenosis is meant, the control of total matter flow which is affected almost by the organisms themselves (by evapotranspiration), and in which zero-net production can be maintained, almost without inorganic matter losses.

The establishment of our modern industrial society led to a rural exodus and the necessity to cover the rising demand for foodstuffs by intensification of agriculture. This development was only made possible by an enormous distortion in water cycles. The impounding humidity which hampers agricultural production, as well as the stable podsol or gley structures, were finally removed. Groundwater levels began to oscillate with each rainfall event. Consequently, there was a quicker erosion of humic particles (important for water- and nutrient retention) from topsoils, as well as the oxidative formation of sulphuric and nitric acids, which led to dissolution and influx of lime to the lake, and by biogenic precipitation into the water sediments.

Losses of matter from the landscape increased even further and today they have risen by 50 to 150 times in comparison to unaffected soil systems. From agricultural areas, areal losses of dissolved solid matter are regularly over 1 tonne per hectare annually. In north German rivers, nutrient concentrations of phosphorus now amount, on average, to 200–500 $\mu g\ P\ l^{-1}$; nitrogen concentrations to 2–4 $mg\ N\ l^{-1}$, and conductivity to between 400 and 1000 $\mu S\ cm^{-1}$.

The increase of nutrient and mineral losses may not only be diagnosed from the elevated concentrations in lakes and rivers. In addition, the water run-off from the landscape has been increasing. By now, the coefficients of discharge (regional run-off to precipitation ratios) have approximately doubled in northern Europe, together with a reduced short water cycle (dew generation), in comparison to a landscape with an intact vegetation cover. Decreased evaporation potentials in areas with poor vegetation, remote from coasts covered with vegetation, is likely to result in a gradual regress of the amount of precipitation and, for some time, in a larger variance of climatic parameters (Figure 2, stages 3 and 4).

The continuing losses of nutrients must be compensated by means of intensified fertilisation. However, this in no way can replace the total losses of matter. One consequence of the continuous losses of water soluble material (e.g. carbonate) is the increased concentration of immobile components almost insoluble in water, e.g. quartz, heavy metals, and not easily soluble toxic organic residues from the industrial as well as from the agricultural production (e.g. pesticides) (Figure 3).

4. Breakdown of the water cycle

With increased matter losses, the growing vegetation increased the proton flow (uptake of base cations) and if it was not sufficiently buffered, the pH decreased, accompanied by the dissolution of toxic substances which then entered through the roots and caused the die-off of vegetation and the whole coenosis. Together with the coenosis die-off, the cooling system is destroyed and the water cycle breaks down (Figure 2, phase 4).

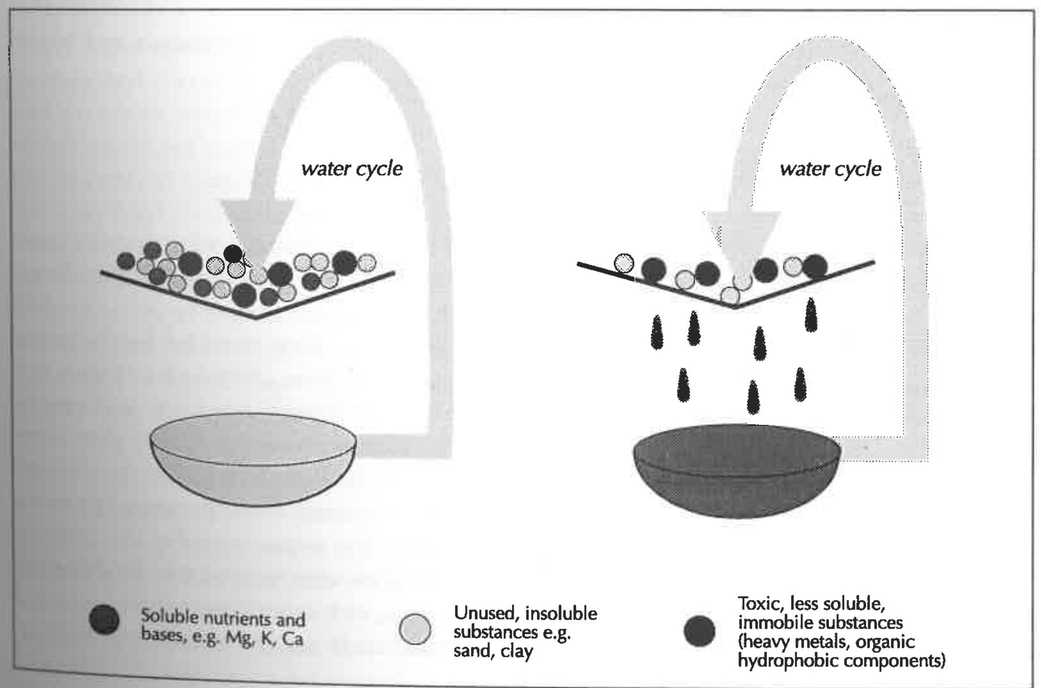
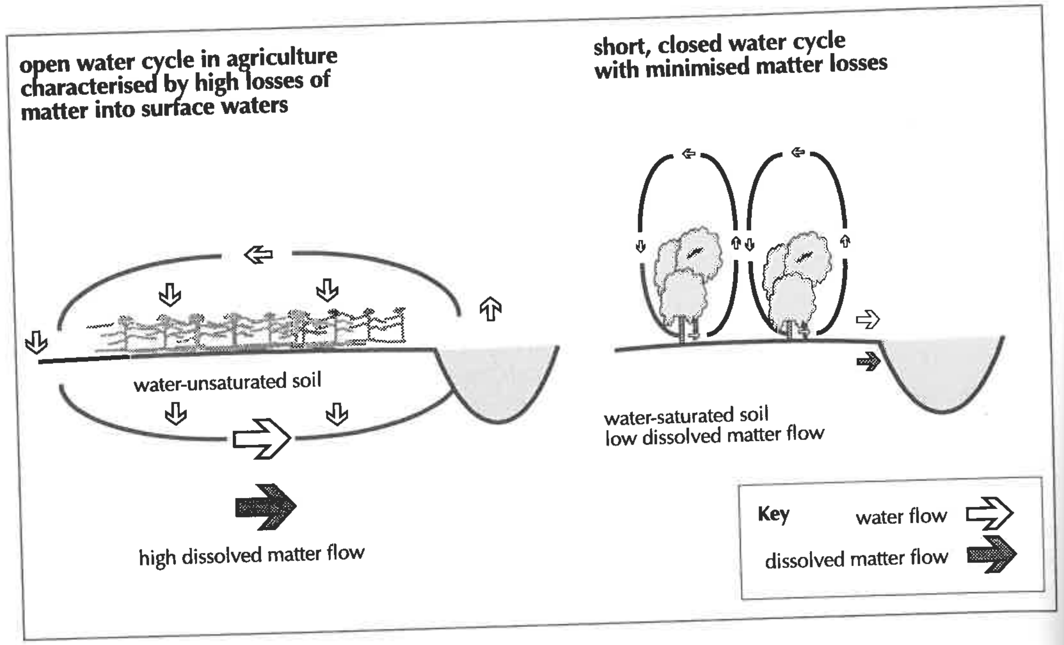


Figure 3. Dynamic model for the accumulation of toxic substances in the topsoil.

Figure 4.
Water and matter flow through vegetation and soil.



Although the processes of irreversible areal losses of matter (base cations and nutrients), transported via surface waters to the sea, occur naturally, they are greatly accelerated by man and his non-sustainable management of water resources. The greater the intervention in the water balance, the faster the degradation of the catchment proceeds. In more southern countries, these degradation processes can escalate even more rapidly because of the higher solar energy input. Finally, the water cycle collapses and the process of desertification expands. This has already been seen in the ancient cultures of Mesopotamia and North Africa.

'Short' and 'open' water cycles

The dissipation of the sun's energy by the water cycle is mainly through what is collectively known as evapotranspiration. The completion of the water cycle requires an area of landscape relatively cooler over which condensation (dew) and possible precipitation (rain or snow) can occur. Such heat sinks, or relatively colder localities, are often provided locally to the sources of high evapotranspiration, by extensive areas of vegetation cover such as forests and wetlands. Water is cycled over short distances, in frequent small amounts, and can be thought of as a closed (little water lost to the system), short-circuited or simply *short water cycle*. In contrast, where the landscape provides no such heat sinks of sufficient thermal capacity (e.g. sparsely vegetated and urban areas), great extremes in temperature can occur and any evaporated water must finally condense a long way away, over the sea, coast or distant mountain range. This cycle can be called a long or *open water cycle*. In this case, most water is not recycled locally and is lost to the system, rain events being infrequent and varying greatly in amplitude. These two types of water cycle are shown schematically in Figure 4.

Open (or long) water cycles result from an unphased, random distribution of energy over the landscape (catchment). Lack of cycled water means less cooling of the local climate by evapotranspiration and greater energy fluxes. This situation is found in catchments with little or no expanse of vegetation, where forests have been cleared and agriculture is mainly arable (cereal or root crops), or where cities and towns are concentrated. Such landscapes have greatly exaggerated amplitudes of temperature (heat energy fluxes). With groundwater levels generally low, soils are usually dry and heated by high energy fluxes, which increases the oxidative processes and micro-organism activity, releasing more soluble ions. Irregular rain events cause groundwater levels to fluctuate wildly. Water percolating through the mainly dry but occasionally wetted soil removes available soluble ions on its way to rivers, lakes and eventually the sea. Open water cycles are therefore high in irreversible matter losses, with *water-unsaturated soils*, high ranges in temperature (extreme diurnal and seasonal amplitudes), and rainfall being irregular and often reduced.

Short (or closed) water cycles, on the other hand, are much more efficient systems with fewer energy losses and more regular energy fluxes of reduced amplitude (reduced potentials). A landscape (catchment) with a short water cycle necessarily has wetlands and a high vegetation cover, the biocoenosis being able to provide cooled areas by evapotranspiration and thus cycle the water rapidly by condensation. A more frequent and regular rainfall maintains a higher groundwater level, which in turn reduces reactivity and microbial activity in the soil, and any surface-water run-off, not retained by the vegetation, removes little in the way of irreversible matter losses. High groundwater levels mean *water-saturated soils* where the metabolic processes are slowed down, the energy flux is reduced, and the transport of matter away from the soil system is minimised.

Short water cycles in the catchment inevitably means wetlands. Wetlands have a high biomass and thermal capacity, high evapotranspiration rates (energy dissipative potential) and a coenotic structure which maintains a high groundwater level and water-saturated soil which minimises matter losses. Considering the dissipation of heat by wetlands, for a typical day in June, in Central Europe, the global solar radiation energy is about 16 MJ m^{-2} day^{-1}. For a wetland reed stand, a typical figure for evapotranspiration might be 4 mm day^{-1} of water (Přibáň & Ondok, 1985). Assuming 4 mm day^{-1} water, the evapotranspiration accounts for about 9 MJ m^{-2} day^{-1} (as a minimum on a hot day) which is cycled to the atmosphere. This is an even higher proportion of incoming radiation if net solar radiation (global radiation less albedo) is considered. Equivalent figures for agricultural land may be only 1 mm day^{-1} water or less, meaning that the energy not dissipated by evapotranspiration must greatly heat the land, causing dust to be convected high into the upper atmosphere.

Open water cycles have therefore detrimental and often considerable effects on the climate of a catchment. A more random distribution of energy, in time and space, affects rainfall patterns, temperature ranges and atmospheric conditions (high winds and unstable vertical air movements). In northern central Europe, for example, it has been found that for many regions the pattern of annual precipitation has been altered. The normal summer peak in rainfall of July and August has been shifted to the autumn months of September and October. Though the total annual precipitation appears not to have been reduced, less water is available during the summer growing season. Thus open water cycles, as a result of man's activities in the landscape, produce significant climate changes.

Matter losses for different land-use types

The differing amounts and quality of vegetation in the catchment, as we have just seen, result in short or open water cycles with different degrees of matter loss. Figure 5 illustrates the proportional amounts of water evaporation, surface water run-off and water seepage through the soil, following a given precipitation on different landscape types, and their associated matter (mineral and nutrient) losses. Wetlands and water bodies have a high ratio of evapotranspiration and surface water run-off but no water can seep down through the soil and little matter is lost to the system. Forest-moor habitat shows a similar pattern but with some water percolating through the soil, a slightly higher surface run-off and matter losses similarly low. The other example habitat types all show proportionally large matter losses through the soil or substrate. City urban areas can have similar amounts of surface run-off as do wetlands, but unlike wetlands where metabolic processes are slow, this surface run-off removes relatively high amounts of matter from streets and storm drains. The high areal matter losses associated with extensive arable agriculture, and to a large extent also meadows, are multiplied throughout large areas of many typical catchments.

Not every agricultural land-use necessarily implies an open water cycle with high matter losses. Two extremes of agricultural practices are:

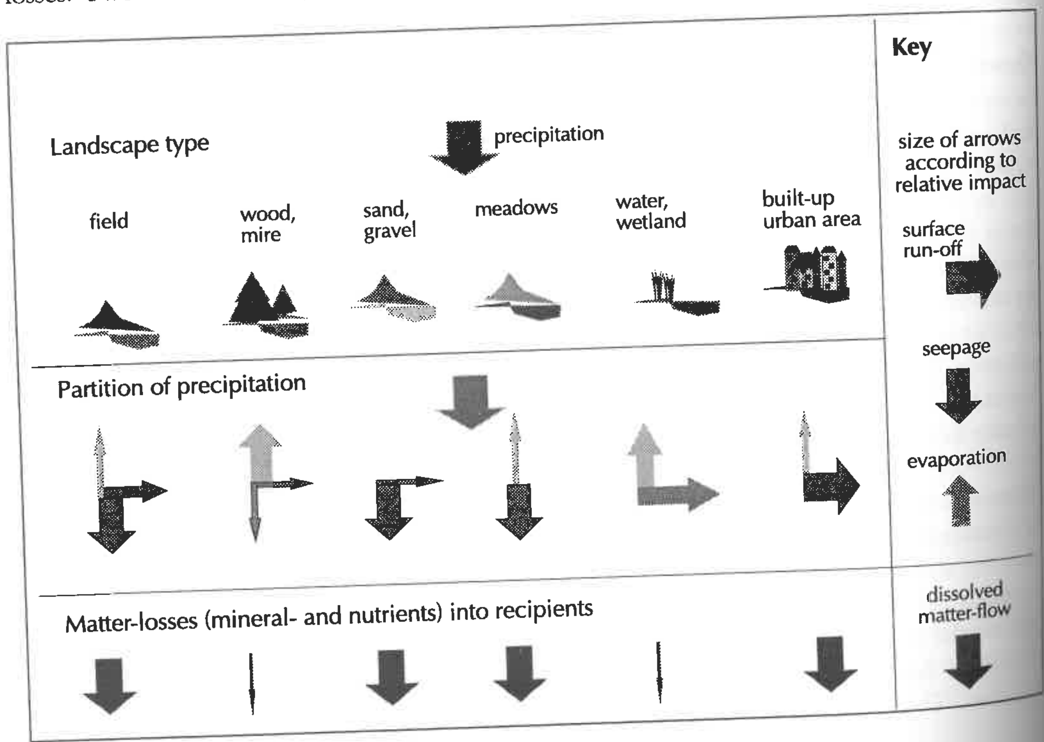


Figure 5. Water and matter transport with precipitation and run-off in various land-use types.

(i) agriculture based on cereals, a *'dry agriculture'* of crops developed from 'steppe-like' grasses of basically dry habitats, which leads to a fast degradation of soil quality, and

(ii) agriculture based on rice-paddies, a *'wetland agriculture'* with a short water cycle and low matter losses – as an example of a sustainable use of land.

It was shown for mainly rural catchments that the specific areal losses from agriculturally-used areas are, within an order of magnitude, between 900 to 1500 kg ha^{-1} $year^{-1}$ of dissolved solid matter. This is without including sodium ions which are almost always man-introduced nowadays. Smaller figures were obtained from forested areas, already low in base cations depleted by previous leaching. It is the agricultural practices which result in the depletion of base cations and the acidification of topsoils. The above figures are equivalent to 20 kg ha^{-1} $year^{-1}$ proton loss for the total catchment, of which 5 to 10 kg ha^{-1} $year^{-1}$ is calcium ions. The areal contribution of proton flow by industry and traffic (burning of fossil fuels) is much lower than from agriculture and forestry. That means that the primary production process is the main problem, unless it is recycled (net productive process). The point-sources, i.e. wastewater, cause 20% of dissolved solid matter losses, whereas about 80% of matter losses are caused by non-point sources, i.e. mainly agricultural sources. The irreversible matter losses from agriculture and urban areas are compared in Figure 6.

This has been the case in almost all cultures based on dry 'steppe-type' crops, i.e. crops intolerant of flooded conditions, such as wheat. Agriculture is necessarily on drained-fields, to avoid any flooding (or water-saturated soils), or on dry fields with water supplied by irrigation, which, in both cases, leads to high levels of soil activity and leaching of dissolved ions during sporadic rain events. Soils are warmed excessively in summer, aiding even greater soil activity and the prevention of cool condensation points and short water cycles.

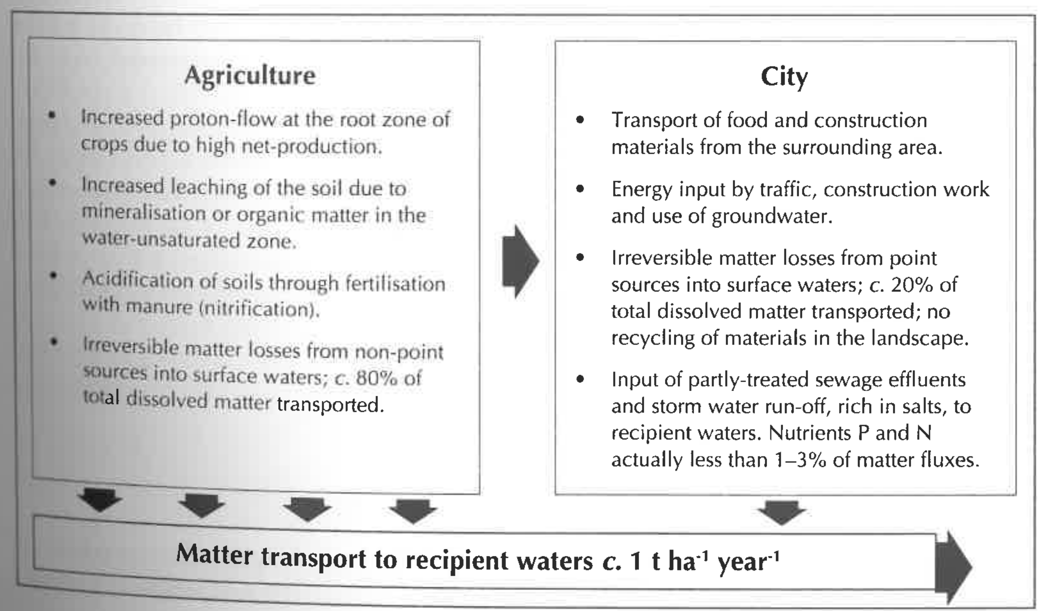


Figure 6. Destabilisation of the ecosystem by urbanisation and agriculture.

However, agriculture based on wetland farming, as for example in rice-based cultures, shows a much more sustainable landscape system. Soils are water-saturated, matter losses are kept to a minimum, short water cycles are encouraged, soil fertility is maintained and the climate remains more predictable.

Holistic ecologists

Given the typical matter losses described above, nutrients by themselves are hardly the pre-eminent and immediate problem, neither in north European or in any other waters. The problem is the complete over-demand and overload on the system qualities, far beyond any system-immanent control span, by an energy and matter flux that is too high and, in connection with it, the degradation of the biological processing structure for the control of transport- and metabolic processes. The problem is the unattended random energy influx, the accelerated water flow, the strongly oscillating groundwater levels, and, as a consequence, the biochemical oxidative erosion of soils (oxidation of ammonia to nitric acid, of sulphur to sulphuric acid, of carbon to carbonic acid, and the associated elution of cations from the soil), that also result in the matter losses. Moreover, the problem is made worse by the forced pumping of groundwater and the discharging of sewage into surface waters.

Regional analyses in north Germany have shown that the problem of acidification, of declining forests, and the depreciation of drinking water by increased nitrates in the groundwater have one single cause, namely the disordered un-phased kinetics of oxidative processes in the drying, aerated, and then by precipitation events, wetted soils. After a period of establishing heuristic models of the process distribution patterns in time and space, say for a catchment area, together with the measurement of the hydrological flow pattern, only relatively simple measurements of conductivity over time are needed to realise the state of the system. (Charge flow densities can then be calculated in energy terms, by means of the Nernst-Peters equation, to establish the energy balance in an ecosystem).

Heuristic regional models and environmental monitoring, on a water cycle and energy (flow-density) basis, provide a means of understanding system coherence, which cannot be achieved by integrating the individual process rates. These are basically non-linear processes, distributed in both space and time, rather than linear ones and cannot be integrated to achieve annual totals, etc. Only holistically trained ecologists, in the future, will be able to mediate the necessary transparency of local and time-relevant ecosystem functions, including man, and contribute to system solutions.

Redevelopment of the catchment

Increase in natural vegetation cover

To begin the redevelopment of catchments where the natural cycles have partially or completely broken down, measures must be taken in all areas of the catchment (Figure 7).

The upper parts of the catchment are the most sensitive to the leaching of soils by water run-off. Thus a major priority in redevelopment will be to ensure that upper catchment areas are covered by un-managed forest, with no net production, in order to secure matter cycling and prevent erosion. More wetlands are needed in headwaters to act as buffers and help compensate for uneven flow rates, (uneven distribution in time and space), and reduce the spontaneity (randomness) of processes. The riparian zones, especially at the confluence of two tributary sub-catchments, are the catching areas for matter losses. Here, especially, flooding areas are important to help stabilise the water cycle.

Riparian buffer zones, with interlaced vegetation structures along water margins, must be adjusted to the conditions relative to their expansion and management. Here, water can be retained in vegetation, and due to the diminished degradation processes, most matter loss can be prevented together with the retention of organic substances. These riparian wetland areas should also increase evaporation run-off ratios due to the higher evaporation. The nutrient and mineral substances accumulated in the biomass and in the soil would have to be recycled by intelligent management measures in accordance with demand. By maintaining optimum growth, the vegetation components hold their retention functions efficiently. The biomass accumulating by the management of the vegetation should serve as a substrate for the restitution of land lying further up the catchment. There, the strongly damaged soils could be recolonised by a low-matter-loss vegetation cover over the upper catchment.

Decrease in the toxicity of pollutants

The reticulated vegetation, with an incessant water cycle, helps solve the problem of a continuous increase in the toxicity of pollutants in the topsoil due to the loss of useful substances (mineral bases and nutrients), and prevents an intensive circulation of dusts in the atmosphere by binding them. The dewy vegetation cover, thanks to the increased condensation through short water cycles, binds the dry deposition, filters those substances which are useful for the vegetation from the dust and channels them into cycles.

Toxic substances that are not very soluble are 'diluted' by the production of organic substances and, thus disappear gradually from the rhizosphere by transfer into deeper, reducing zones, with almost no water transport, (and therefore a low reduction potential), and, as far as heavy metals are concerned, by sulphitic deposition. Roots and rhizomes, for example, have selective biological membranes which are probably the only low-energy means of detoxifying soil by dilution with a living biomass. Toxic substances are also fixed by means of a reduced transport possibility with the water in gley and podsol soils.

Reduced areas of agriculture production

To support present population levels in the catchment, food production must be managed intensively in smaller areas. The necessary increase in biomass and vegetation cover in the

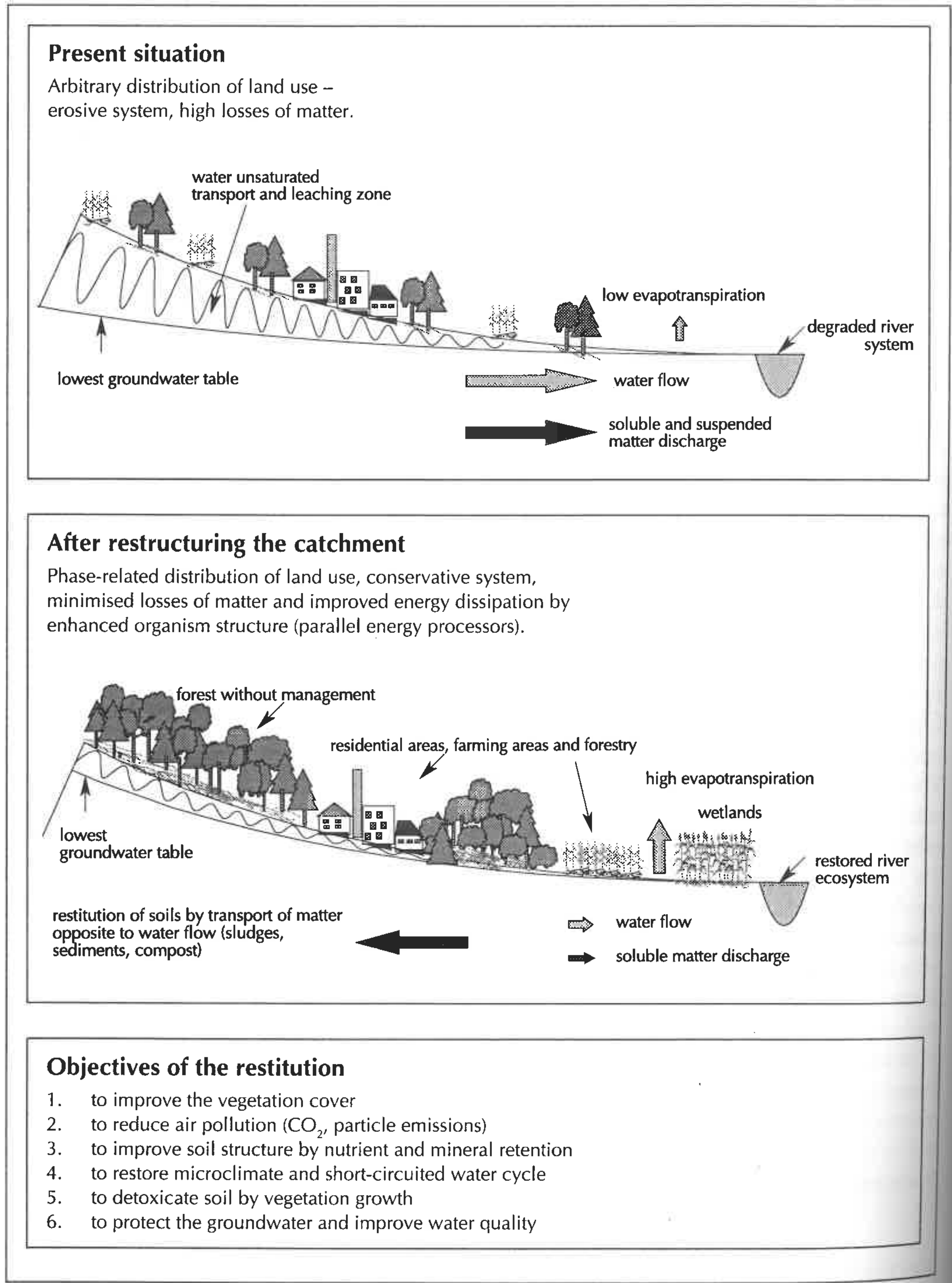


Figure 7. Restitution model of water cycle in the catchment to increase the sustainability of the ecosystem.

catchment area could not be achieved by a spatial expansion of agriculture, unless much of this agriculture was based on wetland-type crops with a substantial increase of soils with a higher water-preserving capacity. Reasonable animal-keeping on smaller areas, with intensive field management in areas more remote from waters, offers the possibility of recreating substantially larger areas with a natural vegetation cover and their proper water cycle.

Improved water and matter storage

Throughout the catchment in general, moist-soil layers should be increased, providing water storage for vegetation and increasing evaporation. As we have seen, cooling systems are crucially important and their distribution has consequences for both climate and the reduction of gradient potentials in the landscape. If cooling systems are regularly distributed over space and time, evaporation does not result in a loss of water but is part of circulation within the system. To keep matter within the system, the groundwater should be stationary. Drinking water supplies should only be taken from surface waters and never from groundwater systems.

Time-series analyses of hydrological processes combined with chemical analyses, in areas with intact vegetation cover, show that vegetation also supplies high quality water that is low in nutrients and minerals, as well as strongly-damped discharge ratios.

Possible solutions for the future

Such measures for the redevelopment of catchment areas for sustainable use, as described above, could support low population densities similar to those existing prior to the industrial age. However, this could not by itself support, in a sustainable way, the population densities found in Europe and elsewhere today. Man cannot turn the clock back to the period of stable climax coenosis and hope to support present-day population levels. Imaginative and intelligent thinking is needed for a sustainable future. Figure 8 shows diagrammatically some ideas as to how matter and water cycles can be closed, natural vegetation cover increased in upper catchment areas, and food production managed intensively in small areas.

Water and matter recycling in rural areas and cities

To close the matter cycles and to make it possible to manage the land without severe matter losses, in areas with low urban densities and their environments, the sewage and clear water flow should be separated by retention spaces similar to the arterial and venous transport system in organisms. The retention spaces consists of parallel or sequentially arranged wetlands and water areas. Such an arrangement is necessary for increasing retention of matter and efficiency of the process. The cleansing process mainly occurs by an intensified evapotranspiration and is followed by condensation and not like before, by a percolation through the soil.

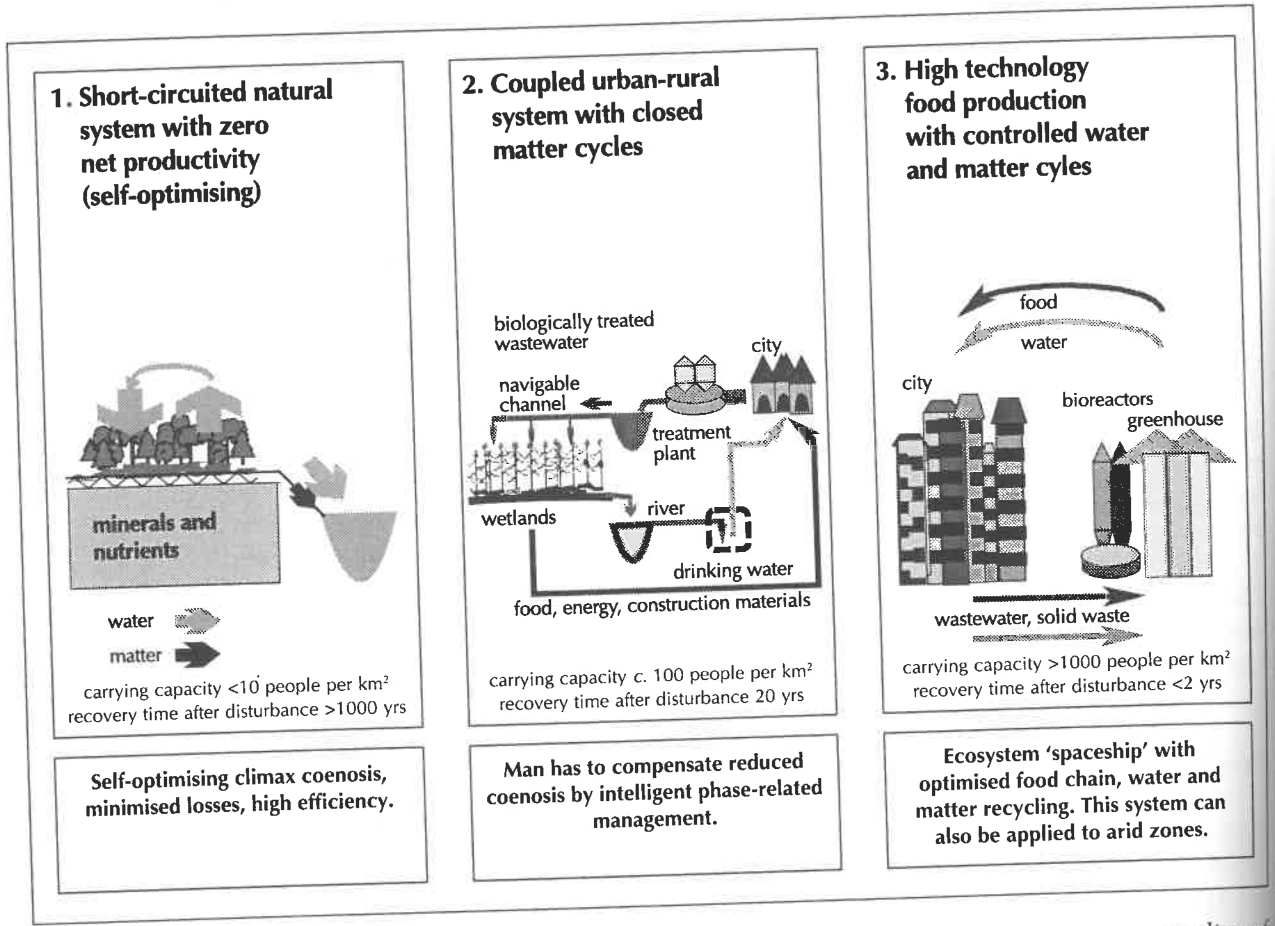


Figure 8. Proposed landscape systems, low in matter losses, to be integrated into sustainable and manageable ecosystems.

Cities show the strongest interventions in the water balance. Through extensive sealing of land they give less opportunity for establishing a short water cycle. Better cooling conditions in the city and a water balance with increased short cycles could be reached by partly de-sealing and the greening of sealed surfaces, facades and roofs of residential, industrial and business areas. So emissions will be minimised and the city climate will be improved.

Extensive creation, or restoration, of wetlands provides the necessary nitrogen sinks during run-off in drained farmlands (Stibe & Fleischer 1990; Fleischer 1990). Large areas of wetlands managed extensively for the production of raw materials and energy should be established, with the recycling of biologically well-treated effluents free from noxious substances into these areas. The use of intensive sites for food production in, e.g., vertical greenhouses with a high degree of recycling, all established in catchments with highly reduced groundwater flow, seems to be a way of stopping the impetuous losses. Vertical glasshouses run using solar energy with completely closed water and matter cycles and a minimal need for water could be integrated into cities for the production of vegetables. Such parallel greenhouse production structures are approaches for ensuring the food production even in arid areas of the third world.

Water farming

Water farming – the use of surface water – is the only sustainable use of water. Groundwater use should be replaced by the use of surface water. Farmers could be paid for the production of water quantity and quality – i.e. become managers responsible for wastewater treatment and drinking water production. This would raise the value of the countryside and possibly encourage populations to move away from city areas. In urban areas, wetlands are needed in order to recycle the waste.

Taxes on resources

One way to incorporate an ecological basis would be to change the rules of society in a way that the limitation of space and time is introduced, in the same way as time and space limits, and effectively structures, natural systems. This could, for example, be realised by linear energy or resource taxes, and a progressive land tax, instead of a tax on persons and work. Such a system requires, of course, a social basis (the right to a minimum subsistence and security), which could be financed from these taxes, instead of a social net. Waste could be converted into resources, and the use of energy could be reduced by optimisation of transport. The necessary step into a post-industrial society, which is limited in its spatial and temporal expansion, could be carried out.

Summary

To understand the causes and consequences of wetland degradation it is necessary to see the wetland as an integral part of the whole catchment area. Many processes have to be considered to understand how the deterioration of the catchment results directly in a non-sustainable wetland or lake. The holistic approach attempts to consider all the processes occurring in the landscape and how these are interrelated in both space and time.

The water cycle is the basis for the metabolic processes in nature. Water is the connecting, transport- and reaction- medium of the biosphere and produces its structure in space and time. The dissipation of energy by the water cycle is an energy-driven process where water and/or organisms are seen as processors which channel the energy dissipation in the system, while the energy (solar radiation) is offered on a daily and yearly frequency. Water, as such an energy processor, shows three dissipative processor properties acting in a recursive way, all three of which involve both cooling and heating respectively: the physical process of evaporation and condensation, the chemical process of dissolution and precipitation, and the biological process of production and respiration (water dissociation and association). Chemical dissolution processes are the main energy losses in ecosystems as matter transported by the water cycle. The irreversible matter losses from a considered system (e.g. catchment) determine the stability of the assemblage of organisms (energy processor structure) and form a selection frame for the coenosis. The efficiency of such a structure (coenosis), with the

coupled dissipative water cycle, is given by the amount of matter cycled in relation to matter lost (charge losses, reacted protons) for a given amount of energy delivered.

Condition and properties of ecosystems, catchments and natural water bodies, as well as their life expectancy, are determined by the non-regenerative losses of dissolved matter (mainly cation charges) from the catchment. In the last hundred years, irreversible charge flows via surface waters to the sea have multiplied, often of the order of a hundredfold. The total matter (salt) losses from agricultural areas are, in many areas, far more than one tonne per ha per year. The traditional water quality control parameters, phosphorus and nitrogen, together only account for 1% to 2% of this loss. Increasing losses of water soluble material in soil systems results in acidification and the concentration of immobile heavy metals and less soluble toxic organic residues, leaving impoverished toxic soils.

The structure of wetland ecosystems is determined by their function as efficient dissipative energy processors (water and organisms as an 'optimised water structure'), in time and space. Wetlands are biotopes where water is present in such an amount, that plants (biocoenosis) control all the processes by controlling the water transport, in a sustainable way. As stable, efficient systems within the landscape, wetlands serve to maintain short water cycles, decrease energy potentials by forming cooling condensation points and hence ameliorate the climate, maintain high groundwater levels and water-saturated soils, retain high nutrient and mineral loads, reduce high nutrient loads in surface waters, minimise oscillations and charge losses (base cations) from the topsoil, produce high biomass and extensive vegetation cover, and maintain water quality in surface and groundwaters.

Due to the deterioration of the short water cycle, climate changes become more sudden and severe, as less water is involved in evapotranspiration (cooling processes) and higher temperature gradients are randomly distributed throughout the catchment.

The recreation of vegetation and a water-saturated soil is a prerequisite for the minimisation of water and matter losses. In other words, the restoration of short water cycles is essential for sustainable development.

The long-term influence of man on the landscape has affected the water cycle and its coupling with energy flow and the transport of matter. Some solutions are briefly described for improvements in catchments and water bodies in the short-term, and more radical solutions to re-stabilise urban-rural integration, water cycles, food production and the balance of natural, terrestrial and aquatic ecosystems, for a sustainable future.

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4. Development of aquatic macrophytes in shallow lakes and ponds

Jan Pokorný

Zonation of macrophytes in oligotrophic lakes

In cold, temperate water bodies where the littoral topography, bottom conditions, exposure to wind and waves, light penetration in the water, grazing, etc., allow the development of a complete constellation of life forms of aquatic macrophytes, they are typically distributed in the littoral zone according to the schematic illustration in Figure 1. Based on the occurrence of different life forms, the littoral zone can be divided into a number of sub-zones. In Figure 1, the terminology of a zonation system useful for the ecological characterisation of littoral sections of Scandinavian lakes is included. The groups of macrophyte life forms include:

1. The *hyperhydates* (*helophytes*) – emergent plants such as the graminids, *Phragmites* and *Typha*, and the herbids, *Alisma* and *Cicuta*.
2. The *ephydates* – floating-leaved plants represented by spirodelids (*Spirodela*, *Lemna*) and nymphaeids like *Nymphaea* and *Potamogeton* with floating leaves.
3. The *hyphydates* – the group of submerged plants including riccids (*Riccia*), elodeids (i.e. plants with long shoots like *Elodea*, *Myriophyllum* and *Potamogeton* without floating leaves), isoetids (submerged plants with short shoots, exemplified by *Isoëtes* and *Littorella*), and muscids (submerged mosses).

The uppermost zone of the littoral, the eulittoral (Figure 1), is delimited by the highest level of the spring high water and the lowest level of the summer low water. The remaining division into sub-zones, as illustrated in Figure 1, is based on the vertical distribution of hyperhydates, nymphaeids, elodeids and isoetids. Thus the lakeward limit of the upper sublittoral coincides with the extension of hyperhydates (helophytes) and that of the lower sublittoral with the deepest occurrence of nymphaeids. In the same way, the upper elittoral ends with the deepest distribution of elodeids and the lower elittoral with that of isoetids and mosses. Below these littoral zones inhabited by macrophytes comes the zone called the profundal.

Depending on environmental conditions, the aforementioned types of plant life forms can be present or missing in lake ecosystems of different character and the deepest level to which macrophytes grow can vary tremendously. In a typical, South-Scandinavian, clear-water lake, macrophytes occur down to *c*. 5 m. In some Eifel maar lakes in Germany, *Nitella*

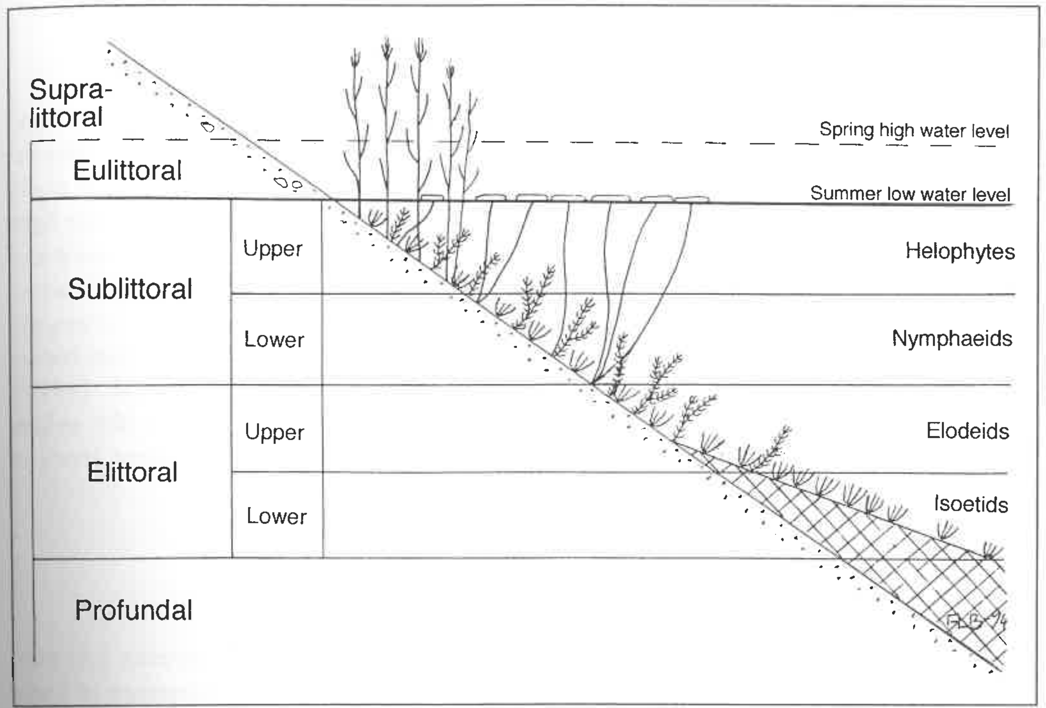

Figure 1. Schematic illustration of the littoral zonation of macrophytic vegetation in an oligotrophic lake.

flexilis grows at a depth of more than 20 m (Melzer 1992) and in a Bavarian lake, meadows of *Chara* are found at a depth of 15 m (Melzer 1976).

The most diverse vegetation is found in the upper littoral sub-zones, with a mixture of life forms. As the water depth increases, the hyperhydates, the nymphaeids, the elodeids and isoetids disappear one after the other, as the limits of their ecological demands become exceeded.

A complete spectrum of life forms of macrophytes, like that in Figure 1, is typically found in oligotrophic lakes with clear water (*Lobelia* lakes), allowing the development of dense carpets of isoetids in the deep-water lower elittoral. In brown-water (humic) lakes the levels of the sub-zones are displaced upwards.

With pollution and increased nutrient concentration, leading to turbidity caused by plankton as well as to overgrowth by periphyton, first the isoetids and then the elodeids disappear, i.e. the lakes lose their lower and upper elittoral zones as structural ingredients and functional sections of their ecosystems. At the same time, the quantity of emergent and floating-leaved plants typically increases.

Further increases in nutrient concentration result in ecosystems characterised by instability, which includes sudden changes and collapse situations. The development to this stage, and the interrelations among ecological factors in such systems, are exemplified by the heavily fertilised fish ponds of Bohemia.

Structure and function of fish ponds under heavy human impact

With an increase in nutrient input, stands of aquatic plants (macrophytes, chlorococcal and blue-green algae) become denser, their biomass per unit area increases, the vertical distribution of this biomass changes, and a shortage of carbon dioxide (CO_2) in the water occurs as pH increases during the diurnal cycle of plant photosynthetic activity. Competition for light and CO_2 are assumed to be one of the crucial processes determining the development and succession in *submersed vegetation habitats* in shallow ponds with increased eutrophication. Figure 2 shows graphically the vertical profiles of biomass distribution, light extinction, pH and dissolved oxygen concentration under different conditions of eutrophy or human impact (fish stock). This scheme is based on works of De Nie (1987), Pokorný *et al.* (1990), Pokorný & Ondok (1991), and does not include emergent wetland plants like sedges, common reed, etc. Below, the ecological conditions in fish ponds at different levels of eutrophy are described as schematically illustrated in Figure 2 (A – F):

A. Oligotrophic stage

In oligotrophic water bodies, growth of macrophytes is limited by lack of nutrients. Likewise, the lack of nutrients dissolved in water limits the growth of algae. Transparency of water is high and only the plants which are able to take nutrients through their roots from the sediment can grow. Plant biomass is accumulated near or on the lake bottom.

Values of pH and oxygen concentration do not change during day and night. Similarly, there are no differences in oxygen concentrations and pH values in the vertical profile, the values of oxygen concentrations being about air saturation level.

Such situations may be found in '*Lobelia* lakes' (*Littorella* sp., *Lobelia* sp. and *Isoëtes* sp.). These conditions were also common in some Bohemian fish ponds – before the fertilising of these ponds and the increased transport of nutrients from the catchment began.

B. Long-lasting oligotrophic to mesotrophic stage

In more fertile water bodies, the aquatic vegetation consists of a relatively high number of species. Water transparency is 2 m or more and the plant biomass is evenly distributed from water surface to the bottom. The tips of submersed macrophyte shoots only rarely reach the water surface, some of them developing floating leaves. Species diversity of the periphyton on macrophytes and of the benthos near their roots is also high.

Nutrient concentrations in the water remain low during the whole season and the development of planktonic algae is therefore limited. Shading and carbon dioxide uptake by the periphyton do not seriously limit the growth of submersed macrophytes as the periphyton is intensely grazed by molluscs, insect larvae and fish.

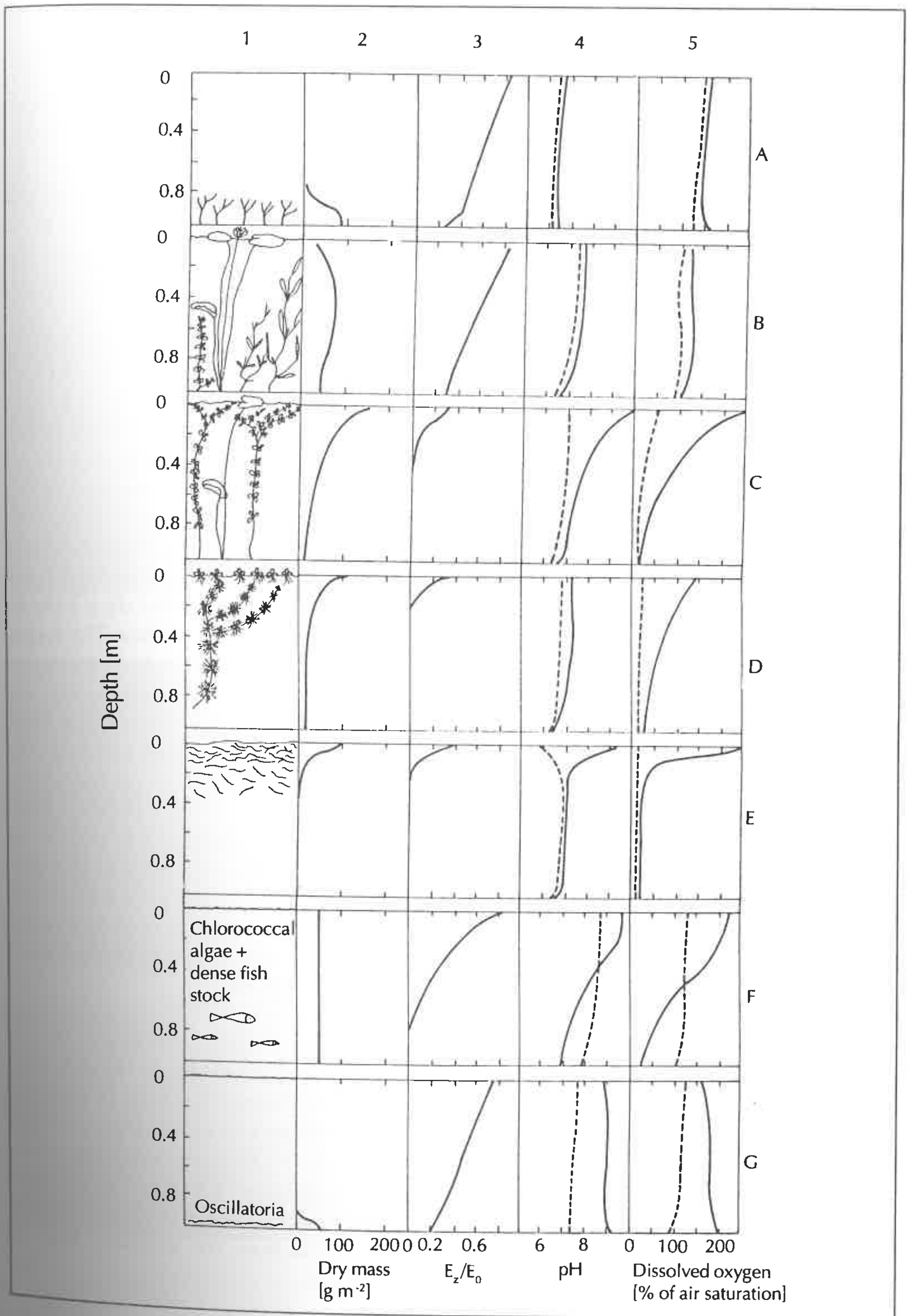

Figure 2. Development of aquatic vegetation (1) from oligotrophic (A) over mildly eutrophic (B), eutrophic (C,D) to hypertrophic (E,F,G) stages; (2) biomass distribution in vertical profile; (3) extinction profile, E_0 = PhAR irradiance at water surface, E_z = PhAR irradiance at the depth z; (4) pH and (5) oxygen concentrations in vertical profile (- - -) night minimum values, (——) day maximum values.

The concentration of oxygen in the water is about air saturation level and no marked differences are observed between the day and night or surface and bottom oxygen concentrations. The photosynthetic uptake of carbon dioxide does not have any strong effect on water pH – neither diurnal changes nor differences in the vertical profile are seen in pH.

At this stage, the aquatic primary production is essentially limited by a shortage of mineral nutrients. Nowadays, fish ponds of this kind are rare, not only because intensive fish pond management is more common, but also because of high levels of nutrient input from the catchment areas.

C. Initial stage of progressive eutrophication

Figure 3.
Stands of richly developed submersed macrophytes.

Higher nutrient input in the water bodies by, for example, manuring and direct fertiliser application (as in the case of fish pond management), and agricultural run-off, brings about a more vigorous growth of macrophytes and makes their stands become denser. Plants grow fast and their biomass accumulates at the water surface. The young, green parts of macrophytes shade the deeper water. Water transparency decreases and even in water bodies shallower than 1 m, irradiance at the bottom may not attain the light compensation point for photosynthesis (less than 3 Wm^{-2}). While photosynthesis prevails at the water surface, respiration prevails at the bottom – steep gradients of oxygen concentrations and pH values develop during daylight hours when no mixing occurs due to temperature differences between water surface and bottom.

As nutrient loading continues, species diversity of aquatic plants decreases but the stand biomass rises (Figure 3). This stage is also characterised by a mass development of periphyton which suppresses the growth of some macrophytes. The respiration rate of the whole community is higher than at previous stages and therefore a lack of oxygen often occurs near the bottom. The rapid decomposition of organic matter at the bottom results in low oxygen concentrations and the release of nutrients from the sediment. Internal loading of the water body starts to play an important role.

Figure 4.
Stands of submersed macrophytes invaded by filamentous algae.

Figure 5.
Cladophora fracta thrives in highly eutrophic fish ponds with low level of fish stock.

D, E, F. Eutrophic to hypertrophic stage

At high nutrient loading, the fish stock plays an important role in the development of water vegetation. At lower fish stock (seasonal mean live biomass of fish under c. 350 kg ha^{-1}), large *Daphnia* (filter-feeders of phytoplankton) are not completely consumed by fish. The high feeding pressure by *Daphnia* (cladocerans) prevents the growth of algae and water transparency is kept high, despite the high nutrient level. A combination of high transparency and high nutrient levels cannot last long and results in:

(a) the growth of macrophytes (D) in shallower waters. At high nutrient levels, macrophytes are very often invaded by filamentous algae (Figure 4). Filamentous algae, namely *Cladophora* sp., use inorganic carbon effectively even at low concentrations, grow fast and develop dense mats throughout the water column (Figure 5). When the filamentous algae reach the water surface (E), they completely shade the water column, pH increases to values above 11 (Eiseltová & Pokorný 1994) and those macrophytes which served as the substrate for filamentous algal growth die off.

(b) the development of *Aphanizomenon* blooms in deeper waters, as large colonies of *Aphanizomenon* cells are not consumed by filter-feeders so effectively and therefore remain, whereas smaller chlorococcal algae are consumed.

At higher fish stock levels (seasonal mean live biomass of fish above c. 400 kg ha^{-1}) large *Daphnia* numbers are reduced and chlorococcal algae develop dense populations of several hundred g chlorophyll *a* l^{-1}. These dense populations of chlorococcal algae (F) bring about low light transparency (Secchi disc transparency less than 0.4 m) which prevents the development of blue-green algae like *Aphanizomenon* and *Microcystis* later in the season when higher temperatures are suitable for their growth. However, later in the summer, other species of blue-greens may occur in the phytoplankton (*Limnothrix*, *Planktothrix*). Fish managers use the number of large *Daphnia* as an indicator of fish stock feeding pressure and are able to prevent water blooms of blue-greens at high nutrient levels by adjusting the fish stock. This stimulates the development of dense populations of small chlorococcal algae (Faina, pers. comm.). Such manipulation of the food web is possible in fish ponds where the fish stock is under control.

G. Development of *Oscillatoria* species

Spring development of filamentous blue-green algae, such as *Oscillatoria*, at the sediment surface results in a high pH in the whole water column and the subsequent release of free ammonia (NH_3) responsible for fish-kills.

Summary

To summarise, the development and succession of submersed vegetation indicates the level of trophic conditions in shallow lakes and ponds, as illustrated in Figure 2. If eutrophy increases, the biomass of submersed vegetation (submersed macrophytes, chlorococcal and blue-green algae) increases and the increased photosynthesis leads to extreme values of oxygen concentration and pH, and subsequent undesirable changes in water chemistry. This leads to unfavourable conditions for many aquatic organisms and degradation of ecosystem structure and functions. Considerable reduction of nutrient loads, both external and internal, is essential in order to re-create properly functioning ecosystems which are sustainable in the long term.

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5. Food web relations

Jaroslav Hrbáček

History and management of lakes and reservoirs

The end of the 19th century started a divergence in the development of limnology and the ecology of fishes. This limnological approach is best documented by the sentence: 'While various reciprocal relations exist between the plankton and the non-plankton animals, there is reason to believe that if all non-plankton animals were removed from a lake and kept out, the plankton, with possibly some minor modifications, would continue to exist' (Welch 1952, p. 277). The paper of Lindemann (1942), which is often considered as the start of the ecosystem approach, is based on the study of a fish-less bog lake. Ivlev (1939) has shown a very high efficiency of transfer of energy from primary production of phytoplankton to fish production. It is a pity that this paper was disregarded. From the high efficiency of the transfer of the energy in the food chain, an intensive pressure of predation, a 'top-down' effect, could have been predicted.

In the investigations of the ecology of fishes, it is usually tacitly assumed that the feeding activity of fish influences only the biomass or the quantity of their food organisms. The possibility that it changes the competitive interactions, and thus the species composition of its food spectrum, is usually not considered.

The term lake or reservoir management usually does not include fish management (Henderson-Sellers 1979, Mitsch & Jørgensen 1989), irrespective of the fact that the well-being of fish, especially predatory ones, is the best continuous indicator of adequate management of water quality for human use. On the other hand, a lot of experience exists with fish management in man-made water bodies. In countries such as the Czech Republic, the area of man-made stagnant water bodies is much larger than that of natural ones. That is the reason why Czech limnology has been concerned not so much with the classical objects of limnology, i.e. lakes, but with man-made water bodies, i.e. fish ponds and reservoirs.

Many fish ponds in Bohemia were established in the Middle Ages, some having a surface area larger than a hundred hectares. By the construction of ponds new water bodies with their biota were formed, or in today's terminology, new ecosystems were established and managed. Since the middle of the last century the management has increasingly intensified. As the result of this endeavour their productivity has exponentially increased (Hrbáček 1969a).

The first reservoirs, for water supply to mills, were also established in the Middle Ages, some of which were managed as fish ponds. Larger reservoirs for power generation and those for water supply are of more recent origin. The main features of their limnology are summarised by Hrbáček (1984).

The aim of this introduction is to indicate that the management of freshwater ecosystems including fish, has a long tradition in ecosystems where its goal cannot be restricted to the restoration of pristine natural conditions. Due to the necessity to manage man-made water bodies, a more comprehensive understanding of the interactions of all components of the biota were necessary.

In the last quarter of the 20th century, the efforts to decrease the undesirable development of algae and blue-green algae in lakes and reservoirs have been based on two concepts. One is to decrease the concentration of the limiting nutrients, the other is to increase the activity of filtrators and browsers. For the latter approach the term *biomanipulation* or more correctly *food web management* is used. The success of food web management depends both on the extent of practical experience and on the theoretical understanding of interactions within the ecosystem of lentic waters. In my opinion neither is fully satisfactory at present so it is impossible to present a concise set of instructions to safely achieve the goals of 'biomanipulation'. In the following paragraphs the results of investigations and the theoretical considerations of a group of Czech limnologists on these topics are presented.

Nutrients-phytoplankton interaction

The fundamental question which has to be addressed from the point of view of both theory and management is which compound or element is limiting for the development of phytoplankton. The term, development, includes both the qualitative (species composition) and quantitative (total biomass of algae in space and time) aspects. Preferably the frequency and biomass of individual algal species should be measured but that is extremely time consuming. Therefore, estimation of the chlorophyll *a* concentration is often used as a substitute for the determination of the biomass or biovolume of phytoplankton. The advantage is that chlorophyll concentration is simple to determine and it directly indicates phytoplankton activity. Its disadvantage is that it does not strictly correlate with biomass. This is because the percentage of chlorophyll *a* in the biomass of algae varies (within certain limits) in different species and under different conditions. All the available methods for its determination present only the static aspect. The dynamic aspect, namely how much organic matter is produced by the algae, is more difficult to assess.

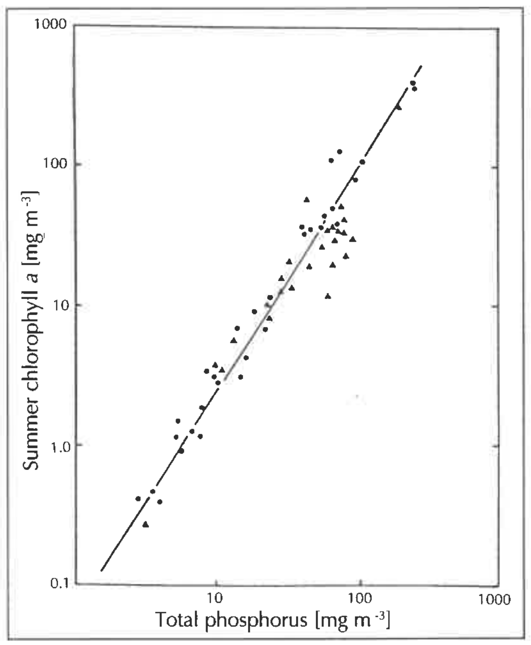


Figure 1. Relation of the seasonal average of chlorophyll *a* concentration to total phosphorus concentration in spring (from Dillon & Rigler 1974).

The course of events during the year fluctuates notoriously both within and between years. A cold May in one year can be followed by a warm one in the next. Seasonal averages are usually robust quantitative parameters due to the effects of autocorrelation (we have hot days only in summer) and compensation (a hot May can be followed by a cold June), more so than their coefficients of variation would suggest. The estimation of the seasonal average of chlorophyll *a* concentration was used to establish the relation between biomass of algae and blue-green algae on the one side and phosphorus on the other. Dillon and Rigler (1974) correlated the concentration of total phosphorus during the spring circulation and average concentration of chlorophyll *a* in the growing season (Figure 1). From the relatively good correlation of data from lakes which differed in total phosphorus by several orders of magnitude, it follows that total phosphorus is the main factor influencing the seasonal average value of chlorophyll in these lakes. If one looks in greater detail, the picture is less satisfactory, i.e., to the value of total phosphorus of 100 $\mu g\ l^{-1}$ correspond values of chlorophyll in the range 10 to 100 $\mu g\ l^{-1}$.

There are several reasons why the phosphorus-chlorophyll correlation is not closer than actually observed:

1. Hydrological, e.g. during the growing season there occurs a considerable inflow of water with a concentration of total phosphorus very different from that of the spring circulation. This is important especially in reservoirs with average retention times lower than half a year so that during the growing season a considerable water volume is replaced. In such cases correlation between the seasonal averages of both total phosphorus and chlorophyll *a* are usually used. One would expect that in this case, the variation should be smaller because the value of total phosphorus includes the phosphorus compounds present in the phytoplankton and therefore these two variables are not entirely independent. Investigations have, however, shown that this expectation is not fulfilled (Hoyer & Jones 1983, Hrbáček *et al.* 1993) and that there is about the same residual variability as in the classical relationship.

2. Physiological, e.g. some other nutrients limit, at least temporarily, the development of algae. Limitation by nitrogen was found in those cases when the N/P ratio was below 15, and limitation by CO_2 in cases where the pH was very high.

3. Methodological, e.g. the seasonal average value of chlorophyll is not comparable to the biomass of higher plants being related to the amount of nutrients present in the soil. Actual concentration of chlorophyll represents a dynamic equilibrium between the increase and decrease of algae. Increases are due to the growth and multiplication of algal cells which, in turn, is related to the actual concentration of nutrients. Decreases in the algae are due to consumption by filtrators or browsers, sedimentation and decay. Consumption of algae by filtrators is connected to the partial regeneration of nutrients due to catabolic processes in animals.

4. Statistical, e.g. sampling during the growing season is too infrequent such that extraordinary high values sampled are not compensated by present, but unsampled low values, and *vice versa*.

5. Inherent variability, e.g. several other factors not mentioned above influence the phosphorus-chlorophyll correlation. This is analogous to the variability in other relations as, for example, the length-weight relation of fish.

Relation of zooplankton and benthos to phytoplankton and fish stock

In the early days of limnology, it was tacitly assumed that the development of algae is due only to the concentration of nutrients. The development of filtratory zooplankton and benthos animals is dependent on the development of algae. The development of fish, such as roach, bream and carp, was assumed to be influenced by the development of its invertebrate prey. The nowadays shorthand term for this aspect of relations is *'bottom-up' control* (resource limitation). However, it is evident that the zooplankton can influence, by their feeding activity, the development of algae, and equally, fish can influence the biomass and species composition of zooplankton and benthos. This relational aspect is now known as *'top-down' control* (control by predators).

The selective effect of fish predation on zooplankton is much more pronounced than that on benthos. The reasons are intuitively assumed to be due to much greater protection of benthic organisms to fish predation, especially to that of most numerous small fish. Zoobenthos biomass per unit area surpasses that of zooplankton frequently during the season but its average turnover rate is lower than that of zooplankton. In fish ponds with an average depth of about one metre, it was estimated that the zooplankton and zoobenthos contribute almost equally to the carp production (Lellák 1957, Kořínek *et al.* 1987). It may be reasonably assumed that the relative contribution of zoobenthos to the fish production decreases as the average depth increases. To assess the quantitative dynamic aspects of the aquatic biota including growth rates is a very complicated and time consuming task. The role of all important food web links will be presented below in the section on the transfer of energy in the food web.

To ascertain the mutual relation of zooplankton to phytoplankton, the correlation of seasonal averages of zooplankton biomass to seasonal averages of the chlorophyll concentration was studied and considerable residual variation found. In backwaters and ponds, a relatively close relationship of the seasonal averages of zooplankton biomass to the concentration of nutrients was found. The composite samples of zooplankton were taken at night from different areas and different strata according to their proportion in the total volume of the water body. Even extreme differences in the fish stock did not affect the above relation in the sense that, as might be anticipated, a higher fish stock would lower the biomass of zooplankton (Hrbáček 1969b).

In water bodies without fish, large *Daphnia* species such as *D. pulicaria* prevailed in the zooplankton biomass. In water bodies with abundant fish (between 10 and 100 thousands per ha of fish of all year-classes – a frequent size of fish stock in oxbows and reservoirs), populations of small *Daphnia* species such as *D. cucullata* and dwarf strains of *D. galeata* and especially of *Bosmina longirostris* and *Ceriodaphnia pulchella* prevailed. In fish ponds with a very low fish stock (about 300 specimens per ha of carp in their third year) at the average concentration of total phosphorus of 0.3 mg l^{-1}, the spring values of chlorophyll decreased to a few µg l^{-1}. Later on in the season, blue-green algae developed water blooms, the biomass of which was highly variable from year to year and from pond to pond. In fish ponds with a fish stock twice or three times as high, higher concentrations of chlorophyll were found throughout the year (Fott *et al.* 1980). It was therefore expected that in reservoirs with a total phosphorus concentration lower by one order of magnitude, a low fish stock may induce a lower biomass of phytoplankton than is the actual one at a high fish stock.

The high fish stock in reservoirs is manifested not only by the presence of small species in zooplankton but also by the slow growth rate of fish which can be identified by the small distances between fish scale annuli.

Effect of different fish stock on the seasonal average of chlorophyll concentration in reservoirs

In reservoirs, it is difficult and time consuming to estimate the fish stock. Monitoring the development of the young-of-the-year (YOY) is especially difficult, yet, the production of this year-class can constitute a considerable part of the annual total fish production. The YOY can thus execute a considerable effect on zooplankton. In the previous section, it was mentioned that the feeding pressure of fish stock in shallow waters does not influence the biomass but the size distribution of the zooplankton. Therefore, the size of the prevailing zooplankton was expected to present a suitable measure of the impact of fish on zooplankton. As a parameter of the size distribution of zooplankton, the ratio of the biomass of large cladocerans (retained in a sieve with a mesh size of 0.72 mm) in the total cladoceran or zooplankton biomass, expressed as a percentage, was established. Cladocerans can be separated from other zooplankton by their selective adherence to surfaces. In this way, samples relatively free from detrital and water bloom particles can be obtained. This parameter has the advantage that it can be relatively easily determined, if desired, in less than 24 hours after sampling. It is not dependent on the method used for biomass determination (e.g. dry weight or colorimetrically as biurett reaction of proteins). It is also, within certain limits (central European climate and total phosphorus concentration within the range 20 – 100 µg l^{-1} as first approximation), independent of the level of primary production. Kubečka (1989) and Seďa (1989) have shown a good correlation of this parameter to the biomass and ration of the fish stock. Individual samples are, however, partly dependent on the nutritional status of the zooplankton. In the case of a high food supply, newborn of cladocerans are numerous and decrease the ratio. Conversely, poor feeding conditions act in the opposite way. During the growing season the feeding conditions of zooplankton are variable and so this effect is, within the seasonal average, negligible.

The seasonal average of chlorophyll concentration and the above ratio of zooplankton biomass was determined during several years in three reservoirs. In total, 26 seasonal observations were made (Figure 2). The range of the percentage of large cladocerans varied

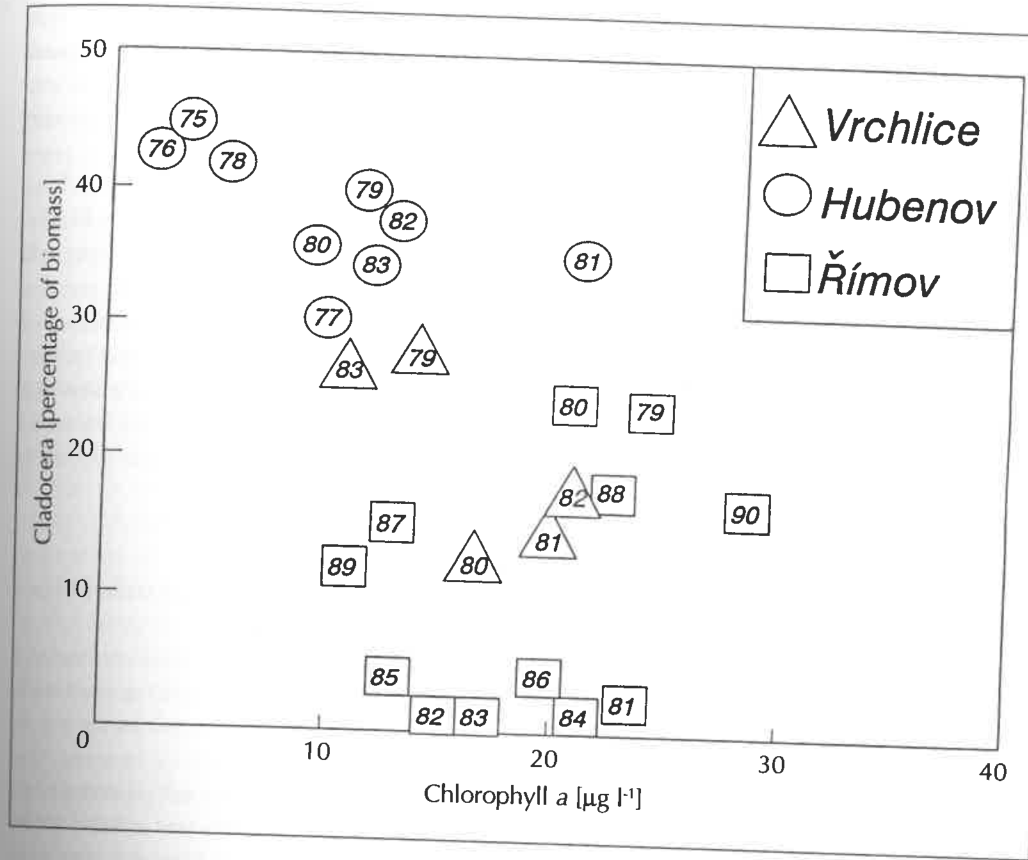


Figure 2. The relationship between the seasonal averages of the ratio of large *Daphnia* biomass to the total cladoceran biomass (%) and chlorophyll *a* concentration in three different reservoirs. The numbers within symbols represent the last two digits of the respective years.

between 1.5% to 45.3%, while the seasonal average of chlorophyll concentrations varied from 3.8 to 29.3 µg l^{-1}, i.e. again nearly within one order of magnitude. The range of the seasonal average of total phosphorus was from 12 to 38 µg l^{-1}, i.e. the maximal value is approximately three fold that of the minimum. From the relationship between the seasonal averages of chlorophyll concentration and percentage of large cladocerans, it follows that at percentages of large cladocerans above 40% (three seasonal observations), the seasonal average of chlorophyll concentrations was well below 10 µg l^{-1} whereas in all other cases it was above 10 µg l^{-1}. A general trend of an increase in the seasonal average chlorophyll concentration was manifested in the Hubenov reservoir whereas in Římov reservoir a decrease was observed (for details see the case studies, in Chapter 8). Total phosphorus concentration irregularly declined in both reservoirs (Hrbáček *et al.* 1993, Hrbáček *et al.* 1994).

When the seasonal average of chlorophyll concentration was linearly adjusted to the same concentration of total phosphorus, the general trend of the increase of the chlorophyll concentration with the decrease of the percentage of large cladocerans nearly disappeared. However, the exceptionally low concentration of chlorophyll in the three years with the percentage of large cladocerans above 40% remained (Hrbáček *et al.* 1993). These years also showed exceptionally high biomass of zooplankton in relation to chlorophyll concentration (Hrbáček *et al.* 1994).

The above discussion on the interaction of phytoplankton-zooplankton-foraging fish concerned only seasonal averages of concentrations of total phosphorus, chlorophyll, biomass of zooplankton per unit area and the ratio of large cladoceran biomass. The literature on this subject emphasises the seasonal course of differences at the species (or group of species) level. My own early observations used this approach but reproducibility of results was poor (Hrbáček *et al.* 1961). Early discussions (Brooks & Dodson 1965) up to the present (DeMello *et al.* 1992 and Carpenter & Kitchell 1992) concern mainly the interaction of fish-cladocerans-filtrable phytoplankton. On the other hand, cyclopoid and calanoid copepods and phytoplankton species which are not directly accessible to filtratory zooplankton present not a negligible part of total plankton and at high fish stock levels the average biomass of copepods can be larger than that of the cladocerans (Hrbáček *et al.* 1994). Organic matter produced by algae and blue-green algae not directly accessible to filtratory cladocerans can be, at least partly, directly utilised in the food chain by copepods and indirectly by bacterial activity in the so-called microbial loop (Stockner & Porter 1988). Thus organic matter is made accessible also to filtrators.

Transfer of energy from primary production to fish production

The approach presented up until now has been pragmatic, i.e. referring to observations and measurements which can be easily accomplished, even if they are not in full accord with the requirements of methods used in the study of productivity. A more rigorous approach is to measure, in addition to the static parameters such as concentration or biomass, the dynamic parameters such as uptake and regeneration of nutrients, growth rate, reproduction, mortality and production, and include them in a simulation model. There are, at least, three kinds of complications or drawbacks, two objective and one subjective, making this approach not as productive for the understanding of the interactions within the aquatic biota as it would appear at first sight:

1. One objective complication is that some factors (e.g. the effect of nutrient concentration on the growth rate of algae, or the effect of food particles on filter-feeding zooplankton) can only be related to unit volume. This increases the necessary number of estimations due to the reality that the activity of organisms is influenced by temperature and temperature is vertically, and within restricted limits also horizontally, stratified and some organisms undergo vertical and partly also horizontal migration. Still further, organisms are not distributed randomly even in the same layer but in more or less pronounced patches. This is especially true for benthic organisms. To get a real picture it would be necessary to take such a great number of measurements that their realisation is not within the possibilities even of a large research team. It is therefore necessary not only to clump species with similar roles in the food web together but also to include some assumptions, the reality of which is not easy to verify.

2. The necessary simplifications (mentioned above) have a drawback - that the description of the energy flow does not have a predictive power, e.g. to determine what kind of influences can change the participation of individual parts on this flow. In this respect, more

powerful is the concept of key stone species and the assessment of how the change in the development of their biomass or numbers changes biomass or species composition of adjacent trophic levels. In the community of standing waters, the fish stock has the role of key stone species.

3. The subjective difficulty is connected to the fact, that quantitative ecology of ecosystems is neither scientifically nor economically a very profitable endeavour for the researcher, as the probability that some principally new facts or phenomena will be found is very low.

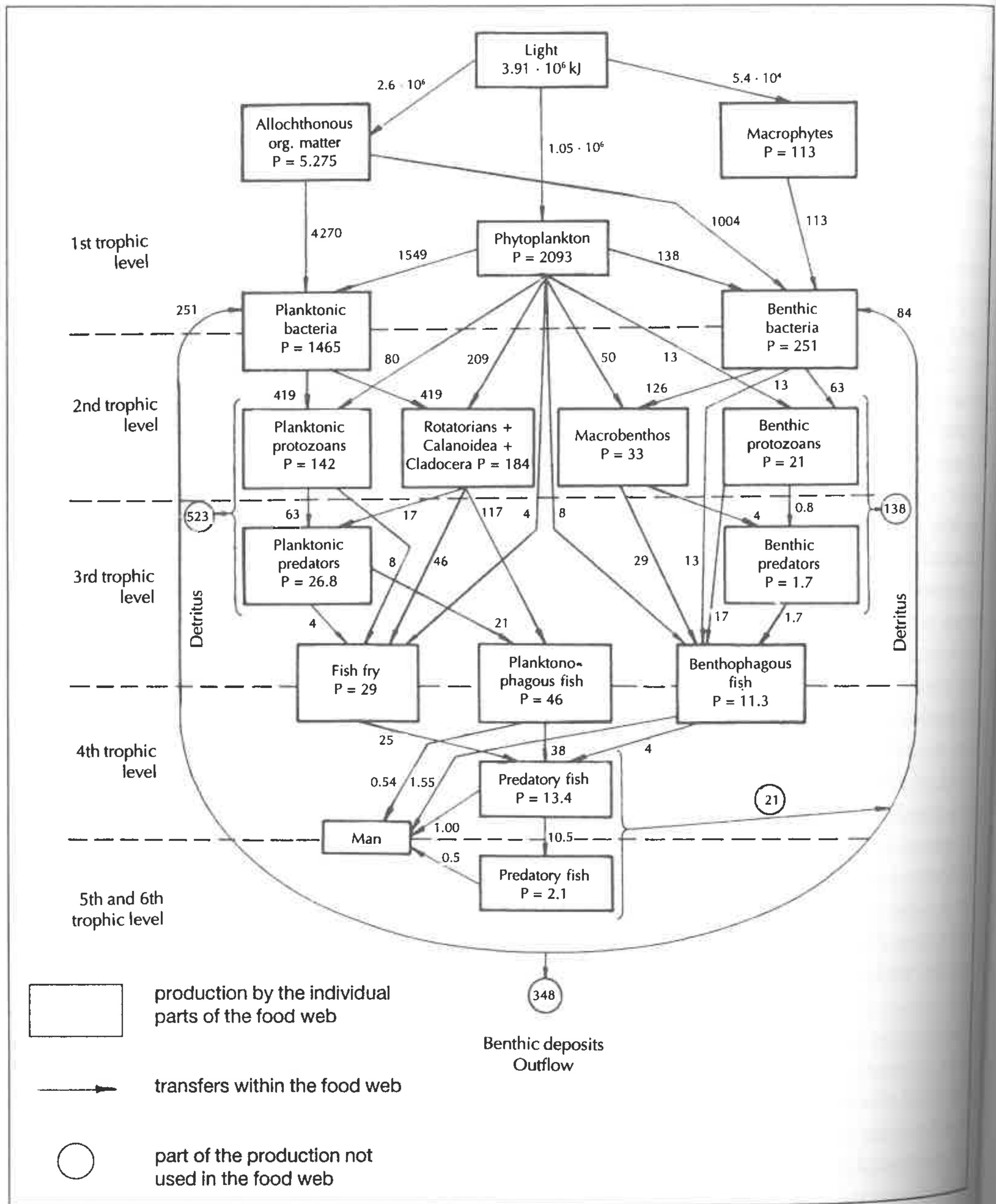
It is, of course, possible, at least partly, to fulfil the aims of the dynamic approach. Primary production of the planktonic algae can be measured in fertile water bodies as oxygen increase in light glass bottles suspended at several depths. Exposure for 24 hours could be considered as ideal, but it has several drawbacks. The most important of which is that the value obtained in this way is smaller than the sum of several shorter exposures. One other, is that in the flask are not only algae, but also bacteria and animals, such that their respiration decreases the concentration of oxygen, though it is possible to estimate the total respiration of the community by the measurement of oxygen decrease in dark bottles suspended close to the light bottle. If we add the value of the decrease in the dark bottle to the increase estimated in the light bottle, we get a value called the gross primary production. Due to the changes in frequencies and distribution of algae as well as varied illumination, this measurement has to be repeated every week, or at least every fortnight, to get a sum for the whole growing season.

In fish ponds, the yield of carp (the dominant fish species) after draining the pond is closely related to the total production of fish, better expressed as net fish production. It is therefore necessary to convert one of these parameters so that we compare either both net or both gross production. For converting the gross to net primary production it is necessary to subtract the respiration of algae from the values of the gross primary production. For this estimation we do not have a direct measurement at our disposal but have to rely on an educated guess. This is not an easy task as the algae present in different horizontal layers live in different light and temperature conditions. Pragmatically, a certain part, usually one third of the summary gross primary production, is subtracted. To convert the net to gross fish production it is necessary to add the part of the production used by metabolism (respiration). This would appear to be an easily measurable task. The difficulty is that, in addition to the effect of temperature, we have to take in account also the effect of activity. So we have again to rely, within certain limits, on an educated guess. This description indicates the limitations involved in the attempt to get a general picture of the flow of energy within a community.

The outcome of this comparison is that in fish ponds the efficiency of the transfer of energy from primary to fish production is about 5% (Kořínek *et al.* 1987). Assuming transfer by the food chain algae–filtrators–fish, this finding indicates an efficiency of about 20% from one level to the other. This is unexpectedly high as it is on the border of the assumed physiological possibilities of the organisms involved. Furthermore, bacterial production and that of invertebrate predators also depend on the primary production.

For the estimation of the fish production in lakes and reservoirs, it is necessary to estimate the numbers and the biomass of individual year-classes of individual species present. The method applied is to catch, estimate the numbers and size of individual year-classes (partly later on from samples of scales) of individual species, mark by fin-clipping and release. Later on, catch again. From the proportion of the marked fish in a later catch, the probable number of fishes in individual year-classes is estimated. From this description of the procedure

Figure 3. Schematic diagram of the quantitative trophic relationships within the Rybinsk Reservoir ecosystem. Data from Sorokin in Kuzin & Shtegman (1972) in kcal m^{-2} y^{-1} transformed to kJ m^{-2} y^{-1} using the factor 4.187 (Hrbáček 1984).



it is apparent that only fish above a certain size, i.e. at least two years old can survive the operation. Smaller fish can be estimated by other less elaborate procedures, e.g. by the estimation of their proportion to the larger fish in catches with very dense seines. When this procedure is repeated over several years, it is possible to estimate the decrease in numbers of individual year-classes from year to year, i.e. the mortality and the recruitment. From these data, it is possible to estimate the total fish production. However, this method is time consuming and hence the data collected are scarce.

In reservoirs, the estimated efficiency of primary production to fish production is only about a half of that of fish ponds (Hrbáček 1984). This lower efficiency can be due to at least three causes: (1) the ambiguous feeding character of some year-classes of fish, such as perch which can, in its third year, feed either on invertebrates or fish, thus adding an additional level in the food chain and lowering the efficiency of energy transfer; (2) a low growth rate of forage fish in reservoirs; and (3) the more 'diluted' conditions of total phosphorus concentration in reservoirs as opposed to fishponds by about tenfold. Comparatively diluted phosphorus concentrations in reservoirs can mean a greater expenditure of energy for fish in searching for and accumulating food.

In the future, when more data will be available, it will very likely be possible to predict the fish production from the concentration of total phosphorus in a similar way as it is now possible to predict the value of chlorophyll concentration from total phosphorus.

The chart (Figure 3) of estimated energy flow through all important parts of the food web of the large shallow Rybinsk reservoir (about 300 km north of Moscow, total area 4550 km^2, average depth 5.6 m, retention time 274 days) shows an efficiency of the transfer of primary production to fish production of approximately 4% which is close to that of fish ponds. On the other hand, the planktonic filtrators are, according to this chart, dependent more on the production of bacteria than on the production of planktonic algae. The prolongation of the food link should again manifest itself in a lower efficiency. This is counterbalanced by a very high input of allochtonous organic matter used up by the production of bacteria. The chart also shows that in this shallow water body the estimated production of comparable levels is, at least, about five times higher in the open water than on the bottom.

The growth rate and yield of the fish stock in ponds

The fish growth rate is dependent on the annual temperature pattern and the interaction between the fertility of the water body and the number of fish of individual year-classes per ha (fish stock).

Carp (*Cyprinus carpio*) became the preferred fish in pond fisheries in Central Europe. It is difficult to decide to what extent this selection was due to its taste, technologies used in pond management (especially to the procedure during the fish harvesting and transportation to the customer) or to its resistance to overcrowding due to its provenance from southern countries.

To reach the market size of about 1.5 kg, carp was grown, in the second half of the 19th century, for about six years, whereas now it is only three years (exceptionally two in warm areas and four in cold).

The length of the period from egg to maturity is regulated first of all by temperature (degree days), and to marketable size, also by the density of the fish stock. Usually only a single year-class of carp is grown in each fish pond. For simplicity, a linear inverse relationship between the size of the fish stock and individual growth is normally assumed. The desirable number of fish to be reared in a pond is calculated from the long-term average harvest of the fish which is related to pond fertility and the desired increase in size during the growing season (Schäperclaus 1961).

The above-mentioned linear relationship is, of course, applicable only within a restricted range. It is obvious that one fish per ha cannot fully utilise all the resources of the pond and grow to a size of several hundreds of kilograms which represents the usual harvest. It was found that at the stock which is close to the maximal growth rate of fish (μ_{max} in bacterial or algal cultures), the maximal yield of fish per unit area is not reached. It should be mentioned that in fish ponds, except in cases of mortality due to diseases or some negative environmental conditions such as low oxygen concentration or toxic substances, the yield represents net production of fish as the natural mortality is negligible. The maximal yield or production is reached at two to three times a higher density of fish stock than is the one at which maximal growth rate is reached. At a still higher density of fish stock the total yield decreases. This is obviously due to the necessity of fish to increase the intensity of searching for less accessible food. Usually it is assumed that this is due to the increased scarcity of food but in my opinion the assumption of lower accessibility is more adequate which includes not only frequency but also size (for a large fish small food items are of limited importance) and protection of the prey (e.g. protection of the larvae of chironomids by a layer of mud). In fish pond farming, natural fish recruitment is undesirable. From the above it is obvious why. It decreases not only the expected growth of individual fish but, in cases of intense recruitment, also the total yield.

Production and yield of fish in reservoirs

In lakes and reservoirs, the yield of fish does not depend only on fish production but also on the fishing effort (Bayley 1988). The fishing effort does not embrace evenly all year-classes of all fish species. Contrary to the situation in fish ponds, in these water bodies a part of the fish stock must remain over the winter to reproduce the next spring. Yield is in these water bodies much lower than fish production.

The important finding in these estimations of fish production is that the first two year-classes, which are usually not caught by anglers, constitute a very important part of the fish production (Figure 3, Mills & Forney 1983, Hrbáček 1984). The proportion of the older

year-classes, which constitute the main interest of anglers, is very low. It is obvious that it decreases as the growth rate diminishes, as a consequence of a high stocking level of fish. Thus the most effective way to improve the stock of game fish is not to increase their numbers by increasing recruitment or even artificial rearing and planting of fry but by the increase of growth rate. As in fish ponds it can be achieved only by reducing the stock. Similarly as in fish ponds, the size of the impact of fish on zooplankton can be estimated by the size of the species forming the bulk of the zooplankton, i.e. by the earlier mentioned percentage of the biomass of *Daphnia* retained on the 0.72 sieve to total cladoceran biomass. In this way, it is also possible to monitor the effect of measures to manage the fish stock. From Figure 2 and the section on the relation of zooplankton and benthos to phytoplankton, it follows that increasing the proportion of large *Daphnia* in the total cladoceran biomass will improve the water quality by reducing the development of algae.

The other possibility, to check for the effects of fish management, is the monitoring of the growth rate of fish, especially that of game fish, from their scales or other parts from which it is possible to identify the age of fish.

There are several possibilities to reduce the numbers or biomass of stock of forage fish:

1. To harvest the small fish by old well-established methods (different kinds of nets and traps).
2. Intensive planting of predatory fish preferably of older year-classes.
3. To decrease the recruitment of forage and coarse fish (for details see Chapter 7 and 8, Food web management).

To my knowledge, up until now in Europe, no attempt to reduce the fish stock was successful enough so that the growth rate of forage fish would considerably increase and large *Daphnia* species (preferably *D. pulicaria*) become a considerable component of the zooplankton biomass during the whole year. At present, *D. pulicaria* is very scarce in lakes and reservoirs in Europe except for high mountain lakes lacking in fish. In North America, on the other hand, there are lowland lakes in which *D. pulicaria* and other large *Daphnia* species are frequent. It may be speculated that this difference between Europe and North America is due to the different main stock of fish; Cyprinidae in Europe and Centrarchidae in North America. The lack of well-established comparisons shows, in my opinion, how little the interactions within the biota of lakes and reservoirs are studied and therefore how little understood.

Another, not fully understood, phenomenon, is the spectacular increase of the stock of predatory fish in reservoirs soon after they are put into operation (Hrbáček 1984) and the subsequent collapse within a few years. A water level rise in reservoirs simulates, in many respects, the situation during spring floods in the inundation areas of rivers.

The historical background to the formation of the biota of standing waters

The description of phenomena and relationships, and conclusions derived therefrom, is based on biota as they exist at the present time. This is, to a certain level, comparable to attempting to derive the meandering of the present shoreline of a reservoir from only the resistance of its different parts to erosion.

For a better understanding of the function of the biota in reservoirs it might be useful to get an idea of how the species composition of these biota came about. It was mentioned that these biota are relatively simple in the sense that the bulk of the biomass of individual levels of the food web is composed of only a few species. In zooplankton, this may be perhaps expected, due to the relatively uniform physical environment. This is probably not true, as in the ocean, in a single vertical haul of a plankton net, the number of species can be ten to hundred fold higher than in reservoirs or lakes.

From the geological point of view, not only fish ponds and reservoirs but also most of the European and American lakes are of recent origin. Zooplankton, contrary to the benthic fauna, do not have endemics, not even in the oldest lakes. Most of the species present in the plankton of reservoirs and lakes is also present in pools and oxbow lakes. There are a few exceptions, as, for example, some species (in Europe, *Daphnia hyalina*) are at present known only from lakes of glacial origin (Hrbáček 1987). *Bythotrephes longimanus* is also reported in the literature only from these lakes. Nauwerk (pers. comm.) has found this latter species, in northern Europe, also in pools. However, none of these species has been reported from reservoirs.

It is claimed that the low fish yield in reservoirs is due to the fact that the species of fish living there are derived from the stock of fish living in the riverine environment (Fernando & Holčík 1982). It is assumed that this fish stock is unable to fully exploit the plankton biomass in the central part of the water body. However, the range of the ratio of large cladoceran in total cladoceran biomass shown in Figure 2 does not support this assumption. The part of the statement concerning the origin of the species can be perhaps enlarged so that by the riverine ecosystem we mean not only habitats in the river itself but also in pools, backwaters and oxbows connected with the river during the flood. In one pool, in the course of one year, the concentration of nutrients can vary by one order of magnitude (Pechar, pers. comm.). In this way, we can perhaps understand why there is not a more expressed difference in the species present in plankton in habitats differing in total phosphorus concentration by three orders of magnitude (e.g. presence of *Daphnia pulicaria* in fish ponds and in the High Tatras mountain lakes above the tree line). Also the course of seasonal interaction between cladocerans and cyclopoid copepods can be understood more properly if we take into account that this relationship developed in pools (Hrbáček *et al.* 1994).

Summary

The requirements of water quality management of reservoirs and that of good fishing are not contradictory. Their improvement can be reached by the decrease of total phosphorus concentration and by the decrease of the stock of foraging fish that usually develops in reservoirs after these are put into operation. The decrease of the fish impact on zooplankton is manifested by the increase of the percentage of large cladocerans in the zooplankton. Also the increase of growth rate of forage fish points in the same direction.

At present, the effects of any measures to decrease the fish stock in reservoirs are not fully satisfactory. Partial success can be expected by the combined application of several measures:

- decreasing the recruitment of fish by the destruction of eggs and larvae;
- harvesting coarse fish by all feasible methods; and
- intensive cultivation and release of larger fry of predatory fish species.

The reason why the biota of the reservoirs are not self-regulatory, producing a growth rate of forage fish close to the maximum, under given temperature conditions, is not well understood. It is speculated that the structure of the interaction between invertebrates and forage fish, developed in conditions which differed considerably from those being found presently in reservoirs.

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6. Planning and accomplishment of redevelopment and restoration projects

Sven Björk

Project goals and limnological investigations

An essential prerequisite in the formulation of a restoration project is to set up restoration objectives for the site. Arguments for carrying through the project should be collected and the future status and use of the lake/wetland clearly determined.

Limnological investigations, over at least a one year period, must provide the basis for the technical design and execution of a restoration project for most types of degraded systems. The analytical part of these studies includes the calculation of a hydrological budget and the recording of the present ecological status, involving investigations of environmental conditions, communities of organisms and ecological interrelationships. Amongst other things, data on the chemistry must be collected in order to estimate input (external loading), availability in the lake (including internal loading) and output of nutrients. As regards the sediments, their horizontal distribution and thickness have to be mapped. The stratigraphy of the sediments should be examined in relation to soil type, and their physical and chemical conditions. It is important to elucidate the integrated role of the upper layer of the sediment in the function of the individual, degraded-lake ecosystem, as well as the conditions for the release and precipitation of essential elements. These studies should be carried out in conjunction with retrospective palaeolimnological investigations. If sediment has to be removed from a lake, it is advisable that the limnologist examines how the sediment behaves on drying and freezing, and asks questions about its utilisation for different purposes (for soil conditioning, in gardens, in agriculture, etc.).

The standard limnological pre-investigation programme should include an investigation of the plankton, macrophytic vegetation, bottom animals, fish and other vertebrates, ornithological conditions etc., with special interest regarding rare and threatened species. Data on productivity should also be obtained to make both quantitative and qualitative comparisons with the system once-restored.

Old photographs and maps as well as records in archives concerning fishery and hunting are often helpful to elucidate former conditions in the ecosystem.

On the basis of the analytical data, the limnological synthesis should explain the present ecological conditions, including the relationships between the catchment area and lake ecosystem, functional aspects and relationships within the system (trophic relations, production and destruction of organic matter, etc.) as well as the future development of the water body in the case that no restoration measures are taken.

The limno-technical plan

Limnological planning of restoration should describe in detail the measures necessary to achieve the agreed goals. As every ecosystem has its individual characteristics, the restoration plans have to be tailor-made for each specific project, and are most appropriately worked out by the limnologist, in cooperation with the staff of the firm responsible for the technical side of the project. Each plan should include detailed descriptions of the technical procedure, cost calculations and the monitoring programme needed to assess its effectiveness. Prognoses for the development of the lake after restoration should be included in the plan, and a follow-up programme to check environmental conditions in the restored lake should be designed. Both positive and negative results should be reported.

Based on experiences collected during planning and execution of a number of redevelopment and restoration undertakings, the procedure of planning, designing, accomplishment and follow-up of a project can be schematically summarised as follows. The specific considerations which have to be taken in case of acidified waters and ecosystems contaminated by poisonous substances are not included here.

I. Selection of ecosystem (examples)

1. Degraded lakes close to towns and cities, water reservoirs, fish ponds.
2. Degraded lakes/wetlands which used to be valuable waterfowl biotopes.
3. Redevelopment of wetlands for improvement of self-purification.
4. Landscape water management according to the type of large-scale redevelopment programme treated in Chapter 3.

II. Project goals

1. General improvement and re-creation of local, pleasant, environmental conditions. Aesthetic reasons.
2. Re-creation of waters for swimming, canoeing, wind-surfing etc.
3. Re-creation of waters and wetlands for waterfowl and aquatic biocoenosis in general.
4. Education. Before-during-and-after-studies. Access to aquatic biotopes for schools. Inspirational demonstration projects. Ecosystem structure and function as dependent on environmental changes.
5. Re-creation of a landscape with sustainable production and living conditions.

III. Project planning

1. Pre-project investigations. The holistic approach in space and time.

A. The long-term development of the ecosystem; palaeolimnological investigations. The ecological conditions immediately before degradation. The status from a regio-limnological point of view.

B. Changes in the catchment area. Present relationships between catchment area – lake/wetland. The external loading. What is the regio-limnological normal loading? Input and output of water and nutrients, calculation of budgets.

C. Internal ecosystem conditions. Investigations during at least one year.

(a) The bottom – sediment, peat: horizontal distribution, stratigraphy and functional role in the ecosystem (seasonal changes).
(b) The water – chemical and physical factors: nutrients, pH, transparency etc. Diurnal and seasonal patterns.
(c) Organism conditions: plankton, primary productivity, macrophytic vegetation, bottom fauna, vertebrates. Changes in evapotranspiration conditions of importance for water-level fluctuations.
(d) Sediment growth rate in lakes, deposition rate of coarse detritus in wetlands (accumulation basins).

2. Project design and implementation

A. The degraded ecosystem, its structure and function. Ecological diagnoses. Remedial eco-medical measures: 'strict diet, medication, surgery'.

B. Design of restoration method(s), calculations of cost.
(a) Measures to be taken in the catchment area.
(b) Measures to be taken in the lake/wetland ecosystem.

C. Information, argumentation, convincing. The public, politicians, administrators. The past and present situations compared with the future development of the ecosystem, with and without restoration measures. Co-operation with decision-makers and contractors.

D. Finalisation of financial commitment. Decision-making.

E. Accomplishment of eco-technical measures.
(a) Ecological and technical cooperation.
(b) Ecological control programme.

F. Measures for food web management, if considered realistic, to tune-up the system.

3. Post-project monitoring and project evaluation

A. Post-project investigations/monitoring during several years for control of sustainable results.
(a) External and internal nutrient loading.
(b) Collection of chemical and physical data, investigations on organism conditions.

B. Ecological syntheses of analytical data. Both foreseen and unforeseen positive and negative results should be reported. Before-and-after comparisons utilised for training and education of students.

7. Restoration methods and techniques

Internal nutrient control – Introduction

Sven Björk

Inland water ecosystems reflect the character – primarily the geology – of the catchment area, with surface and groundwater as carriers of solid and dissolved matter. Through rainwater and dry deposition, these systems also mirror conditions and activities transmitted from the air.

The *external loading* of nutrients is decisive for the original productivity in water bodies – within the limits set by light, temperature, precipitation, water renewal etc., for different areas at different latitudes. Properly-working lake ecosystems act as sinks for organic and minerogenic matter and this material in turn participates in adsorption reactions and chemical-complex formation. Nutrients are accumulated in the sediments and bound there. However, overloading by nutrients from external sources gives rise to an increased production of plant material in the lake and this in turn causes a spectrum of processes at the water-sediment interface. These processes may result in the leakage of nutrients accumulated in the sediment. This surplus supply of nutrients to the lake water constitutes the *internal loading*.

When the internal loading is added to the external one, a lake ecosystem undergoes sudden and dramatic changes in the form of a rapid increase in primary productivity. This leads to a fast degradation of the whole system – severe oxygen deficiency, increased sediment growth rate, fish-kills, etc. Such changes in the character of an ecosystem hit by internal loading, were first described and explained by Ohle (1955, 1965, 1971) and Thomas (1955, 1963), who designated the changes as being very sudden.

As already stressed, restoration methods and techniques aimed at a reduction or elimination of the internal loading in irreversibly damaged lakes should be applied after the external loading has been reduced to, or close to, normal. Phosphorus precipitation, aeration, sediment treatment and sediment removal are methods to counteract internal loading and to transform the ecosystem from a nutrient source to a nutrient sink.

If sustainable results are to be achieved, methods for the reduction and control of macrophytic vegetation have to be combined with methods for the adjustment of environmental conditions, as for example, by increasing water depth.

Efforts to manage the food web might be a method to tune up the function of an ecosystem after the nutrient level has been brought under control and the necessary environmental conditions for all stages of the life cycle of esteemed species have been realised.

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Phosphorus precipitation

Klaus-Dieter Wolter

Overview

High concentrations of phosphorus in the water column cause an increased primary production and undesirable algal blooms. Application of phosphorus-binding compounds to the water brings about the inactivation of phosphorus thus decreasing its availability for primary producers. The application of phosphorus precipitation should be restricted to those cases where, after redevelopment of the catchment, no considerable loading of phosphorus to the lake is left.

After the diversion of nutrient loading from a lake, some lakes continue to show prolonged high nutrient – especially phosphorus – concentrations. If this is observed in lakes with a rapid water exchange (retention time considerably less than 1 year), a permanent high phosphorus concentration indicates additional, yet unrecognised sources of external loading, or high phosphorus release from lake sediments – *internal loading*. Nutrient release from the sediment can be minimised by a suitable sediment treatment (*cf.* Sediment treatment, Chapter 7).

In lakes with long retention times (considerably more than 1 year), lake recovery can be accelerated by the precipitation of phosphorus out of the water column (in-lake treatment). For this purpose, *aluminium* (aluminium sulphate, $Al_2[SO_4]_3$) and *iron* compounds (iron-III-chloride, $FeCl_3$) can be used. They form relatively stable *phosphorus-binding compounds*, which sediment out in the form of a gelatinous floc.

As these preparations always have to be used in an over-stoichiometric amount, relative to the phosphorus in the water, an additional phosphorus-binding capacity is created at the sediment surface. For this reason, every in-lake precipitation may be viewed as a sediment treatment too. The principle conditions formulated in the chapter 'Sediment treatment' (p. 75) are also valid for phosphorus precipitation from the water column.

Beside aluminium and iron, some authors also recommend the use of *lime*. In this case over-saturated calcium oxide or hydroxide solutions are applied to the water. Calcium together with hydroxide and phosphate forms apatite, a phosphorus-containing mineral, but no successful technical treatments of lakes with this preparation are known:

$$10\ Ca^{2+} + 2\ OH^- + 6\ PO_4^{3-} \rightarrow Ca_{10}(PO_4)_6(OH)_2$$

By several authors, the use of *fly ash* from the combustion of coal or oil was proposed. Fly ash may bind phosphorus by its high specific surface and the content of CaO, MgO, and Al (III). Nevertheless, the use of fly ash is not recommended, because toxic effects by heavy metals or by drastically increasing or decreasing pH after the treatment were observed.

The methods of phosphorus precipitation from the water column are best suited for smaller, shallow lakes. The application of in-lake treatment should always be preceded by a significant reduction of nutrient loading from the catchment area in order to achieve effective and long-lasting results. Otherwise, the positive effects of treatment can be overwhelmed in a few months by continued external loading.

Effect of aluminium and iron compounds on pH of lake water

Aluminium sulphate and iron chloride have to be used with care because of their effect on the pH of the water. When applied to the water, the preparations undergo hydrolysis which liberates protons as described in the reactions below:

$$Al_2(SO_4)_3 + 6\ H_2O \rightarrow 2\ Al(OH)_3 + 3\ SO_4^{2-} + 6\ H^+$$

$$2\ FeCl_3 + 3\ H_2O \rightarrow Fe_2O_3 + 6\ H^+ + 6Cl^-$$

In water bodies with low alkalinity, the introduction of iron chloride or aluminium sulphate could use up buffer capacity, which would result in *acid conditions* in the lake. Although the coenotic structure in the lake will change, toxic effects on fishes and benthic organisms can hardly be observed, as long as pH does not fall below about 6.

In order to prevent acid conditions, it might be necessary to add calcium carbonate (in the form of fine particles with a high reactivity) together with the iron or aluminium preparations, to provide a sufficient amount of buffer in the water and at the sediment surface.

Precipitation with aluminium sulphate (alum)

In the case of aluminium, inorganic phosphate is bound directly or by adsorption to aluminium hydroxide:

$$Al(OH)_3 + PO_4^{3-} \rightarrow AlPO_4 + 3\ OH^-$$

$$Al(OH)_3 + PO_4^{3-} \rightarrow Al(OH)_3 \sim PO_4^{3-}\ (aq)$$

The best fraction to bind to aluminium is phosphate-phosphorus. Organic fractions of phosphorus are bound less. Dissolved organic phosphorus is less effectively removed than particulate phosphorus.

Total phosphorus in the form of particulate phosphorus (algae, detritus, bacteria) cannot be bound to aluminium hydroxide directly, but the particles may be enclosed into the forming aluminium flocs and subsequently settle to the sediment. For this reason, precipitation may also be carried out during the vegetation period. In this case bathing should be prohibited during the treatment. However, the trapping of particles into the aluminium flocs only occurs when high concentrations of alum (above 5 g Al^{3+} m^{-3}) are used, which may be restricted by the cost of the chemicals or the lack of buffering capacity (alkalinity) in the lake water.

The surplus aluminium settles to the sediment and increases its phosphorus-binding capacity. With time and with further sedimentation on top, this static layer of aluminium is buried deeper in the sediment. Although phosphorus binding to this layer can be observed in the laboratory for several years, in practice the burying of the aluminium leads to a quick cessation of the phosphorus binding to the aluminium particles. For this reason, long-lasting phosphorus trapping probably cannot be observed. Compared to iron the use of alum is cheaper.

Toxicity of alum

The use of aluminium sulphate was often coupled with toxic effects to organisms. Therefore, treatment with alum has to be carried out with care. Concentrations below 50 µg Al l^{-1} in the lake water during treatment will probably have no harmful effects on organisms. In low-buffered waters, the treatment with alum has led to the enrichment of aluminium in fishes. There are indications of bioaccumulation in rainbow trout (*Oncorhynchus mykis*) and chronic toxicity in Chironomidae (Cooke *et al.* 1986). In some cases aluminium is taken up by aquatic plants, thus reducing their physiological ability of phosphorus-uptake by the roots.

Laboratory experiments with aluminium and also with iron have often shown increased mortality of certain species. In lakes, these effects are usually lower, because the unhomogeneous distribution of iron and aluminium during treatment allows for escape reactions by the organisms.

Precipitation with iron chloride

Phosphorus precipitation with iron has been used in many cases. Similarly as with aluminium, the phosphate is bound in iron-phosphate minerals or it is adsorbed to iron oxide-hydroxide:

$$FeO(OH) + H_3PO_4 \rightarrow FePO_4 + 2\ H_2O$$

$$FeO(OH) + PO_4^{3-} \rightarrow FeO(OH) \sim PO_4^{3-}\ (aq)$$

From the phosphorus fractions, inorganic phosphate is best bound by iron. No significant bond of organic phosphorus has been observed. In contrast to aluminium, it is not possible to precipitate organic phosphorus-containing particles by their inclusion into the forming iron flocs. For iron, chronic toxic effects on organisms in the lake are not known.

After sedimentation, iron continues to bind phosphorus at the sediment surface. In contrast to aluminium, the bond of phosphorus to iron is redox sensitive. When the sediments become anoxic and hydrogen sulphide is formed, iron-bound phosphorus is released from the sediment and may be transported to the water column. To prevent this event, the treatment of sediment (Sediment treatment, Chapter 7) has to be in many cases combined with phosphorus precipitation.

However, the redox-sensitive behaviour of iron has also some positive aspects. As lakes and their sediments are dynamic structures, being altered by processes of production, sedimentation and respiration, the dissolution of iron may be viewed as a process enhancing the iron buffering capacity. Iron dissolved in deeper sediment layers migrates along its concentration gradient to the sediment surface. Generally, an enrichment of iron occurs at the reductive/oxidative boundary layer near the sediment surface, where, the exchange of phosphorus with the water and especially the binding of phosphorus from the water is a quantitatively significant process. Therefore, iron can be seen as a dynamic phosphorus trap, also functioning if the lake system shows some spatially or temporally restricted increase in phosphorus concentration.

Treatment schedule

First of all, an analysis of processes involved in phosphorus metabolism in the lake have to be made. Measuring concentrations of individual phosphorus fractions during one or several years would allow the identification of the most suitable period for treatment. The best time for phosphorus precipitation is when the phosphate fraction is at its relative maximum –

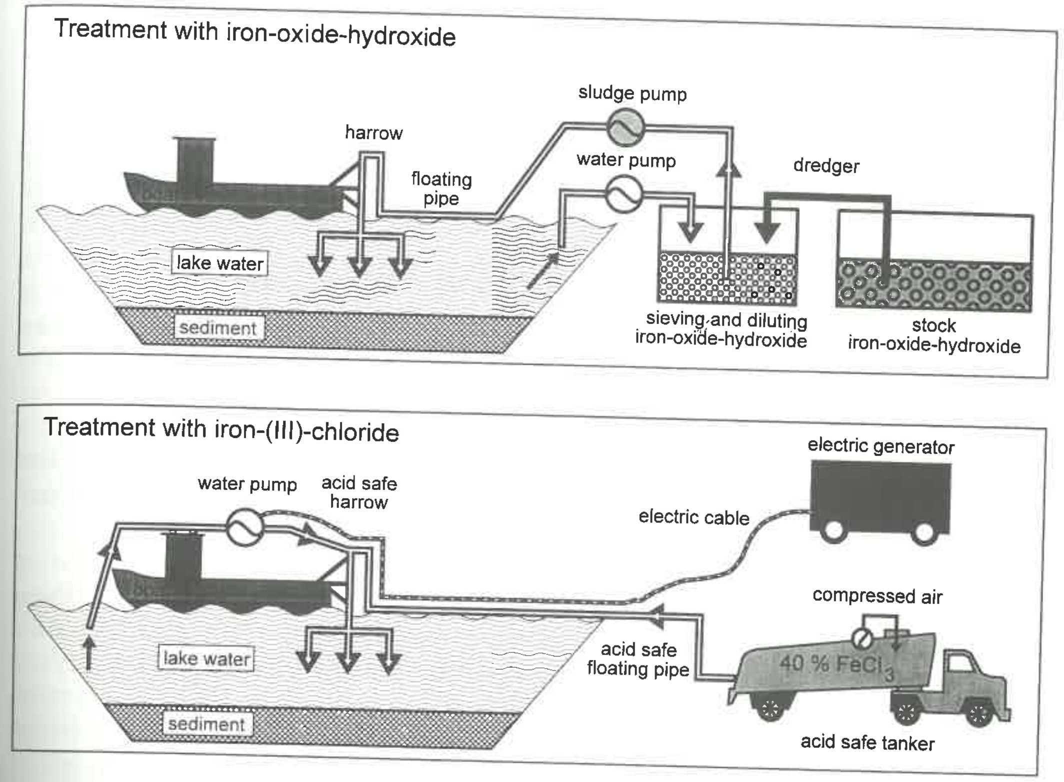

Figure 1. Schematic illustration of the application of iron oxide-hydroxide and iron chloride to the lake water.

Figure 2. A floating tube is used for the distribution of iron oxide-hydroxide and iron chloride which are applied by a harrow from the boat to the lake.

usually from late autumn to early spring. The treatment of the lake has to be finished before intensive plankton growth starts in spring.

The use of solid preparations is recommended only when these can be distributed over an ice area with high weight-bearing capacity. Fine solid particles of the acid-forming preparations have to be handled with care. Distribution from aeroplanes may allow fine particles to drift and cause damage to the lake surroundings. After applying the preparations on the ice, its structure changes, the ice weakens and should not be walked on. Most often, the preparations are applied in dissolved form. Figure 1 shows a schematic illustration of the application of iron oxide-hydroxide and iron chloride to the lake. The solutions are transported to the lake by flexible, high-pressure polythene tubings and applied by a harrow from the boat to the lake (Figure 2).

In the case that high amounts of dissolved organic substances in the water prevents flocculation of the iron or aluminium compounds, fine mineral particles (clay, bentonite) may be added to reduce the activity of these organic substances.

Chemical analyses of the water should be made during the treatment of the lake. These analyses include:

- iron or aluminium concentrations
- concentrations of individual phosphorus fractions
- chloride and/or sulphate concentrations
- pH
- alkalinity.

After the treatment, further monitoring and control of these parameters, together with parameters indicating the biological production of the lake, should be performed.

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Aeration

Bo Verner

Background

Oxygen deficiency caused by an increased supply or production of organic, degradable matter in the lake ecosystem, can be compensated through the addition of oxygen. The method chosen and the amount of oxygen that has to be added depends on how seriously and for how long the lake has been suffering from lack of oxygen. In the simplest case, the amount of oxygen to be added should be just enough to cover the extra need caused by an increase in productivity, i.e. production and degradation of organic matter should be in balance. In other words, naturally and artificially available oxygen should ensure that the degradation rate keeps pace with the production rate.

In addition to the normal spring and autumn circulation periods, shallow lakes and watercourses are subject to temporary circulations caused by the wind. By nature, however, these circulations appear irregularly, and between them, the water may rapidly develop thermal stratification. With this stratification formed at the interface between water and sediment, oxygen shortage or even anaerobic conditions may develop with the subsequent release of nutrients. During the following circulation period, these nutrients are brought up to the photic zone and contribute to an increased primary production. To prevent the stratification at the water-sediment interface and to secure favourable oxygen concentration of the water, the technique of *diffuse aeration/destratification (DA/D)* can be applied.

In deep lakes, the duration and intensity of the spring and autumn circulation periods vary widely between years and consequently so will the oxygen supply to the water. To prolong and complete the circulation, the DA/D method can be applied for one or more weeks. An installation for this purpose can be made fairly small, as the stability of the water is low which means that little energy is required to keep it circulating. During the stratification period, when the oxygen consumption in the sediment layer depletes the supply furnished during the circulation period, *hypolimnetic aeration* is recommended. The HYPOX method allows for aeration of the hypolimnion without disturbing the thermal stratification thus preserving the natural conditions of the ecosystem.

Diffuse aeration/destratification

Working principle

Compressed air is released through holes in a perforated pipe placed on or just above the lake bottom (Figure 1). The rising air bubbles generate a vertical flow of water towards the surface. Oxygen transfer takes place as the bubbles emerge from the holes, rise through the water and then burst at the surface shedding an oxygen-saturated film onto the surface layer.

Additional oxygenation from the atmosphere occurs due to the induced turbulence. The circulated mass of water is directly proportional to the cube root of the discharged air flow. The practical conclusion of this physical fact is that the more the discharge of compressed air installed can be spread over the bottom area, the higher becomes the oxygenation efficiency.

Design and installation

Perforated polyethylene pipes are placed on the lake bottom (Figure 1), spaced to cover the deeper part of the lake or distributed along the watercourse. They are anchored in a suitable manner depending on local conditions and the diameter of the hose being used. A special technique has been developed for drilling the air release holes directly into the plastic pipes to get the calculated outflow along the whole pipe system, which takes into consideration varying depth and pressure drops. The method excludes expensive nozzles or diffusers and there are no problems with corrosion or clogging. The aeration system is dimensioned in accordance with local requirements: the morphometry of the lake, the desired oxygen concentration, the flow and pressure of compressed air and the optimum balance between operating and investment costs. All installations are designed for each individual project in order to meet the customer's specifications, e.g.:

- in areas where maintenance dredging has to be done (or for other reasons), an integrated buoyancy pipe makes it easy to lift and move the air pipes aside;
- in canals, where very long pipes are used, starting up is facilitated by the installation of water evacuating valves at the far ends;
- to avoid stirring up the sediment, the perforated pipes can be placed at a certain distance above the loose sediment.

The air supply lines can be laid out from a raft and successively provided with an appropriate ballast. By running the compressor during the installation, water is prevented from entering the air supply lines or the perforated pipes which greatly facilitates the handling. The air line between the compressor and the water is often placed in a small covered trench. At the very edge of the water, special arrangements may be required to protect the air line from ice-movements during the winter and from possible interference caused by boating in the summer.

Compressed air supply and control

The air is supplied by a compressor plant installed close to the shore. The air pressure required is the total sum of the hydrostatic pressure at the deepest point of the aeration system and the pressure losses over the air supply lines, the perforated pipes and in the air release holes. The compressed air must be oil-free, supplied by a non-lubricated compressor or a standard compressor equipped with a separate good quality oil separator/filter. Modern compressors are normally silenced package units but special arrangements may have to be

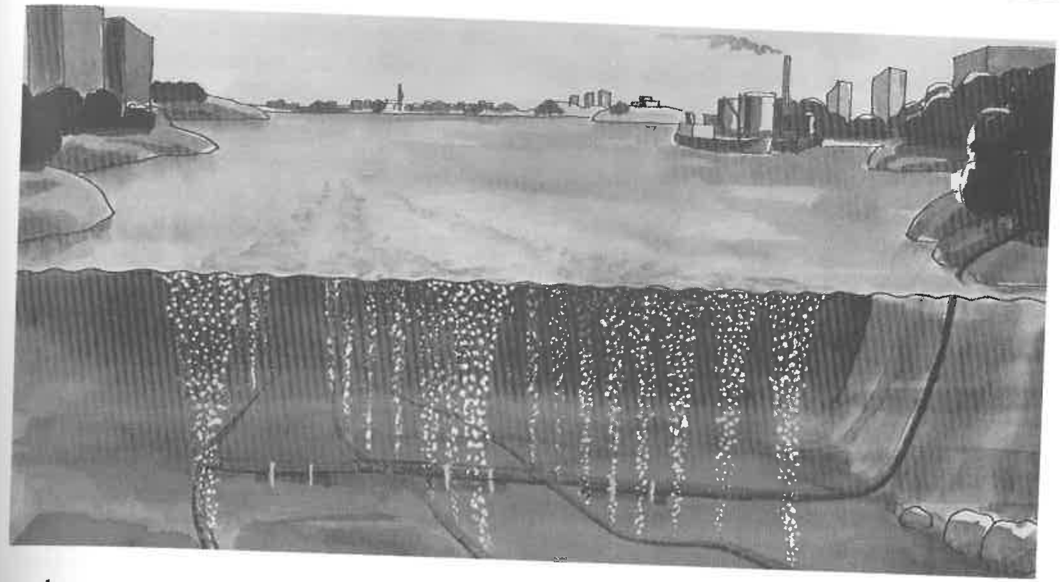

Figure 1. Diffuse aeration/destratification method, suitable for shallow lakes, to prevent thermal stratification at the water-sediment interface and to secure favourable oxygen concentration of the water.

made in view of the sensitive lake environment. A house is not required for package-type compressors, but a shed is recommended. Screw-type compressors are used for large systems and require a minimum of service.

Results

Aeration prevents the formation of a temperature gradient and subsequent oxygen concentration gradient at the water-sediment interface. Thus the temperature and oxygen concentration stay homogenous from surface to bottom. Aeration supplies the water with oxygen that supports decomposition processes keeping the sediment surface oxidised. This in turn reduces the recycling of nutrients, which lowers the organic production in the lake.

Hypolimnetic aeration

In a temperate climate, and especially in stratified lakes and reservoirs, the distribution of oxygen is dependent on the lake morphometry, in particular on the relation between the volume of the warmer upper water layer, the epilimnion, and the cooler bottom water layer, the hypolimnion. By studying the concentration and distribution of dissolved oxygen in the lake, a fairly good picture of the lake's general health can be obtained. The amount of oxygen consumed in the hypolimnion during a stagnation period – measured as the oxygen deficit – provides an indirect estimate of the ecosystem's productivity. Eutrophic lakes produce an excess of organic matter due to a high nutrient supply. The remnants from the metabolism accumulate at the bottom of the lake. The decomposition of this matter requires more oxygen than the ecosystem may be able to provide. If the oxygen in the hypolimnion is completely consumed, the condition becomes critical. Fermentation processes transform both organic and inorganic matter. Methane and hydrogen sulphide are produced. Nutrients, such as phosphorus and nitrogen, are released and dissolved in the water and then distributed during the circulation period. This increases the nutrient concentrations and the productivity escalates even more.

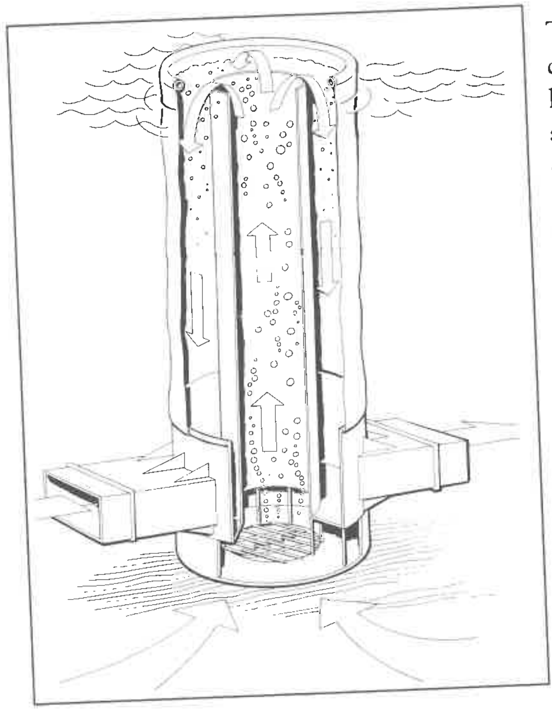

Figure 2. HYPOX aerator, developed to supply oxygen to the hypolimnion without disturbing the thermal stratification.

The HYPOX aerator (Figure 2) has been developed to supply oxygen to the hypolimnion without disturbing the thermal stratification. In this way, a high oxygen concentration is maintained throughout the stagnation periods and the release of nutrients from the sediment is minimised. It is very important to make a careful limnological investigation and state a definite diagnosis of the lake's condition before any HYPOX installation can be made. Only then can the adequate number, size and location of the aerators be determined.

Design and working principle

The HYPOX aerator consists of two concentric tubes with lengths equal to the installation depth. The outer tube has a number of outlets close to the lower end. The unit is permanently anchored to the bottom by means of concrete weights. A compressor on the shoreline supplies the aerator with compressed air via a hose placed on the lake bottom. Through the diffuser placed under the inner tube, the airflow is disintegrated into fine air bubbles. As the bubbles rise through the inner tube, an upward water flow is generated (the airlift pump principle). During the intense contact between the air bubbles and the water, and when the water flow meets the atmosphere, oxygen is transferred to the water. When the water spreads over the rim of the inner tube, the flow velocity is reduced. The water flow then turns downward through the space between the tubes and leaves the unit as a number of horizontal jets through the outlets and spreads over the sediment. The oxygen-poor water from the deepest portion of the hypolimnion is drawn towards and through the aerator, becoming oxygenated, and then spread back into the hypolimnion without any temperature increase of the water. In this way, the oxygen is supplied for the required decomposition and mineralisation of organic sediment.

Table 1. Dimensions and oxygenation capacity of the HYPOX aerators.

Oxygenation capacity	50 – 2,000 kg day^{-1}
Air consumption	5 – 150 l s^{-1}
Diameter	1 – 10 m
Height	5 – 25 m

All HYPOX aerators are designed and manufactured to the customer's specification to suit a particular project. Table 1 gives a general idea of the possible capacity range and dimensions. Larger units are designed on request.

Oxygen transfer efficiency

Oxygen transfer efficiency is the ratio between the oxygen supplied with the air-flow delivered by the compressor to the oxygen in the water. It is possible to obtain a high oxygen transfer efficiency of an aerator by having long contact times between air and water resulting in a low flow velocity. However, for low flow velocities, the oxygenation capacities, for example, decrease. For a given capacity demanded, the number and/or size of the units installed then has to be increased, thus increasing the investment cost. So, although a high transfer efficiency results in a low running cost, it is, of course, the total cost that is important. Main factors contributing to oxygenation efficiency are the following:

- solubility of oxygen increases considerably in cold water. The HYPOX aerator runs in the cold hypolimnion and benefits from its low temperature;
- the oxygen transfer efficiency depends on the oxygen concentration of the intake water. The more the intake water is depleted of oxygen, the better the transfer efficiency.
- likewise, the lower the oxygen concentration of the aerated water required, the higher the efficiency;
- some fractions, representing the BOD (biological oxygen demand) and the COD (chemical oxygen demand) portions, may be instantaneously oxidised already in the aerator unit. This fact must be included when calculating the efficiency; and
- the transfer efficiency also depends on the specific compressor efficiency. This, in turn, depends on the air pressure needed and the compressor design.

Operation

The HYPOX plant is normally started after the spring circulation period to counterbalance the increasing oxygen demand in the hypolimnion. The HYPOX units are commonly run throughout the summer stagnation period with only a short stop around the autumn circulation period, thereafter restarting for the winter if required. The design of the system makes it possible to run during the winter without disturbing the ice cover. To maintain a specific oxygen concentration, the oxygenation capacity of the HYPOX units can be matched against the varying oxygen consumption by reducing the airflow to the units or running a reduced number of units.

Results

The results of hypolimnetic aeration vary with the characteristics of the ecosystem – be it the eutrophicated lake or the iron-manganese rich drinking-water reservoir. However, some common typical results from HYPOX aeration are summarised below and in Figure 3.

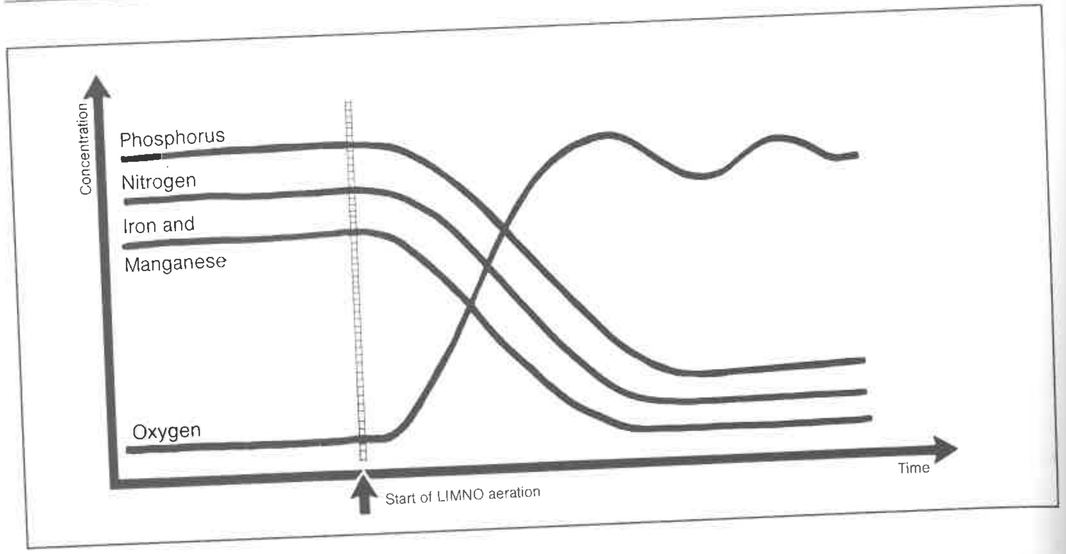

Figure 3. Common results from HYPOX aeration.

Oxygen
When the HYPOX plant is started in a completely oxygen-depleted hypolimnion, the full installed air capacity is supplied to all the HYPOX units. A rapid increase in the oxygen concentration is achieved. When the desired oxygen concentration is reached through aeration during the stagnation period, the system can be adjusted to run with reduced capacity – just enough to maintain this concentration.

Phosphorus
With the increased oxygen concentration, there is normally an immediate and steep drop in the phosphorus concentration from say 0.8 to 0.05 mg l^{-1}. At the same time the concentration of iron is rapidly reduced. This initial reduction is supposed to be caused through precipitation of ferric iron hydroxide with adsorbed phosphate. A slower decrease in the phosphate concentration following the first rapid one is normally dependent on its adsorption to the successfully oxidised sediment surface.

Nitrogen
The aeration also causes a drop in the inorganic nitrogen concentration usually from more than 2 to less than 0.3 mg l^{-1}. For example, if the nitrogen occurs mainly as ammonium ions in the hypolimnion before aeration, the aeration typically brings about a rapid reduction in the concentration of ammonium and a synchronous increase in nitrate.

Iron and manganese
The drop in the iron concentration has already been mentioned in connection with the phosphate reduction. The concentration of ionic iron is exceedingly low in aerated waters, most iron occurs as ferric hydroxide in particulate form. The solubility of manganese is considerably higher than that of iron, however, their chemical reactivity in freshwater ecosystems are similar. In drinking-water reservoirs where aeration has been applied, the concentration of both the metals have been almost completely suppressed, thus improving the drinking-water quality and reducing the preparation cost.

Transparency and chlorophyll
The normal variations of these parameters are large due to, e.g., difference in meteorological and hydrological factors and irregular diffuse leakage of nutrients into the epilimnion. Consequently, the effect of aeration on transparency and chlorophyll has to be studied over a longer time period as this effect is not as immediate as it is on the elements above.

Sediment treatment

Wilhelm Ripl

Introduction

Eutrophication problems in lakes are, generally, caused by an increased flow of nutrients and/or degradable organic matter to the lake basin. If such an excessive inflow has continued over a long period of time, the metabolism in the lake will change and the sediment's structure and function will alter. This change concerns an increased primary production which is often not adequately compensated by decomposition processes due to an insufficient oxygen supply. This means that both nutrients and organic matter are accumulated at the sediment surface, poising sediments at low redox conditions and in a state of high potential reactivity even if the external loading is reduced. This is the reason for the sudden change in lake metabolism after prolonged eutrophication, and at the same time the reason for a lake to be buffered against any effects of reduced loading. This behaviour could be called a hysteresis or delay in reaction after changed external conditions.

If a quick improvement of a lake's condition is the objective of restorative measures, this delay in improvement can be met by sediment treatment *in situ*. It has to be stressed, however, that internal lake measures have to be necessarily augmented by unloading measures in the catchment. Otherwise, any short periods of improvement obtained will be limited by the metabolism caused by external nutrient or energy (organic matter) loading.

Internal restoration measures and processes involved

Mineralisation of organic matter

Usually, the first problem to tackle is to increase mineralisation processes in step with production by supplying oxygen to the top sediment layer. The supply of oxygen is necessarily accompanied by water movement, thus increasing reactivity in a transport-limited sediment surface. Both the introduction of pure oxygen as well as the addition of very strong oxidising agents such as, for example, potassium permanganate and peroxides have been tried. However, these substances proved to have a very low oxidation efficiency caused by the loss of gaseous oxygen and partly also by the sterilisation of the sediment surface. In contrast, aeration measures using compressed air have shown good efficiency in some cases, especially in deeper, drinking water reservoirs having small areas of sediment to be treated and having dissolution problems of iron and manganese. High concentrations of iron and manganese increase the treatment costs for the preparation of drinking water.

In most eutrophicated lakes, however, the majority of sediments have accumulated over long time periods and much smaller layers have been deposited during periods of eutrophication. This is despite the fact that deposition rates during recent periods of eutrophication can be at least ten times or more that of earlier sedimentation rates. In these cases, a combined treatment, increasing phosphorus-binding on the one hand and oxidation of easily degradable organic matter on the other, can be a solution.

Combined sediment treatment with nitrate and iron

Table 1. Lake Lillesjön (southern Sweden, treatment accomplished in 1975).

Catchment area	1.0 km^2
Lake area	4.2 ha
Mean depth	2.0 m
Maximum depth	4.2 m
Treated sediment area	1.2 ha
Chemicals used	13 t $FeCl_3$ 5 t $Ca(OH)_2$ 12 t $Ca(NO_3)_2$
Methods of application	direct injection by sediment harrow
Treatment period	3–4 weeks

Further measures for redevelopment
- stoppage of sewage inlet
- removal of vegetation

Table 2. Hambuttenpfuhl and Karutschenpfuhl (small ponds within Berlin, treatment accomplished in 1991).

Catchment area	2.4 ha; 23.3 ha
Lake area	0.28 ha; 0.29 ha
Mean depth	0.8 m; 0.9 m
Maximum depth	1.9 m; 3.7 m
Treated sediment area	about a lake area
Chemicals used	4 t $Ca(NO_3)_2$
Method of application	entry into the lake water
Treatment period	several days

Further measures
- iron treatment of the sediments (500 g Fe m^{-2})
- aeration by circulating plant
- clearing of shore trees to minimise foliage (leaf litter)

The combined measures of iron treatment and oxidation by biochemical denitrification processes were applied, for the first time, in Lake Lillesjön (southern Sweden). The experience showed that a single treatment can change lake metabolism instantly and provide water of bathing quality for a long period. Lake Lillesjön was treated in 1975 and its metabolism has not significantly changed since. Some data for the treatment of Lake Lillesjön and of Hambutten and Karutschen ponds (Berlin) are summarised in Tables 1 and 2.

The reactions involved in this type of lake restoration, by sediment stabilisation, are processes which occur naturally in lakes in the top sediment layers. In lakes with relatively low nutrient loading, the phosphorus from the catchment is quickly deposited by precipitation with iron or lime compounds, introduced into the lake from groundwater contributions. Such an immediate precipitation, during periods of high flow in winter and spring, makes little phosphorus available for maintaining the pelagic biological production activities in the pelagic compartment of the lake. The small planktonic production in such lakes is mineralised in the pelagic zone by a sufficient oxygen supply during the water circulation twice a year. The retention function for phosphorus in the sediment is very high because of a greater than stoichiometric content of iron, or other phosphorus-precipitating agents, in relation to the phosphorus inflow.

An increase of production in the lake, by nutrient contributions or an input of organic matter from the catchment, leads to the accumulation of degradable organic matter at the sediment surface layer. This enrichment of the substratum is followed by increased bacterial activities and increased uptake of oxygen, causing extended periods of anoxic conditions at the sediment surface. This spontaneously reduces the oxidation potential, and other electron acceptors, rather than oxygen, are used for mineralisation (denitrification, desulphurication, methane production).

In this reaction scheme, denitrification is the first process. This provides terminal oxidation of organic matter by the production of carbon dioxide and the liberation of molecular nitrogen, described by the following chemical equation:

$$4\,NO_3^- + 5\,CH_2O \rightarrow 5\,CO_2 + 2\,N_2 + 3\,H_2O + 4\,OH^-$$

This process is still in a redox range where, iron is not reduced to bivalent ferrous ions, or where, hydrogen sulphide competes for ferrous ions with the phosphorus ions. This means that phosphorus is still not dissolved from the sediments and does not lead to enhanced primary production in the pelagic waters.

Although natural, denitrification is not a very significant process in natural lakes, because nitrogen in the oxidised stage is seldom provided in sufficient amounts by tributaries. This is especially true during the periods of stagnation when oxidised nitrogen compounds quickly disappear by this process.

After the disappearance of nitrate, desulphurication, described in the equation below, takes place in the sediments. The process of desulphurication oxidises organic matter and produces CO_2, thereby reducing sulphate:

$$2\,CH_2O + SO_4^{2-} + 2\,H^+ \rightarrow H_2S + 2\,CO_2 + 2\,H_2O$$

Usually, sulphate is ubiquitously present in water at sufficient concentrations, maintaining desulphurication at the sediment surface. However, H_2S is produced and this compound depletes remaining traces of oxygen, the last traces of oxidised nitrous compounds, and reacts with iron to form ferrous sulphide, which leads to the liberation of phosphates from the sediments in just the moment when all ferrous hydroxides are reacted to sulphides:

$$2\,FeO(OH) + 3\,H_2S \rightarrow 2\,FeS + S + 4\,H_2O$$

$$2\,FePO_4 + 3\,H_2S \rightarrow 2\,FeS + 2\,PO_4^{3-} + S + 6\,H^+$$

At this moment, free hydrogen sulphide poisons the sediments and prevents higher faunistic components in the benthic areas.

Usually, when this process is stopped by the lack of sulphate in deeper sediment layers, methane production takes place:

$$2\ CH_2O \rightarrow CH_4 + CO_2$$

Since methane is only slightly soluble in water, sediment mixing occurs by methane bubbles and phosphates are liberated to the pelagic waters causing excessive primary production.

The knowledge of these processes in the sediment, along with their time and depth distribution, provides possibilities for tailor-made solutions for sediment treatment with the aim of inactivating sediments. Usually, the application of both nitrate and iron at the sediment surface is necessary. The treatment of nitrate and iron is usually conducted in the latter part of the spring circulation.

Experience has shown that it is convenient to start with an iron treatment. It destabilises hydrogen sulphide and other sulphide compounds in the sediment:

$$2\ FeO(OH) + 3\ H_2S \rightarrow 2\ FeS + S + 4\ H_2O$$

Thus, these reduced compounds do not have to be oxidised in total by added calcium-nitrate solutions, which makes the nitrate treatment a little more efficient.

These treatments have to be evaluated by sediment experiments conducted in the laboratory where iron, phosphorus, and energetic conditions (amount of easy degradable organic substances) have to be analysed. Various additions of the agents have to be tested with the sediments in question. By conducting experiments with iron and nitrate treatment, both the areal dosage and the concentrations of these agents can be evaluated. For nitrate, the penetration of the surface sediment layers by calcium nitrate has to be examined. With respect to iron, it is necessary to establish a dosage sufficient for excessive phosphorus-binding, even if a desulphurication process in the sediments should occur. The case studies mentioned in this handbook (Lillesjön, Hambutten- und Karutschenpfuhl, Lake Gross-Glienicker) can only be regarded as examples. The dosages in these cases should not be seen as general recommendations.

In water bodies with low alkalinity, the introduction of iron compounds, in the form of iron chloride, could use up the buffer capacity, which would result in acid conditions in such lakes.

$$2\ FeCl_3 + 3\ H_2O \rightarrow Fe_2O_3 + 6\ H^+ + 6\ Cl^-$$

Acid conditions, however, retard the denitrification activity. Although the coenotic structure in the lake will change, few toxic effects to fishes and benthic organisms can be observed, as long as pH in the lake does not fall below about pH 6.

In order to prevent acid conditions, it might be necessary to add calcium carbonate, in the form of fine particles with a high reactivity, together with the iron, to provide a sufficient amount of buffer in the water and at the sediment surface.

In some cases, there is a high amount of iron accumulated in the sediments and this iron is shown to be sufficient for binding the phosphorus. In such lakes, the addition of iron can be omitted, and the addition of nitrate would be sufficient.

In other lakes, where iron has been depleted in the surface sediments by low redox conditions for long periods, only the addition of iron would be sufficient, especially when this measure is carried out in very shallow lakes, where water movement and oxygenation of the sediment surface is provided throughout the whole year. This is also true for lakes where artificial mixing or aeration measures are carried out at the same time, which was the case in Lake Gross-Glienicker (see case study, Chapter 8).

In our whole-lake studies, it was shown that for internal sediment stabilising measures, each lake has to be treated individually according to the morphometric conditions on the one hand, and the water transport through the system and its pattern during the year on the other. This shows that standardised dosages of chemicals or standardised areas to be treated, in relation to the total lake area, cannot be given. The treatment can only be evaluated by thorough investigations over an adequate time period, with a sufficient amount of sediment sampling stations in each lake. The sufficient number of stations required is determined by the homogeneity and isotropy (distribution) of the sedimentation process in each specific lake.

Application of the chemicals to the lake

In the first whole-lake treatment experiments, the chemicals were injected into the sediment with a harrow-like device dragged over the sediment surface in order to cover the total area to be treated. For Lake Lillesjön the treatment device is shown schematically in Figure 1. The concentrations of the treatment solutions were chosen in such a way that the distribution of the chemicals in the sediment surface was controlled by the gravity of the solutions.

It was shown that such treatments give excellent results; however, the costs of the chemical distribution are extremely high. Recently, some experiments have been conducted where the chemicals are mixed with the tributaries, or where the addition of nitrate was achieved by dephosphorised and nitrified effluents from treatment plants. It was shown that even this kind of treatment, when applied in a proper way, produces good and lasting results.

As an example of this kind of treatment, a proposal for the Schlei estuary (northern Germany) is shown in Figure 2. Iron applications were conducted by premixing the agents on shore and distributing the iron over the lake with a tubing mounted on a boat. The iron solution was introduced at about one metre below the water surface. Of course, the iron flocks colour the water temporarily. But within one or two days, the iron flocks settle to the

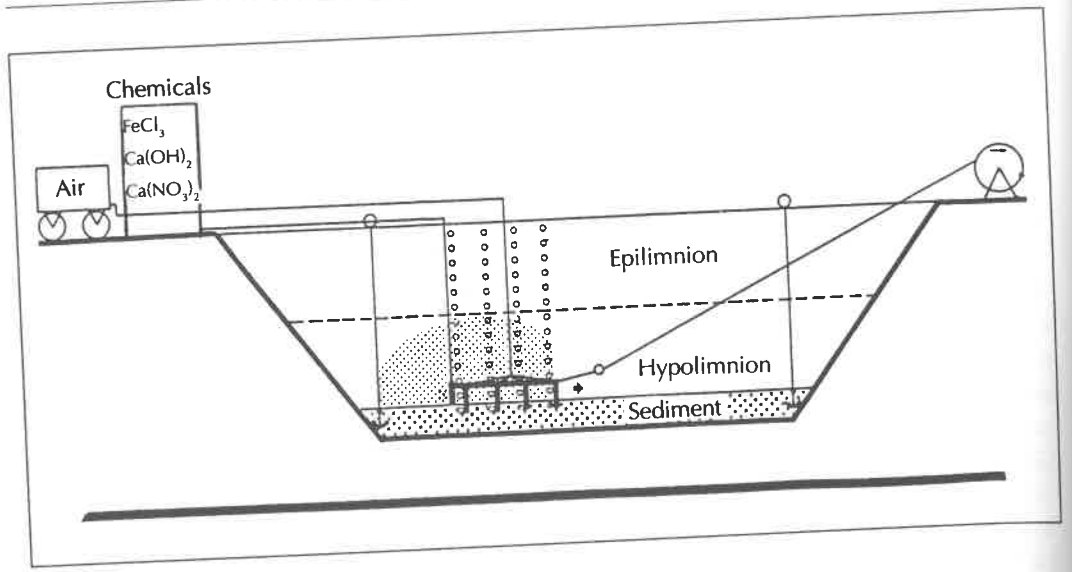

Figure 1. Equipment used for the treatment of Lake Lillesjön (southern Sweden).

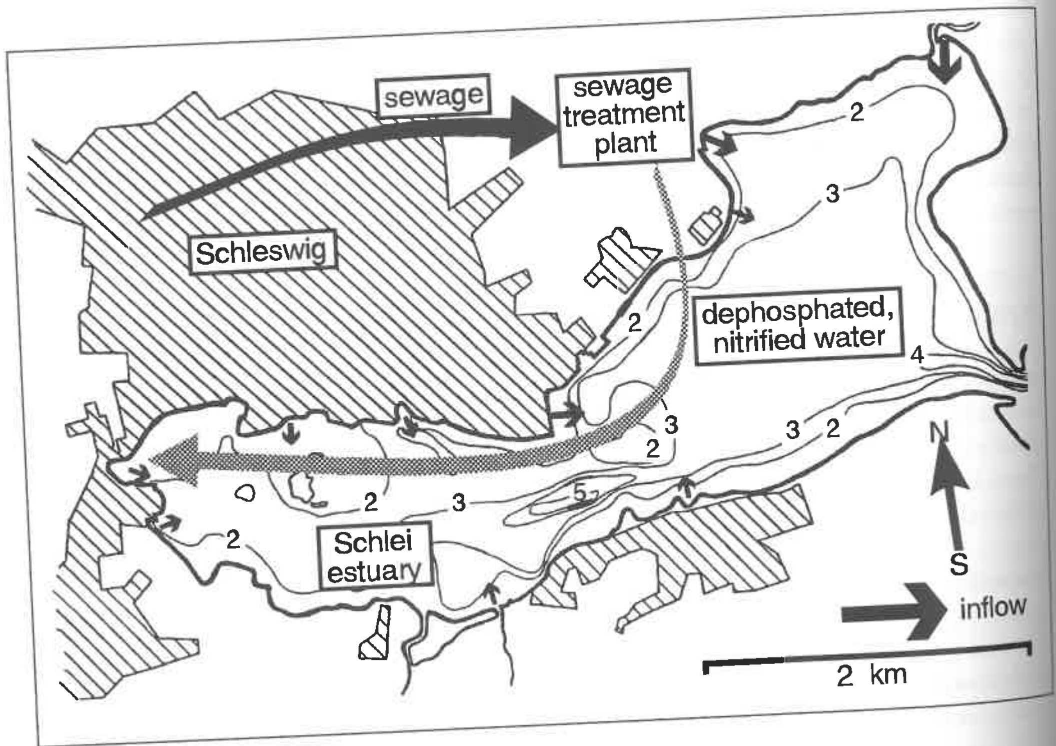

Figure 2. Proposal for the treatment of effluents of a sewage plant in Schleswig (Schlei estuary, northern Germany).

bottom, leaving a dephosphorised water which is necessary to prevent excessive primary production in spring and avoid the loading from organic degradable matter by algal sedimentation. The phosphorus bond to iron is possible through two different processes: the phosphate can be bound in iron-phosphate or adsorbed to iron oxide-hydroxide (see Phosphorus precipitation, page 63).

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Sediment removal

Sven Björk

Background

The removal of nutrient-rich sediment deposited in irreversibly damaged lake basins must be considered the most radical and definite method to restore such lakes. For a number of reasons this method should be practised only in shallow water bodies suffering from heavy sediment deposition. In waters deep enough to preserve their lake character with open water, other methods for the normalisation of their internal loading and metabolism should be applied, with the primary aim of stopping the release of phosphate from the top sediment layer into the water column. Compared with sediment removal such methods are technically less complicated, as well as being fast and cheaper. However, in shallow lakes in which the ageing processes are taking place at a high speed, sediment has to be removed in order to re-create sustainable, balanced systems.

Suction dredging

Technical requirements

For the removal of defined layers of top sediment without roots of growing macrophytic vegetation, suction-dredgers have to be used. As most water bodies selected for restoration through suction dredging are small, dredgers should be small, lightweight and easily transportable on a lorry from one lake to another. During dredging operations, no turbidity should be caused by nozzles, dredgers and pumping, because this is a sign of stirring up and moving sediment, indicating undesirable fertilisation of the lake by nutrient-rich interstitial water. In some of the early projects, commercially-available suction-dredgers which were not constructed for this specific purpose, had to be used. In the case of the Swedish demonstration project of restoration through sediment removal – the Lake Trummen project, (*cf.* case study, Chapter 8) – a special nozzle was developed, whereby it was possible to avoid turbidity.

In summary, experience gained during the 1960s and 1970s from sediment suction-dredging projects made it clear to limnologists that the dredging technique must meet the following three demands:

- no turbidity should be created;
- the admixture of water to the sediment should be no more than that required to allow it to be pumped. As a rule, the nutrient-rich top sediment can be pumped without any addition of water; and
- the proportions of sediment and water should be constant when homogenous sediment is pumped.

Horizontal control of dredgers

So far, two methods for controlling the horizontal movements of dredgers have been applied, viz. by means of:

- two hydraulically operated spuds (stabilisers) at the rear of the float;
- wire arrangements.

In the first case, the nozzle moves along the circumference of a semicircle as the float is fixed to the bottom by alternating starboard and port spud – or both spuds are fixed to the bottom and the nozzle is mounted on a moveable arm at the stem of the float. In the second case, the moving pattern is dependent on the arrangement of the wires from the float to the points of attachment on the shore or to anchors. In both cases, rows of buoys have been needed to mark treated areas.

Pumping and deposition of sediments

The conditions for the deposition of removed sediment on land varies from case to case depending on dredging technology and the space available at the specific water body selected for restoration. If it is necessary to deposit the pumped sediment in settling ponds, the run-off water, being a mixture of lake and nutrient-rich interstitial water, has to be treated in order to avoid pollution downstream of the settling ponds (Figure 1).

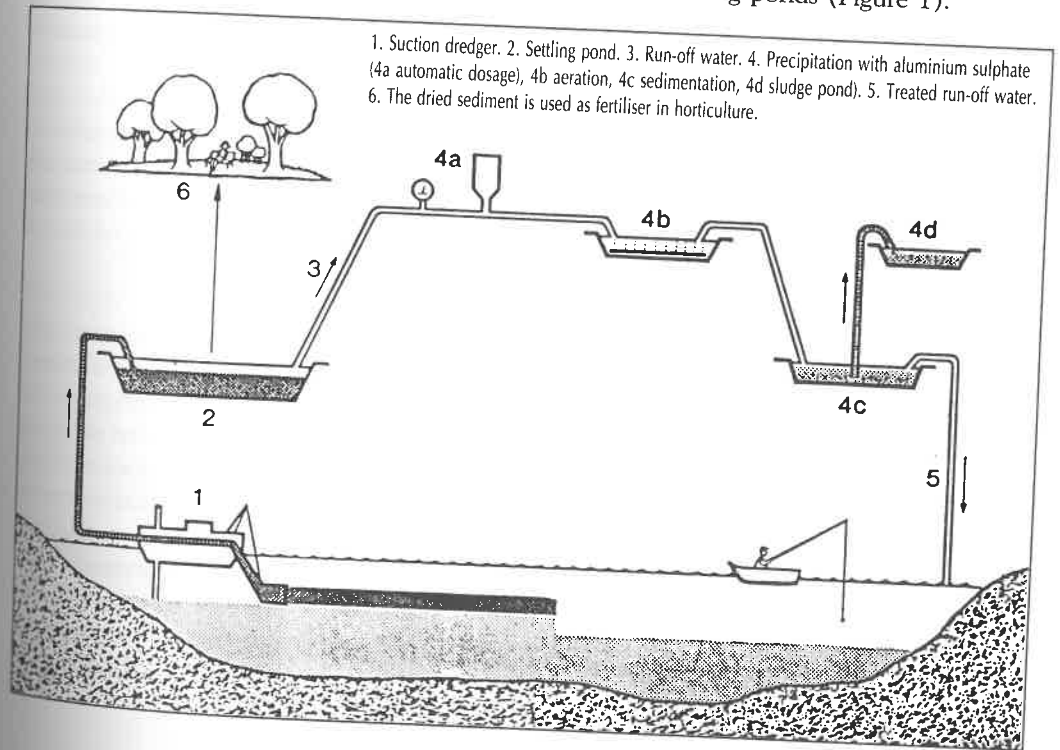

Figure 1. Schematic illustration of lake restoration by means of suction dredging. From Björk, 1972.

Because degraded lakes that need dredging are often located in urbanised regions, it is sometimes necessary to pump the sediment long distances to suitable deposition sites. This is made possible by means of the installation of booster pumps to increase the pressure in the pipeline, in order to overcome the distances and increases in elevation. In order to protect the booster pumps from mechanical damage, a grid for the separation of stones ought to be installed between the nozzle pump and the first booster pump.

One of the aims for a new, suction-dredging technology designed to meet the three limnological demands described above, is to make it possible to deposit the sediment directly onto fields. Thus the need to construct settling ponds and treat the run-off water is avoided. The new equipment is characterised by being automatically controlled. This is based on a continuous analysis of the pumped medium. Detailed maps showing the stratigraphy and horizontal distribution of the type of sediment that should be removed constitute the basis for the operation.

Automatically controlled suction dredging technology

In principle, an automatically controlled suction-dredger consists of three parts: the float, the nozzle, and the automatic control system.

1. The float is designed in the most simple way. Its size is determined by local conditions (size of the lake, waves, transport conditions, etc.). It is imperative that the size of the dredger corresponds to the size of the actual water body in order to optimise transport and installation costs.

2. The nozzles developed for soft organic sediment include pump and grid for protection against stones. (Primarily, the sediment (gyttja) to be removed does not include stones. However, stones, ammunition, bicycles etc., have often been thrown into lakes, especially in urbanised regions). For other types of sediment, a special design and function of the nozzle is needed to fulfil the limnological demands.

3. The control system consists of measuring devices from which signals are continuously transmitted to the unit directing the position of the nozzle that pumps a predetermined concentration of sediment, the speed of movement of the dredger, its location on the lake etc. The measuring devices of the automatically-controlled sediment pumping system include reading of water depth, density-, flow- and other meters for recording physical and chemical parameters of pumped sediment and sediment/water mixtures.

The type of precision dredging most often desired for lake restoration is the removal of sludge, layer after layer, by pumping sediment according to a programmed, constant, dry matter concentration. A serious drawback concerning old types of suction-dredgers was the pumping of large volumes of 'unnecessary' water. This made the construction of big settling ponds inevitable. Through automatically controlled pumping, 'unnecessary' water is avoided,

because the dredger can be programmed to pump sediment with the smallest possible admixture of water. This is accomplished through the continuous control of the nozzle's position and movement (horizontally and vertically).

In summary, the highly flexible, computerised, automatic control system makes it easier to find deposition sites for the sediment. The reasons for this are:

- The amount of water can be minimised or optimised with respect to transport costs, sediment mixtures for special purposes, deposition conditions, etc.
- In order to compose a soil mixture appropriate for agricultural field crops, horticultural mixtures, etc., fertilising elements, organic matter, and mineral particles of chosen fraction, can be continuously added to the pumped sediment before its deposition. The pH can be adjusted and substances for the binding of metals, for example, can also be added.

Provided the pumped sediment has proved to be suitable as a medium for fertilising and soil conditioning, direct deposition on arable land is a cheap and ecologically sound method of returning matter from eutrophicated water bodies back to the catchment area. Removal of contaminated sediment needs special caution and methods should be designed specifically for each project.

When contractors offer dredging operations for the restoration of lakes, the high pumping capacity of conventional dredgers is often emphasised as a competitive argument. However, high capacity is not at all the decisive qualification. On the contrary, it is often a disadvantage for this type of suction-dredging. Instead, precision in the removal of a limnologically-defined sediment layer is decisive. In this connection, high capacity dredgers are problematical because of the lack of correspondence they show between pumping capacity and the bottom area that has to be cleaned per unit time, in order to supply the dredger with the volume of sediment for which it has been designed. Because the dredger cannot possibly move at the speed required in order to collect enough sediment from a layer 20 to 50 cm thick, the action results in the pumping of either excess water or sediment which should remain in the lake.

The limnological aim of a restoration project by sediment removal from a eutrophicated lake is to carefully clean a defined *area* within which the polluted sediment causes the troublesome internal nutrient loading. It is the cleaned *area* that should be controlled in the lake after the project has been finished. The dredged/cleaned *area* is much more important than the *volume* of removed sediment, because some of the underlying sediment which for the lake ecosystem is harmless or, through its adsorptive capability, even beneficial, can have been removed, thereby unnecessarily increasing the total volume of pumped sediment. The high precision required to clean the bottom areas according to the limnological restoration plan, demands positioning equipment with an accuracy of the order of centimetres. The required precision for the lateral movements of dredger and nozzle is obtained by means of laser technology. Cleaned areas should automatically be recorded in detail during the dredging operation.

A prototype of an automatically controlled suction dredger was launched in Vajgar fish pond in the town of Jindřichův Hradec, South Bohemia, in July 1991, *cf.* case study, Chapter 8.

Removal of sediment overgrown by macrophytes

Eutrophicated, shallow lakes with heavy water blooms are typically devoid of submersed plants but often characterised by luxuriant stands of floating-leaved and emergent littoral vegetation. The presence of living roots in the sediment causes great problems and often makes suction-dredging impossible.

Before dredging a polluted lake with large stands of floating-leaved vegetation (*Nuphar*, *Nymphaea*, *Potamogeton* etc.), the petioles and stems should be cut immediately above the lake bottom. After that, the rhizomes and roots have to be cut by a rotavator mounted on pontoons (Figure 2). In soft bottoms, it is possible to loosen the rhizomes from the bottom by the use of cutting bars (one horizontal and one vertical), also mounted on pontoons (Figure 3). The loosened material, drifting at the water surface, is transported to wind-exposed shore sections. It is sometimes necessary to carry out this type of treatment in sub-areas surrounded by floating booms. After finishing the work within the sub-area, all loosened material is dragged, with the boom encircling it, to the shore where the material is removed from the lake by means of a conveyor belt.

The pre-treatment of the lake bottom, to get rid of rhizomes and roots, in order to make suction-dredging possible, ought to be carried out at least one year before the dredging starts. Root material remaining in the bottom has then decayed to such an extent that it is no longer a problem and the negative ecological effects (turbidity), caused by the treatment of the bottom, are remedied by the subsequent dredging.

Plaur formations developed in the lakeward vegetation zone should best be cut in portions suitable for being towed to the shore and removed by an excavator. Draglines, amphibious excavators, etc., are used for cutting the plaur into manageable pieces. Firmly rooted vegetation along shores should, if necessary, be removed by draglines and excavators. All such work should also be carried out in due time, for ecological reasons, before the final suction-dredging takes place.

Multi-purpose machines have been constructed for the removal of bottom material, including all kinds of living plant material, from densely overgrown wetlands (Figure 4). Pumps are mounted inside a bucket, in front of which a revolving drum with knives cuts roots and plant stems to a pumpable mixture. The machine is very easily transported on a lorry, has the ability to move into and out of a water body by itself, and to open up free water in, for example, drained and completely overgrown wetlands (see also Lake Hornborga case study, Chapter 8).

Figure 2. Pontoon machine equipped with rotavator for treatment of roots and rhizomes of aquatic macrophytes. Constructor Emil Cronqvist. Lake Långasjön, Sweden, 1972. Photo: Sven Björk.

Figure 3. Pontoon machine equipped with horizontal and vertical cutting bars. Constructor Emil Cronqvist. Lake Hornborga, Sweden, 1969. Photo: Sven Björk.

Figure 4.
Multipurpose dredger, type Lännen Watermaster. Constructor Lännen Engineering, Finland. The dredger is easily transported on a lorry and capable of loading and unloading itself as well as launching itself into wetlands and lakes. By courtesy of Lännen Engineering.

Excavated bottom material can also be loaded into a pulper/macerater, transferring it to hydraulic plunger pumps having the capacity to transport it (at a pressure of up to 100 bar) over several kilometres, even with differences in height of tens of metres.

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Macrophyte control

Sven Björk

Some information about methods and techniques for quantitative control and qualitative governing of emergent and floating-leaved macrophytic vegetation is given in the Lake Trummen-Vajgar and Lake Hornborga case studies (Chapter 8). In this chapter, some complementary additions are made to the information given in these case studies.

Notes on basic macrophyte ecology

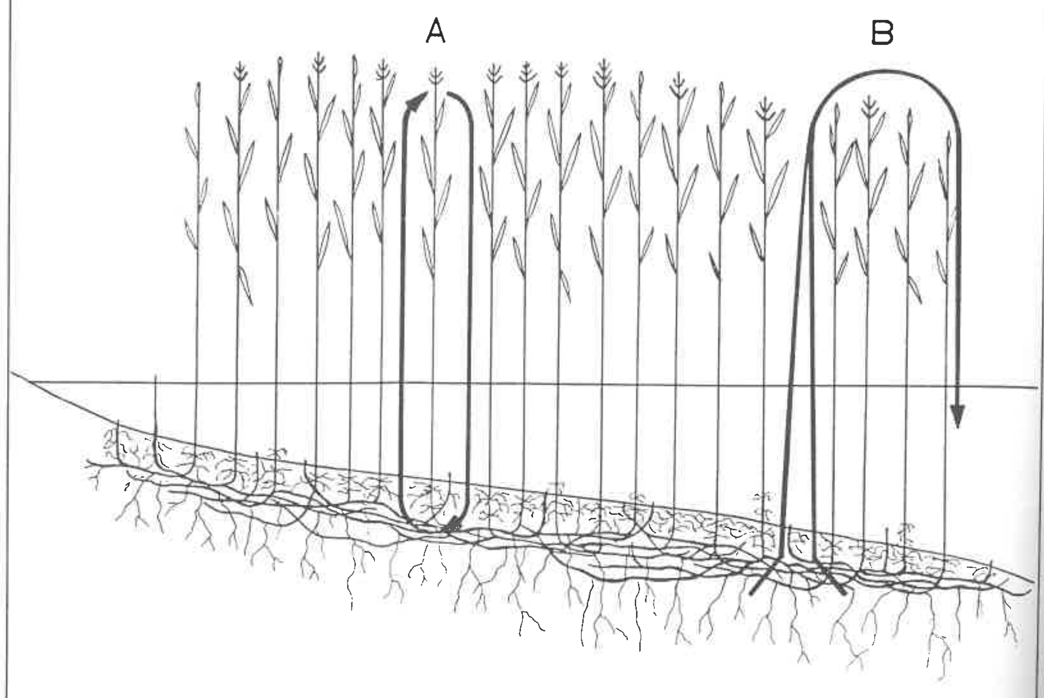
Degraded wetlands with open water, suffering from eutrophication, are characteristically turbid because of heavy plankton blooms during the summer. As the light conditions are bad in the water, such wetlands are typically devoid of submerged vegetation. However, floating-leaved and emergent macrophytic vegetation is often luxuriant with stands expanding. The lakeward margin of the reedbeds can be developed as plaur formations.

In shallow lakes which have become productive because of a moderate external supply of nutrients or by a lowering of the water level, submersed macrophytes, *Ceratophyllum*, *Elodea*, *Potamogeton* and *Myriophyllum*, sometimes reach such a quantitatively rich development that a reduction is needed to attain the goal of the management programme. Even charophytes, like *Chara tomentosa*, can appear in such masses that a reduction is required.

When dealing with lakes of this type, the reduction of submersed vegetation has to be made with care. This is because the ecosystem might oscillate between two different types of structure and function, viz. (1) one characterised by richly developed submersed vegetation and clear water, and (2) the other characterised by richly developed phytoplankton, turbid water and no or sparse submersed vegetation. A switch from submersed vegetation in clear water to phytoplankton and turbid water can also follow the introduction of grass carp (*Ctenopharyngodon idella*, which eats both common and rare plant species) as a means to diminish macrophytes. The available high concentrations of nutrients are utilised by primary producers, either the submersed macrophytes or the phytoplankton.

In perennial macrophytes with well developed rhizome and root systems, essential nutrients are assimilated from the bottom. In spring, these nutrients are translocated to the growing shoots but then, at least in part, returned, together with soluble carbohydrates, to the rhizomes before the end of the vegetation period (Figure 1). In contrast to those elements which are more or less preserved within the plant, other elements are not transported back to the rhizomes in autumn. This means that, for this group of elements, there exists a one-way transport from the bottom, via the plant, straight to the water and the top sediment layer (Figure 1).

Figure 1.
Schematic illustration of the role of *Phragmites* and other rooted perennial aquatic macrophytes in the transportation of elements within the limnic ecosystem. From Björk 1967.



Arrow A indicates the uptake from the bottom of elements which are transported to the shoots but returned to the rhizomes at the end of the vegetation period. Arrow B indicates the uptake from the bottom of: (i) elements which remain in the shoots at the end of the vegetation period and are eventually transferred to the sediment with the detritus and (ii) elements which are leached from the shoots and thus transferred to the water during or after the vegetation period.

The fact that essential nutrients can be preserved and repeatedly utilised in the development of shoots, as in a perennial plant like *Phragmites australis*, has important ecological consequences. Thus an occasional addition of nutrients (e.g. a temporary outlet of sewage) to the superficial soil layer can have a prolonged effect on the productivity, as tested in field experiments with sewage (Björk 1968). In lakes where eutrophication has resulted in the luxuriant growth of perennial macrophytes, diversion of sewage does not have any immediate effect in the form of reduced productivity because the essential nutrients are re-circulated. The plants assimilate as much nutrients as they need for their normal physiological function. Above this level, a 'luxury consumption' can appear.

Nutrients preserved within the perennial plant system can also have been taken up during a long time from the soil within the rhizosphere. The deeply rooted macrophytes in the minerogenic littoral zone frequently seem to be furnished with nutrients from the subsoil water from which an enrichment can take place. Thereby the standing crop can successively increase quantitatively. The nutrient concentration of the water penetrating the ground and seeping through the littoral zone is, therefore, important for the development of the deeply-rooted aquatic vegetation growing in minerogenic soil landwards from the organic sediments.

The reproduction strategies of different aquatic plant species (*cf.* Sculthorpe 1985) is of decisive importance for their dispersal and recolonisation within treated wetland areas. The quantitatively dominating species are characterised by rapid vegetative reproduction. Although the distribution of *Phragmites australis* is dependent on both generative and vegetative reproduction, the colonisation within a wetland with this species already established, most successfully takes place by means of rhizomes and drifting rhizome pieces. The seeds of *Phragmites* have very specific environmental demands for germination and development of seedlings. These demands are realised only under conditions corresponding to those found in the moist portion of the littoral zone or appearing over large areas during low water periods. New stands of the common reed never develop from submersed seeds, in contrast to *Schoenoplectus lacustris* which successfully colonises new areas in open water by means of both plant fragments and submersed seedlings (*cf.* Ekstam *et al.* 1992).

The aim of the management

In the planning of a management programme, all interests focused on the overgrown wetland area have to be taken into consideration, and the different approaches openly discussed to reach the best possible constellation of measures and results of biotope design. The most common combination of interests are recreation (like swimming, canoeing and windsurfing), ornithology and angling. In the last case, aspects on the ecological demands during the whole life cycle of actual fish species deserve attention. The overall aim ought to be to create varied environmental conditions to enhance a high diversity in flora and fauna. A mosaic of macrophyte stands and open water is most often optimal, although some bird species need large reedbeds for nesting and other species need large open water areas. Aspects such as the environmental requirements during migration, nesting and moulting periods, as well as the food requirements of young waterfowl, ought to be paid attention to.

Pre-project investigations and planning

If no possibilities exist to change the environmental conditions in order to obtain sustainable ecosystems, eutrophic shallow lakes of this type need repeated treatment. The dimensions of the wetland and the extent and cost of the work that has to be executed at different time intervals is, of course, decisive: whether it could be done at all, manually or using suitable machines. No work causing possible disturbance is allowed to be carried out during the breeding season of wetland birds.

All activities for the control of macrophytic vegetation should be preceded by a compilation of the deterioration history of the wetland, mapping of vegetation, water depth during the year, bottom conditions (organogenic, minerogenic, soft or hard, location of boulders etc.), nesting sites of bird species, etc. The information collected during the pre-studies is of great importance for the correct evaluation of results, as well as for correcting the improvement of methods.

Methods

Technical equipment

Among the three different types of methods generally mentioned in connection with the reduction of undesirable vegetation, i.e. biological, chemical and mechanical, only mechanical methods are treated in this handbook because they are the only ones possible to have under continuous and complete control. The available technical equipment covers a wide spectrum from simple scythes, small boats with cutting bars, to expensive aquatic harvesters. In addition to the amphibious and pontoon machines mentioned in Chapter 8 (Trummen and Hornborga case studies), a pontoon harvester for removal of submerged vegetation has been constructed for Scandinavian wetlands (Figure 2). The cut material is automatically loaded and brought to land by the harvester. Contrary to other machines with about the same function, this type can be easily transported from one site to another. Pontoon machines can also be equipped with rakes for collection and transport of cut material or masses of free-floating plants to the shore. The most suitable propulsion of pontoon machines is by paddle wheels (Figure 2).

The need to remove cut plant material

Plant material cut by mowing-machines or loosened by rotavators should be transferred to land. If, in large projects, plant material is disintegrated in a macerator and pumped to land, precautions should be made to prevent pollution from the sap. Easily degradable fresh material and plant sap consume oxygen and act as a fertiliser.

Slowly degrading matter like coarse *Phragmites* detritus covering the bottom constitutes an unsuitable substrate for submersed plants and for bottom animals. Such detritus is also easily moved along the bottom by water currents and can accumulate in stands of, for example, *Schoenoplectus*, originating from submersed seedlings in areas of open water. Contrary to the stems of *Phragmites*, those of bulrush (*Schoenoplectus*) wither in autumn. However, because the basal portions remain, the bulrush stands growing in open water act as very efficient traps for drifting detritus, both minerogenic and organogenic. Most intensively, the accumulation among stubbles and young shoots takes place during the windy spring high-water periods. In this way 'verlandungs'-islands are created. Because of the tough rhizome system of *Schoenoplectus* (acting like reinforcing netting) such vegetative islands are often difficult to get rid of.

Time schedule

In principle, the most suitable time for cutting perennial emergent plant species is when they have their highest concentration of nutrients in the shoots. Then the removal of the standing crop means a maximal depletion of nutrients from the plant. In the case of *Phragmites*, this stage is reached just before the appearance of the panicles. Repeated cutting

Figure 2. Pontoon harvester Limno-Combine with one horizontal and two vertical cutting bars (above). Automatic unloading of cut plant material (below). Constructor: Emil Cronqvist. Lake Magelungen, Stockholm, 1993. Photo: Stefan Cronqvist.

Figure 3. Re-growth of *Phragmites* shoots after burning of stems and the layer of accumulated coarse detritus. Lake Hornborga, Sweden (cf. Hornborga case study, Figures 2 and 3). Photo: Sven Björk 1973.

can result in the extermination of a species within a treated area. However, provided the environmental conditions are not altered, the same or other species will successively recolonise the cleared bottom.

Although the removal of green emergent plant material from a wetland also means a certain reduction in the total nutrient capital of the ecosystem, the real importance of this activity for the nutrient economy is rather negligible. However, when dealing with the huge biomass of submersed plants like *Ceratophyllum* and *Myriophyllum*, it is imperative to remove it from the wetland before any decomposition of the plant material has started, i.e. before the release of nutrients from the easily degradable matter.

Another aspect to observe is that the cutting should be done before seeds have become fully developed and germinative, and easily get lost from the cut material of, for example, *Potamogeton*. Both when cutting standing crop and rotavating the root systems of hydrophytes, especially submerged plants, late during the vegetation period, the cleaned area can also be seeded with hibernacula of various kinds, i.e. buds adapted for overwintering and propagation.

Use of fire

In wetlands overgrown by emergent vegetation and with the bottom covered by a layer of coarse detritus (Hornborga case study – Figure 3), it is sometimes possible to make use of fire to retard the ageing process or as a means to prepare for treatment of the root-felt (Hornborga case study – Figure 6). The best conditions for burning are mostly in late

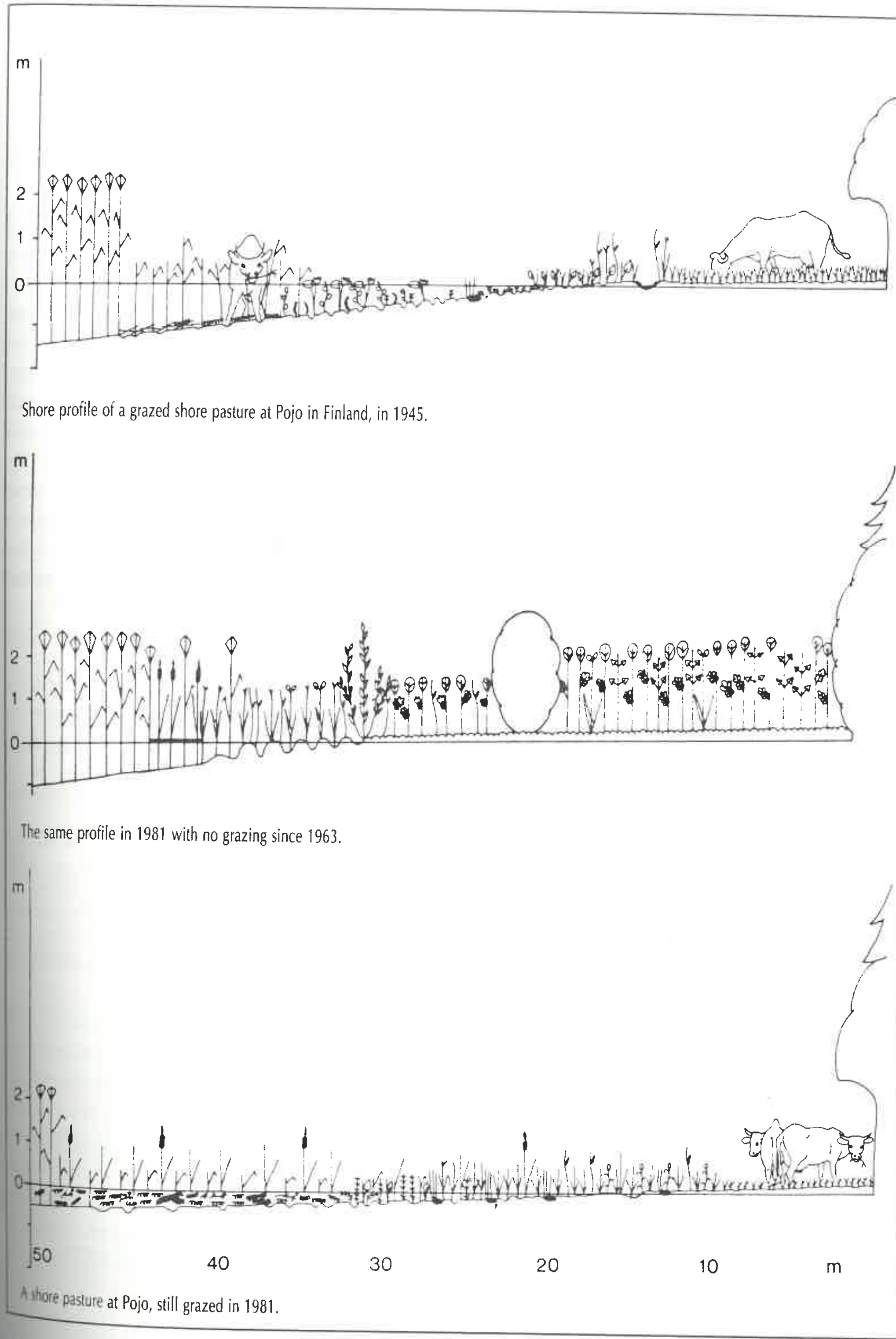


Figure 4. The effect of grazing on the littoral vegetation. From Luther & Munsterhjelm 1983.

summer and early autumn (appropriate training for firemen). Because the rhizomes and roots are not damaged by the fire (Figure 3) and because better light and temperature conditions appear after removal of the detritus cover, the procedure results in an increase in biomass in the following years. If the aim of the treatment is to get rid of the emergent vegetation, rotavation of the root-felt is necessary after burning the stems and coarse detritus layer.

Management of the littoral zone

In Sweden, cooperation between environmental protection agencies and the army includes, among other things, the use of all-terrain carriers for management of the transitional 'blue zone' of open water separating land from the richly developed emergent vegetation surrounding eutrophic lakes. As long as grazing by cattle existed, this zone was kept open up to the depth which cattle reached when grazing aquatic vegetation (Figure 4). When grazing stopped, it rapidly became overgrown, and nesting as well as foraging sites for waterfowl were lost. By means of military tracked vehicles (training of drivers) the root systems of the emergent perennial plants within this littoral zone is destroyed and it becomes productive with respect to a diverse fauna, insects, molluscs and flora (*Lemna*, *Ceratophyllum*, *Hydrocharis*, etc.). Like grazing, this form of artificial treatment has, of course, to be repeated along shore reaches not exposed to ice- and strong water movements.

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Food web management

Josef Matěna, Vojtěch Vyhnálek and Karel Šimek

Introduction

Hrbáček *et al.* (1961) and Hrbáček (1962) found that biotic relationships between the fish stock and the plankton are at least as important for the momentary composition of the plankton association as the influence of the physical and chemical factors. The term *biomanipulation* for measures used to reduce the consequences of eutrophication was introduced by Shapiro *et al.* (1975). *Biomanipulation*, or more correctly, *food web management* means the *top-down control* of the ecosystem. Primarily, it means the management of the fish stock. From the point of view of the management of water bodies, the aim of food web management can be defined as a cost-effective and nature-friendly way to improve some parameters of water quality. However, it has to be stressed that food web management cannot be regarded as a substitute for reduction of nutrient loads.

Methods and effects of food web management

There are several possible approaches to managing the fish stock:

- *poisoning* (e.g. Shapiro *et al.* 1982, Reinertsen & Olsen 1984);
- *selective catch* of undesirable planktivorous species (e.g. Benndorf *et al.* 1988, Kasprzak *et al.* 1988);
- *enhancement of the stock* of piscivorous fish (e.g. Benndorf *et al.* 1984, Edmondson & Litt 1984); and
- *water level manipulation* during spawning of planktivorous fish (Kubečka, unpubl. data).

The changes in fish stock (decrease of biomass of planktivorous species) induce a shift in size and even species composition of zooplankton. In an optimal case, large *Daphnia* species (*D. pulicaria*) replace smaller ones (*D. galeata*, *D. cuculata*). Large daphniids are generally more effective filtrators of algae compared with smaller species.

The *decrease in phytoplankton biomass*, as a consequence of increased grazing of herbivorous zooplankton, is the most important expected effect of food web management. A comparison of the phytoplankton biomass before and after the fish stock management can be used as a criterion of the success of this effort. Furthermore, the position under the 95% confidence limit of a chlorophyll-phosphorus relationship (e.g. Dillon & Rigler 1974) can serve as a suitable criterion of the positive effects of food web management (Hrbáček *et al.* 1978). In addition, effective management of food webs can be indicated by an increased concentration of soluble reactive phosphorus in the euphotic layer throughout the growing season (Hrbáček *et al.* 1978). Both the latter criteria are only applicable in phosphorus-limited systems.

In some cases, a reduction of total phosphorus was observed as a result of food web management (Reinertsen & Olsen 1984, Stenson *et al.* 1978, Scavia *et al.* 1986, Wright & Shapiro 1984). On the other hand, no reduction of nitrogen compounds can be expected.

The role of the microbial loop

Only recently has the importance of the microbial food web been recognised (Sherr & Sherr 1988). It plays an essential role in stabilising the effects of food web management. In plankton of most aquatic ecosystems, bacterial production amounts to about 30% of primary production on an aerial basis (Cole *et al.* 1988). If we assume a growth efficiency of 10–50% (Pomeroy & Wiebe 1988), planktonic bacteria process more than half of the organic carbon produced in aquatic systems. While the response of zoo- and phytoplankton communities to management of higher trophic levels have already been documented, the responses of heterotrophic microbial processes have become a 'hot' topic of recent limnological studies (e.g. Christoffersen *et al.* 1993, Jürgens 1994). Bacteria are grazed by a wide variety of protozoans and metazoans in freshwaters (Sanders *et al.* 1989, Berninger *et al.* 1991). Changes at higher trophic levels may cascade to bacteria and protozoans via a number of pathways (Christoffersen *et al.* 1993). For example, cladocerans can, and do, graze both bacteria and protozoans in addition to algae, whereas most copepods do not graze on bacteria (Stockner & Porter 1988).

In general, two extreme situations can be expected:

1. In systems with a low stock of planktivorous fish, large daphniids develop a high standing stock, which causes a decrease in protozoan abundance resulting in low protozoan grazing pressure on bacteria (Riemann 1985, Jürgens 1994). Direct predation on bacteria by macrozooplankton could funnel a significant fraction of bacterial production up the food web (Pace *et al.* 1990). Naturally this situation develops during the annual spring clear-water phase of most temperate freshwater bodies and is characterised, compared to the remainder of the season, by a significant drop in abundance of microheterotrophs: by a factor of 2–4 for bacteria, 5–15 for heterotrophic flagellates, and by a temporal decrease or even disappearance of ciliates (Šimek *et al.* 1990, Vyhnálek *et al.* 1991, Šimek & Straškrabová 1992, Jürgens 1994). It follows that especially protozoan abundance and species composition might be a sensitive indicator of the food web management efficiency, as indicated in experimental enclosures (Riemann 1985).

2. Under the same nutrient loading, a high abundance of planktivorous fish usually results in a much higher phytoplankton biomass. Bacteria are grazed primarily by abundant protozoans, and then relatively little bacterial carbon will be transferred to higher consumers because of respiratory losses at each trophic step. Since this situation is more common and develops naturally, during the spring or summer phytoplankton blooms, then the question arises: What is the fate of high primary production in the absence of large herbivores?

As recently reviewed by Sherr & Sherr (1988, 1992), heterotrophic nanoflagellates are as important in consuming prey of ≥ 2 µm (mostly phototrophic cells) as they are in consuming picoplankton, and overall protistan herbivory can rival, or exceed that of microcrustaceans under certain circumstances. The real proportion of primary production consumed by protozoa in such systems, however, as well as the strong shift in size structure and species composition which might be expected, has scarcely been documented (Beaver & Crisman 1989, Sherr & Sherr 1988, 1992).

Food web management – its merits and limitations

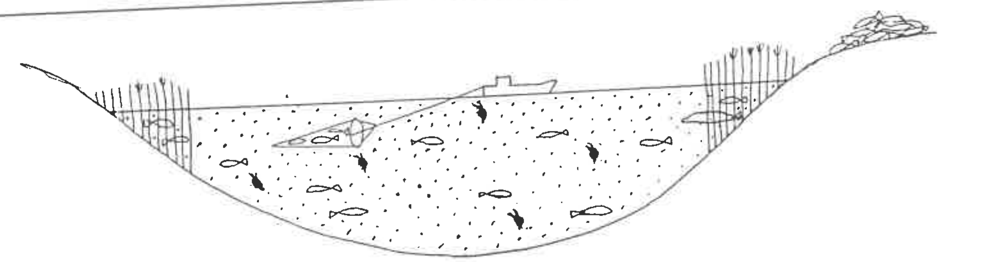
The current experience with management of the food web is not consistent. The effect of decreased planktivorous fish biomass on the structure of zooplankton was often documented, but not always a positive effect on phytoplankton biomass was observed (DeMelo *et al.* 1992). Reduction of phytoplankton biomass was achieved more often in small shallow waters than in large stratified ones.

The increased grazing pressure of herbivorous zooplankton can favour large non-edible species, especially colonial blue-green algae (Hrbáček 1964, Porter 1977, Lynch 1980). Therefore, changes in size structure, species composition and seasonal development of phytoplankton can be the consequence of food web management, but the average biomass will remain unchanged (Benndorf *et al.* 1988). In shallow waters, governing the food web can induce an excessive development of macrophytes (van Donk *et al.* 1990). The basic question then becomes: 'How to prevent growth of non-edible phytoplankton species or macrophytes at high levels of phosphorus and increased water transparency'. From this point of view, the aim of food web management (low biomass of phytoplankton at high levels of phosphorus and sufficient light) can be understood as an 'artificial situation'. The tendency of the ecosystem to return to the 'natural situation' can be expected and therefore, a constant input of energy will be necessary to maintain and stabilise the positive effect of the food web management.

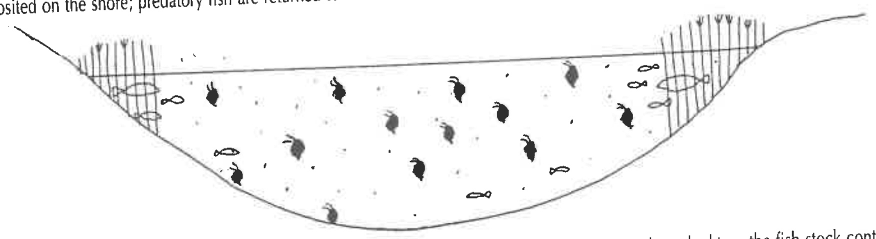
It is known that management of the food web is more efficient at low levels of phosphorus input. In the case of high external (Benndorf *et al.* 1988) or internal (Lammens 1988) phosphorus loading not all the desired improvements can be achieved. The *food web management-efficiency threshold* seems to reach a value of about 0.6 g P m^{-2} $year^{-1}$ (Benndorf 1987, Benndorf & Miersch 1989). Therefore, food web management is recommended as an effective tool accelerating the recovery of lakes after a reduction of phosphorus loading (Søndergaard *et al.* 1990).

Even under high nutrient loading a positive effect was achieved using both top-down and bottom-up control, in London reservoirs (Duncan 1990). These reservoirs are supplied with water from the eutrophic River Thames (7 mg N l^{-1}; 1 mg P l^{-1}). The jetting water input prevents stratification and ensures a thorough vertical mixing of the whole water column. Due to the depth of these reservoirs of 11 to 23 m, the phytoplankton is regularly transported into deeper water layers with insufficient light and, thus, its growing potential is light-

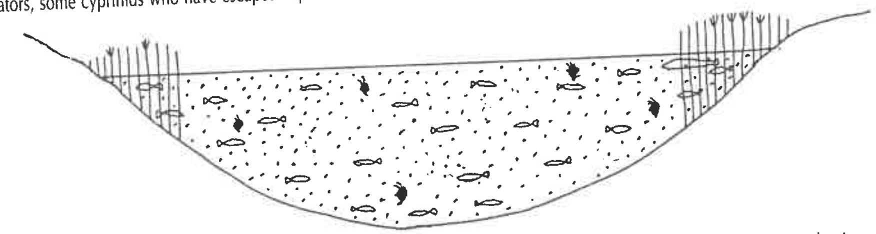
Figure 1. Schematic illustration of the effects of 'restoration' efforts by reduction of planktivorous fish in a polluted lake with high concentrations of nutrients.



Trawling to reduce cyprinid fish biomass in order to favour the development of large daphniids. The more easily caught first few tonnes of fish are deposited on the shore; predatory fish are returned to the lake.



The theoretical aim to be realised supposedly with the development of large daphniids grazing on phytoplankton; the fish stock containing predators, some cyprinids who have escaped capture and fish who have immigrated from water bodies connected with the lake.



Excellent reproduction conditions (physical, food) for cyprinid fish results in a rich biomass of uncatchable small fish, reduction of zooplankton and, after a period of oscillations, re-establishment of high biomass of phytoplankton, including water blooms typical for the polluted lake exposed to external and/or internal loading.

limited (bottom-up control). The low fish biomass of 20–150 kg ha^{-1} (Duncan & Kubečka 1994, Seďa & Duncan 1994) presents the top-down part of management of these reservoirs. As a result the algal development is reduced by the grazing impact of considerable biomasses of large daphniids – *Daphnia magna*, *D. pulicaria* and *D. galeata*.

In stratified reservoirs used for drinking water supply, the water is often taken from the hypolimnion with generally low numbers of living algae. Penetration of algal cells into the drinking water does not cause serious problems in this case. On the other hand, the hypolimnion in deep reservoirs is often lacking oxygen coinciding with high concentrations of manganese. It is possible to suppose that a further positive effect of food web management in stratified water bodies could be the improvement of oxygen conditions in the hypolimnetic zone as a consequence of the reduced amount of phytoplankton sedimenting from the epilimnion. This could help to overcome problems connected with high concentrations of dissolved manganese in an anoxic hypolimnion. Therefore, additional criteria, such as oxygen and manganese concentrations, should be introduced in stratified drinking water reservoirs.

There is only a poor knowledge about relationships between the effects of food web management and the technology of drinking water production. The most recent results showing the relationship between toxic compounds and trophic structure of aquatic ecosystems (Barica, *pers. comm.*) have to be taken into account. Higher accumulations of toxins in organisms was found in oligotrophic rather than in eutrophic water bodies.

Conclusion

Positive effects of food web management can be more easily achieved in smaller, fully controlled, shallow water bodies, such as fish ponds, than in natural lakes and reservoirs. The situation that often occurs in lakes and reservoirs where reduction of planktivorous fish was attempted is schematically illustrated in Figure 1. As it is very difficult to achieve the total removal of planktivorous fish from natural lakes and reservoirs, the fish left in a lake reproduce at a very high frequency and the conditions in the lake may revert to the situation which existed before the fish removal.

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8. Case studies

Control of external nutrient loading – Lake Ringsjön, Sweden

Bo Verner

Background

Overloading of lakes by nutrients from external sources often creates long-term eutrophication problems. To re-create acceptable environmental conditions in lakes suffering from eutrophication, it is essential to control the source of nutrient loading. In some cases, the reduction of *external loading* is sufficient to return a lake to its previous condition, in others, where *internal loading* from the nutrient-rich sediment prevents recovery, in-lake restorative measures need to be employed.

The control of external load, however, of both *point and non-point sources* of nutrients, should always be incorporated into any restoration project. Lake Ringsjön, in southern Sweden, is a good example illustrating the type of measures undertaken in order to reduce diffuse nutrient loading from surrounding farmland.

General description and type of disturbance

Lake Ringsjön is situated in Scania, the southernmost county of Sweden. Its drainage area is 347 km² having 14 tributaries and one outlet, the river Rönneå (see Figure 1). The land use of that area is illustrated in Table 1. The northern and northwestern part of the drainage area is covered mainly by forest with many recreational houses while the southern part is arable land. The population in the area is 21,000 people of which 15,000 live in urban areas. The lake comprises of three basins: Lake Sätoftasjön, Östra Ringsjön and Västra Ringsjön with a total area of 40 km² and a volume of 184 Mm³ (Table 2).

	Forest	Arable land	Wetland	Urban area	Other	Total
ha	13480	13030	1020	420	6750	34700
%	39	38	3	1	19	100

Table 1. Land use in the catchment area of Lake Ringsjön

	Sätoftasjön	Östra Ringsjön	Västra Ringsjön	Total
Area (km²)	4.2	20.3	18.4	42.9
Maximum depth (m)	17.5	16.5	5.4	17.5
Mean depth (m)	3.0	6.1	3.1	4.7
Volume (Mm³)	12.8	124.8	46.6	184.2
Retention time (year)	0.33	0.86	0.3	1.11

Table 2. Morphometry of the three basins of Lake Ringsjön

The water level of Lake Ringsjön has been regulated. In 1883, the lake surface was lowered by 1.5 m in order to obtain arable land. Until 1987, the lake was used as a drinking-water supply, with a maximum allowed withdrawal rate of 1125 l s^{-1}

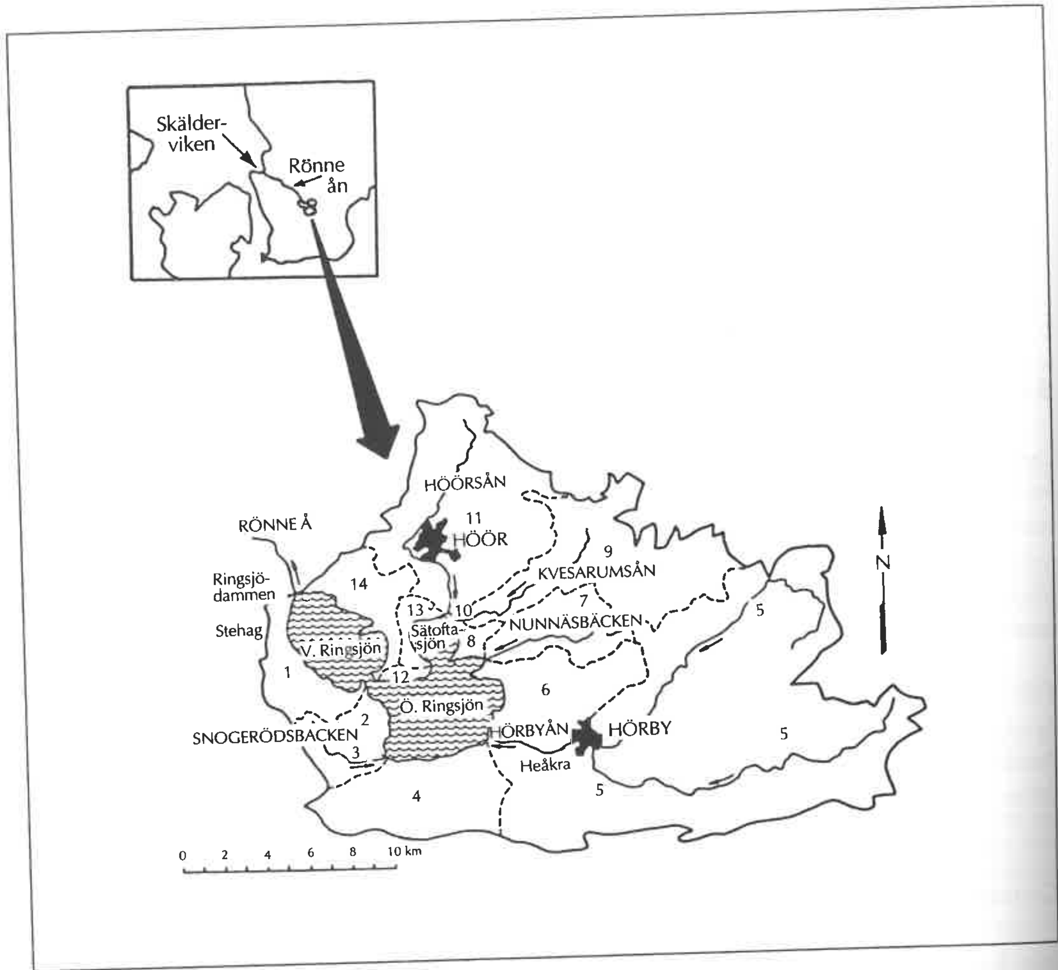

Figure 1.
Location of Lake Ringsjön in southern Sweden.

according to the concession given. Later, due to increased water demand from the cities in northwest Scania and due to degrading water quality, Lake Ringsjön was abandoned as a water supply source.

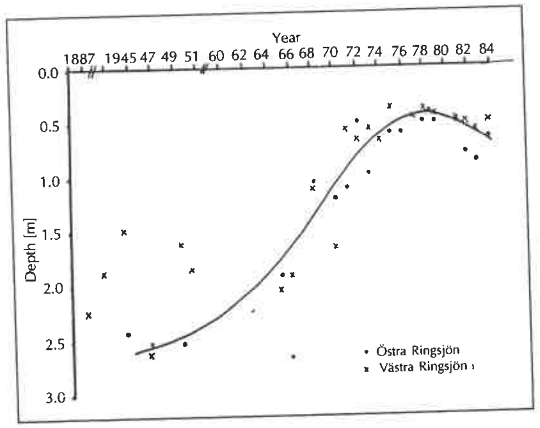

Figure 2.
Summer transparency of the water in Lake Ringsjön from 1887 until 1984.

Since the early 1960s, severe algal blooms have occurred (Björk & Lettevall 1968) and the water transparency rapidly decreased (Figure 2) as a result of increased phosphorus and nitrogen concentrations, caused mainly by diffuse leakage of these nutrients from farmland. The water quality decrease has meant that nowadays Lake Ringsjön is rarely used for swimming and that professional fishing has almost totally ceased.

Investigations

Degradation of water quality in Lake Ringsjön had been noted in the 1960s. This resulted in demands for further treatment of the sewage water from municipalities around the lake. However, thorough studies in the 1970s revealed that the improved sewage treatment which was done in 1975 and 1978 in the towns of Hörby and Höör and their adjacent settlements did not have the expected effect, e.g. nutrient concentration in the lake and its tributaries did not decrease. The need for an extensive lake redevelopment and restoration programme was realised and a committee formed in 1980. This comprised representatives from the cities and communities around the lake, farmers, fishermen, nature conservationists and lake home owners. A survey carried out in 1981 showed that 70% of the phosphorus and 80% of the nitrogen was coming from farming. The limnological investigation in 1981–82 revealed the following:

- the water flow, quality and matter transport in the main tributaries showed big variations. A distinct relation between the percentage of farmland and the concentration of dissolved nutrients in the tributaries was found. Direct measures of the annual P and N supply indicated that the land around the lake leaks more nutrients than was estimated. The P-leakage from farmland is amongst the highest measured in Sweden.
- 90% of the annual matter transport to Lake Ringsjön takes part during the winter months. A very small part is supplied during the vegetation period from May to September.
- the relative importance of the pollution from sewage treatment plants has decreased considerably. The matter transport from all sewage treatment plants contributes roughly 1% of P, 1% of organic matter, and 5% of N, and is therefore negligible considering the total load to the lake.
- the direct deposition of P and N on the lake surface from rain made up for 7–8% of the total load.
- the internal phosphorus load varied considerably between the years studied. In all three basins and in particular in Östra Ringsjön, the phosphorus released from the sediment formed the biggest pollution source during the summer. Even without any external load, the water quality in Lake Ringsjön would be very poor.
- the water quality in the lake has markedly degraded since the middle of the 1960s. In ten years, the value of total P during the summer was almost five times higher and that of total N three times higher.
- since 1890, a change in phytoplankton composition has occurred. The increased nutrient load resulted in an extensive development of blue-green algae and simultaneous reduction of diatoms and N-fixating algae. Since the 1960s, the total biomass of phytoplankton has rapidly increased.
- a successive accumulation of P and N in the top sediment layer has occurred during the 20th century. This is especially substantial for P in Lake Sätoftasjön and for N in Östra Ringsjön. Regarding heavy metal contamination, Lake Sätoftasjön is the most exposed, especially considering cadmium, copper, lead and zinc. The PCB content was high in

all three basins, especially in Lake Sätoftasjön and has to be considered a high potential ecological risk.

It was concluded that Lake Ringsjön has been exposed to an unnaturally heavy load of nutrients for many years. The supply of fertilisers from the surrounding farmland is still causing deterioration of the water quality. In addition, nutrients are released from the sediments during the summer. To improve the water quality of Lake Ringsjön, the annual supply of phosphorus should be reduced by 70%, i.e. the existing load reduced from 30 tonnes to about 10 tonnes. To reduce the supply of nutrients from the tributaries and restore the lake sediments will take a very long time, if it is at all possible given economic constraints.

Measures undertaken

In 1985, Lake Ringsjön and its drainage area was declared specially sensitive to pollution according to a supplement of the Environmental Protection Law, with the following regulations and recommendations:

Regulations under the Environmental Protection Law

Use of fertilisers

1. The application of manure, sewage sludge and commercial fertilisers to farmland must be based on a survey of the nutrient content of the soil, especially considering phosphorus, and not exceeding given recommendations. This soil survey of all land should be done by 1988.
2. Manure spread on unsown land should be ploughed in, within 36 hours.
3. Manure spread during the period from 1 December to 28 February must be ploughed in, on the same day.
4. Manure must not be spread within 10 metres of an open ditch or water course, if the ploughing did not take place the same day as the manure was spread.

Storage of fertilisers

5. The storage facilities for manure should be furnished such that all manure, urine and liquid manure can be collected. The storage capacity should correspond to at least eight months production. Farms with more than 15 heads of cattle must have a tank for urine and liquid manure. These arrangements must be ready by the end of 1990.
6. Storage of manure directly on the ground (compost) is not allowed during the period from 1 December to 28 February.
7. The storage capacity for silage, beet mass and other juicy feed stuff must be designed so that all juice can be collected.

Drainage

8. Drainage water from the milking rooms of dairy farms with more than eight cows should be treated in sludge traps followed by an infiltration bed or collected in a manure tank. If washing-up detergent with a phosphorus concentration less than 2% is used, then treatment in a sludge trap followed by a filter outlet will be sufficient. These arrangements should be ready by the end of 1990.
9. Sewage water from a single farm should be treated, at least in a sludge trap followed by infiltration. Other similar treatment methods are valid. This arrangement must be ready by 1988.

Recommendations according to paragraph 39 of the Environmental Protection Law

Fertilising

1. Manure should be analysed with respect to the nutrient content. If there are large deviations from average values, the analysis should be repeated.
2. The supply of commercial fertiliser should be adapted to the result of the soil analysis, the amount of manure used and the harvest level.
3. Manure should be spread primarily in spring to crops having a long vegetation period and secondly to crops sown in autumn.
4. Manure should not be spread after crops of potatoes or peas.
5. The spread of commercial fertilisers on frozen soil should be avoided.
6. Manure and commercial fertilisers should be carefully ploughed in, especially on fields where surface water appears.

Cultivation

7. A protection zone of at least one metre should be kept along open ditches or other water courses, when ploughing or using other cultivation means.
8. On steep slopes the ploughing should be carried out so that erosion caused by surface drainage is prevented.
9. Harrowing in autumn should be avoided in erosion sensitive areas.

Miscellaneous

10. Detergents with no, or low, phosphorus content should be used whenever the water quality permits.

The Regulations 1–9 are mandatory according to the Environmental Protection Law while the Recommendations 1–10 are advisory which, however, can be made imperative by the County Government Board.

Effects of the Regulations

The external loads of phosphorus from the tributaries have decreased, on average, from 27 tonnes in the 1980s to about eight tonnes in the beginning of the 1990s, which corresponds to the reduction required. The variations between different years, however, are large, determined by the precipitation and its distribution over the year.

The reduced load can mainly be assigned to:

- extension of sewage treatment plants in some cities
- improvement of single drainage systems
- development of manure storage facilities
- improvement of milking shed drainage and use of low phosphorus washing-up detergents
- rational use of manure based on the soil nutrient content and prevailing spring spread
- erosion preventive measures
- extended areas of 'green land' outside the vegetation period

Table 3. Costs of research projects to restore Lake Ringsjön

Research project	Cost (thousands US$)
Limnological investigations	58
Agricultural investigation (1)	110
Investigation of tributaries	51
Agricultural investigation (2)	31
Cultivation of water plants	850
Cyprinid reduction	783
Protective crops	70
Total	1,953

The reduced phosphorus load to the lake resulted in lower P-concentrations in the lake water, from more than 300 µg l^{-1} in the beginning of the 1980s to about 50 µg l^{-1} in 1989. The reduced P-concentration has resulted in lower algae production. This has influenced composition and size of zooplankton which in turn has changed conditions for the fish community.

Research projects

Various projects of an experimental character, aiming to test methods of restoring the lake, have been presented to the committee (Table 3). Among these, the following ones were chosen to be carried out:

- nutrient reduction by cultivation of water plants
- field experiments with protective crops
- lake restoration by reduction of cyprinid fish

Overall evaluation

Since the beginning of the 1980s, Lake Ringsjön has been at the centre of the debate regarding diffuse nutrient loading from farmland. The eutrophication of Lake Ringsjön, its status, the nutrient leakage from the drainage area and the enforced regulations on farmers will all be of great interest in future debates on the environment. The studies carried out and still continuing in Lake Ringsjön are probably the most extensive in Europe.

The regulations have resulted in a considerable reduction in the external nutrient loading and there are some signs of improved water quality. However, due to natural variations in the weather, rainfall, farming, etc., and due to the internal loading, it is far too early to conclude whether the measures taken will be sufficient to regain earlier levels of water quality and how long this will take.

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Restoration of eutrophic lakes by phosphorus precipitation – Lake Gross-Glienicker, Germany

Klaus-Dieter Wolter

Introduction

Until 1989 the city of West-Berlin was isolated from the surrounding landscape by the Berlin wall. Lake Gross-Glienicker (in German, Gross-Glienicker See) was located right on the border between West-Berlin and former East Germany. From the point of view of water quality, the lake was one of the best in Berlin during the 1950s and 1960s. However, during the 1970s and 1980s, the lake showed increasing amounts of phosphorus. Due to the political situation, there was little possibility to start investigations and to control the high nutrient loading. Fortunately, since the unification of Germany, it was decided to redevelop the catchment and to restore the lake.

Pre-investigation showed that it was possible to divert the nutrient loading away from the lake. However, no response of the lake to the removal of external nutrient loading was observed. Given these circumstances, restoration measures in the lake itself had to be undertaken. The precipitation of phosphorus from the water, by treatment with iron, seemed to be a suitable measure. In lakes, like Lake Gross-Glienicker, with high amounts of easily degradable organic substances in the sediment, a release of iron-bound phosphorus from the bottom can occur. To increase the mineralisation of the reduced sediment and to avoid the reduction of iron compounds in the sediment, oxidising agents need to be delivered to the sediment surface.

Catchment area and lake basin

Lake Gross-Glienicker developed from a subglacial channel during the last Weichselian glaciation. It is orientated from north to south, the channel shaped basin being 1800 m long and about 400 m wide. The lake area is 0.68 km^2, with a maximum depth of 11 m and mean depth of 6.5 m, its volume being 4.2 Mm^3.

The lake catchment area (Figure 1), determined by the topography of the surroundings, is about 16 km^2, mostly used by urban settlements, agriculture and forestry. In the 1920s and 1930s, the settlement had the character of a holiday settlement. Since the 1950s and 1960s, more and more houses were built, but no central sewage treatment was installed. Sewage was disposed into collection hollows. It is probable that some of these hollows were not water tight and worked like soakaways, thereby loading the groundwater with nutrients.

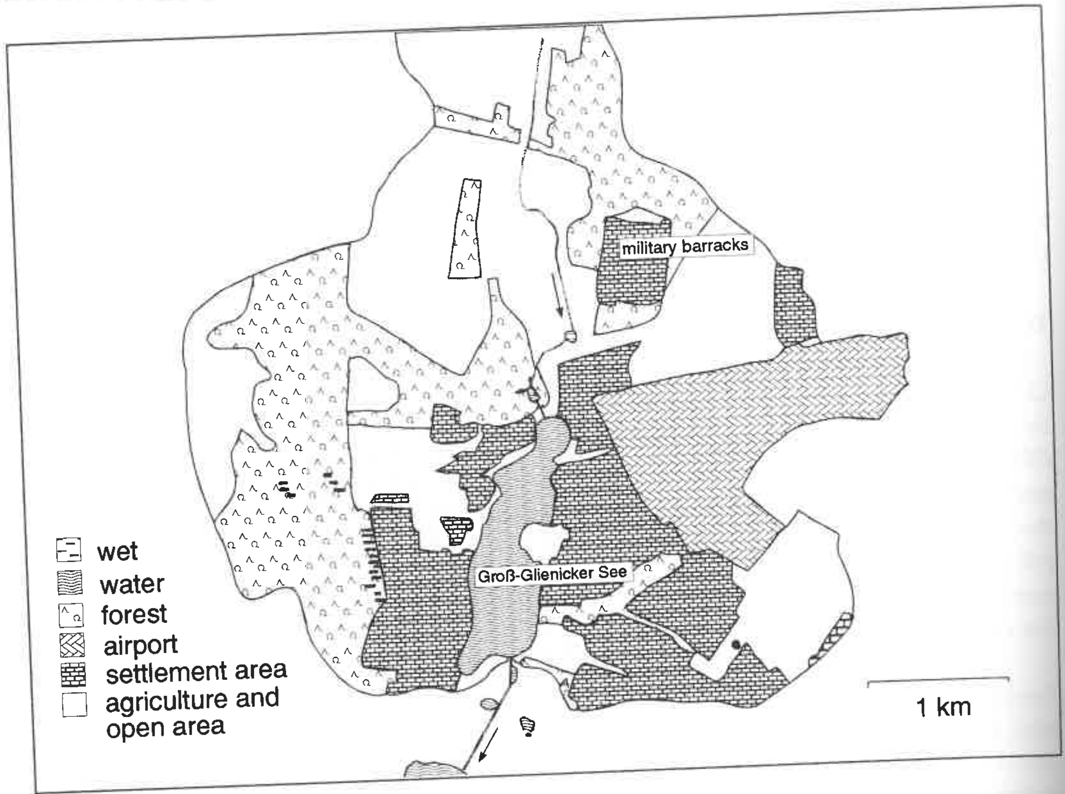


Figure 1. Land use in the catchment area of Lake Gross-Glienicker.

In 1972, military barracks were built to the north of the lake. Their sewage treatment plant was almost ineffective and the sewage, after passing two little ponds, flowed into Lake Gross-Glienicker. About 380 m^3 of sewage reached the lake per day and most likely, this was one of the most significant sources of nutrients.

Since the 1920s or 1930s, a waterworks had been located on the east side of the lake. As a consequence, the groundwater table had been lowered, preventing some of the nutrient-loaded groundwater from flowing into the lake. However, the pumping of the groundwater also brought negative consequences. The soil and the aquifer were degraded by leaching of nutrient-binding agents like iron, small silt and clay particles. This was probably the reason why, within a few years after the closure of the waterworks in 1977, the concentration of phosphorus in the lake water increased remarkably.

Given the relatively small catchment area and the restricted inlets, it was predicted that after the main sources of nutrient loading were cut-off, the lake could be redeveloped successfully.

Processes in the lake

In the Berlin region, the prevailing wind is from a westerly direction. Due to the north–south orientation of the lake basin and lack of kinetic energy brought into the lake by any inflow, the stability of thermal stratification is high. In normal years, the stratification develops in April or May, with a thermocline in summer between 5 to 6 m.

Lake Gross-Glienicker showed some typical signs of a lake with a high nutrient content. In summer, the transparency of water decreased to values lower than 1 m and blue-green algae became an important phytoplankton group. In the bottom water layer, the hypolimnion, oxygen was depleted usually by May, soon after the establishment of thermal stratification. Consequently, hydrogen sulphide was formed.

The littoral zone of the lake was poorly developed. Reeds were missing in nearly all parts of the lake, submersed macrophytes could hardly be found. Therefore, the positive benefits of these surface enhancing structures for nutrient cycling was missing. Coenoses in the lake were dominated by phytoplankton. Due to production of hydrogen sulphide, the benthos was poorly developed.

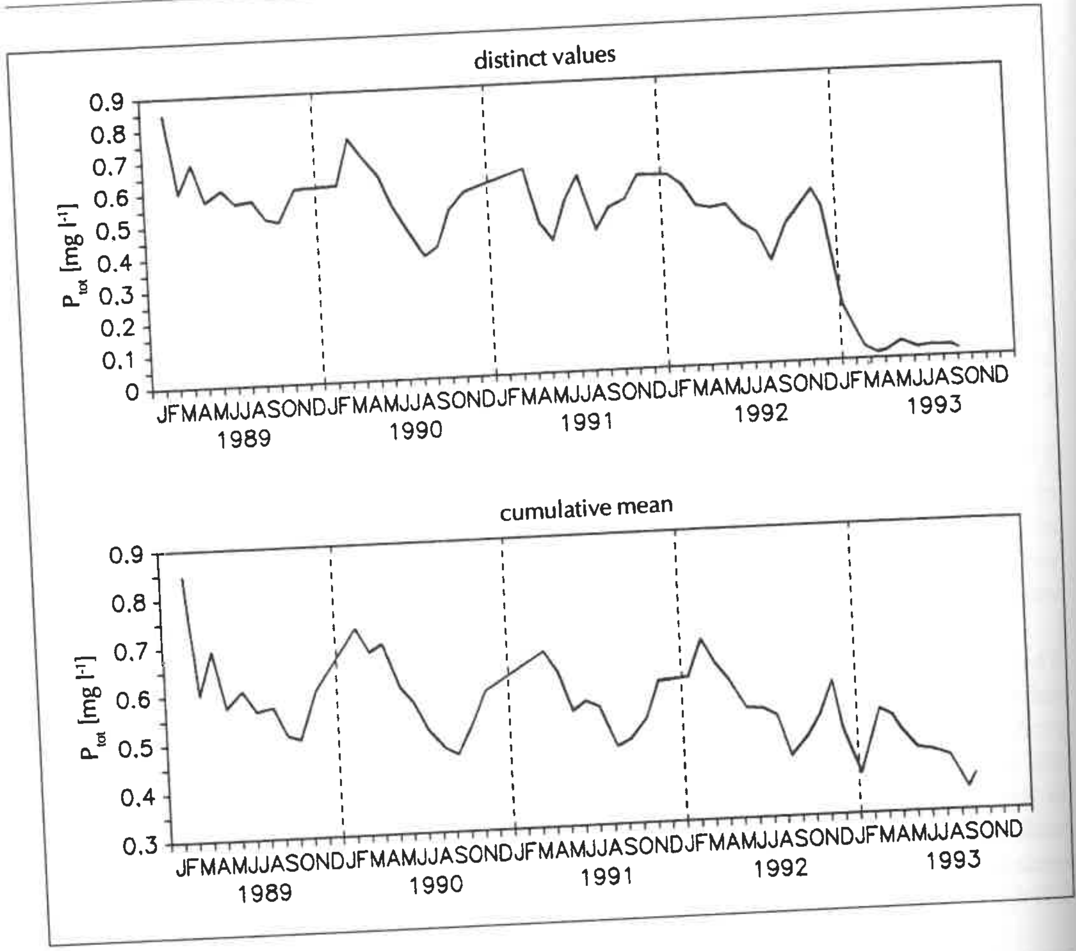
The retention of phosphorus in sediments is controlled by processes of sulphate and nitrate reduction. In Lake Gross-Glienicker, the large amount of easily degradable organic matter produced by phytoplankton in summer, settled down to the bottom of the lake. As oxygen and oxidised nitrous compounds (nitrate) were lacking in summer, the organic substances in the sediments were degraded by sulphate reduction mainly. In this process hydrogen sulphide is formed. The hydrogen sulphide can reduce iron and form iron sulphide thus releasing the iron-bound phosphorus. Usually, sediments with high S/Fe-ratios (above 1 or 1.5) do not show a good phosphorus-binding capacity. (See also Chapter 7, Sediment treatment).

With sediment analysis of iron, sulphur, and phosphorus, the state of the sediments can be evaluated. In Lake Gross-Glienicker, the molar ratio of sulphur to iron (S/Fe) was 1.3, indicating that a relative high amount of iron is present in the form of iron sulphide. Consequently, relatively little phosphorus was found in the sediments (0.2% dry matter).

As shown by pre-investigations, the upper sediment layer contains about 33% dry matter of organic substances, most of them easily degradable, about 22% dry matter of lime, and up to 41% dry matter of acid insoluble residue (AIR). Besides these main components, the sediment content of iron, sulphur, and phosphorus is particularly important for the evaluation of phosphorus exchange between water and sediment. In Lake Gross-Glienicker, sulphur reaches 1.5% dry matter and phosphorus about 0.2% dry matter, while the iron content of 2% dry matter is very low in comparison to other Berlin lakes. Depending on the geological structure of the catchment area, some lakes in the Berlin region contain much more iron in their sediment. Under oxidised conditions, this additional iron is able to bind phosphorus.

Knowledge of these processes helps to anticipate the process determining the phosphorus concentration in the water. In Lake Gross-Glienicker, analyses of the water had been

Figure 2.
Volume-weighted concentration (means: n = 4) of total phosphorus in Lake Gross-Glienicker. A. distinct values, B. successive calculated yearly pattern (cumulative mean).



started more than three years before any effective measures to redevelop the lake were undertaken. During the restoration procedure, activities such as mixing the water by aeration and phosphorus inactivation by the addition of iron led to significant changes in concentrations of all chemical compounds involved in these processes. The assessment of how these processes altered during the restoration can only be done with a knowledge of the processes before restoration.

In temperate lakes, one of the main processes causing phosphorus-related problems is phosphorus net-release from the sediment during the vegetation period. In Lake Tegeler, in Berlin, for example, phosphorus release can be demonstrated by the phosphorus-balance obtained by measuring phosphorus input, output, and its content in the lake. In Lake Gross-Glienicker, there was no measurement or estimation of inflows and outflows until now, and therefore, a balance of nutrients could not be calculated. However as it was assumed that Lake Gross-Glienicker had no significant outflow of water during summer, some conclusions about phosphorus processes were made knowing the concentration pattern of phosphorus in the water.

Phosphorus showed decreasing concentrations between March and August in the years

1989–1992 (Figure 2). This means that, under the given circumstances, sedimentation was the main phosphorus process during summer. In comparison to other lakes, this untypical pattern of Lake Gross-Glienicker was caused by the lack of iron in the sediment. The increase of total phosphorus during the cold season from about September to March was probably caused by mineralisation of organic sediment and inflow of phosphorus-rich water via surface or ground waters.

Redevelopment and restoration of the lake

The aim of restoration of Lake Gross-Glienicker, situated in the densely populated area of Berlin, was to use the lake for recreation, bathing and fishing.

The concept for restoration and redevelopment of the lake was based on the analysis of the processes determining its nutrient budget. Two principles had to be respected in order to restore the lake and its catchment area:

- to keep water in the soil thereby achieving a maximum evapotranspiration (short water cycle; see Chapter 3), and
- to keep minerals and nutrients in particulate or solid structures, i.e. soil, terrestrial vegetation, emersed and submersed macrophytes and sediments.

The basic prerequisite for the redevelopment of the lake was the diversion of sewage. The sewage from the military barracks situated north of Lake Gross-Glienicker was diverted prior to restoration measures in the lake itself being undertaken. In subsequent years, soakaways in the settlement area should be replaced by central sewage discharge into a biological sewage plant. At the same time, the seepage of groundwater into the lake should be minimised. This can be performed by decreasing the rate of groundwater formation by increasing evapotranspiration with the aid of increased vegetation cover. Raising the water level in the lake as high as possible also ensures that the seepage of water from the lake to the soil increases, thereby minimising nutrient load to the lake by that path.

Hypolimnetic aeration

During the previous years, when the lake was heavily loaded with nutrients, a large amount of algae was produced which died and settled on the bottom of the lake. During summer, when the hypolimnion was anaerobic, no oxidative degradation of the organic matter occurred. Therefore, a lot of easily degradable organic substances accumulated in the sediments. Even today, these sediments exhibit a very high capacity for oxygen consumption, which also influences the water body. This oxygen consumption can be controlled by the hypolimnetic aerators.

The first step in the restoration of the lake itself was the installation of four hypolimnetic

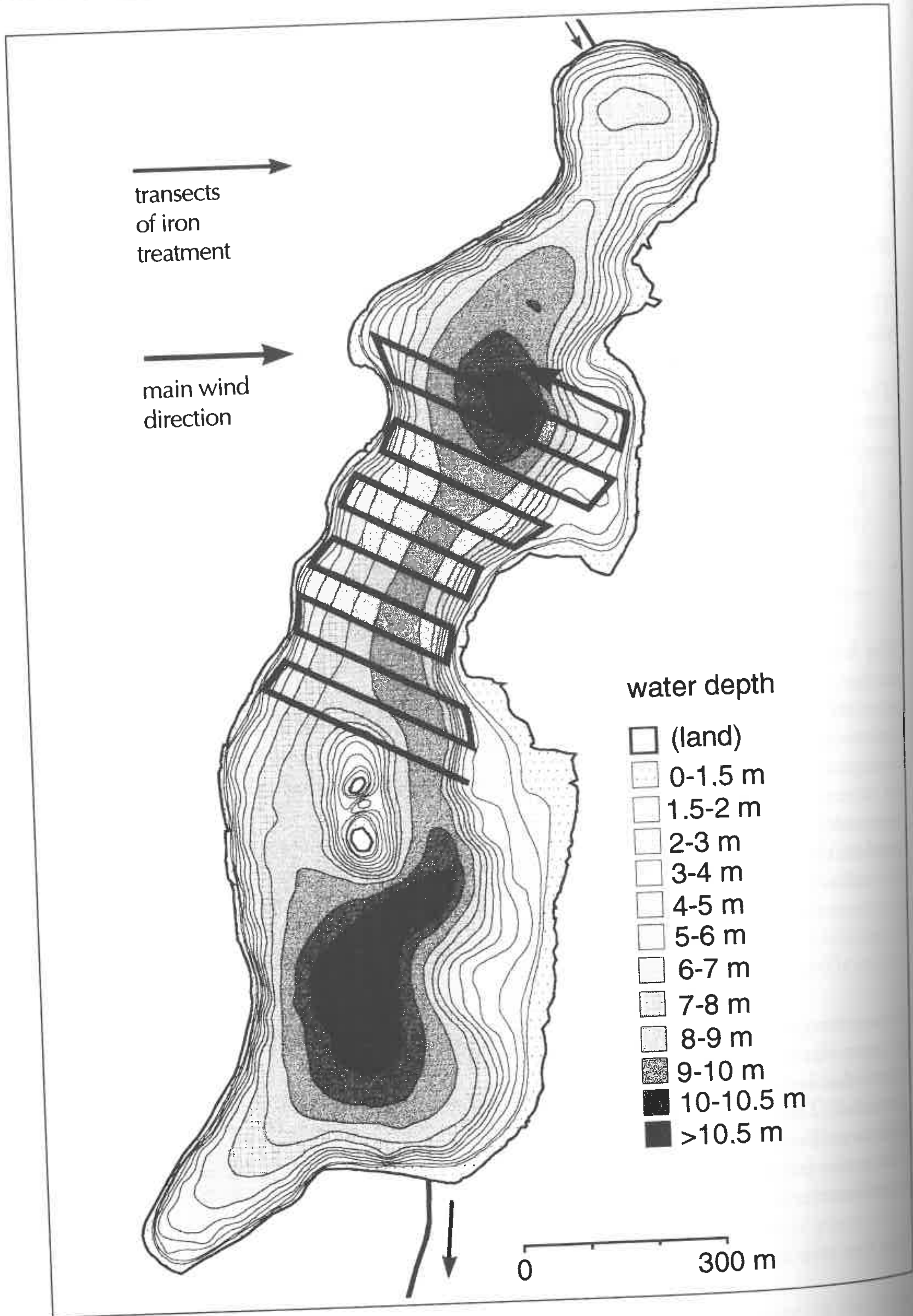


Figure 3. Map of lake depth. Arrow in the lake area shows transects during iron treatment.

aerators. The main function of the aerators was to increase the oxygen concentration in the hypolimnion. In addition, aerators also serve as equipment for controlling the water flow at the sediment water interface (see Chapter 7, Aeration). Oxidising agents are brought to the sediment surface. These oxidising agents can mainly be oxygen or naturally occurring nitrate. Aerators are controlling and enhancing the supplies of these oxidising agents and thus increasing the binding of phosphorus, especially to trivalent iron.

Iron treatment

As a second step, iron in the form of solid iron hydroxy-oxide and dissolved iron chloride was added to the lake water, in order to enhance the phosphorus binding capacity still further. The iron treatment was planned to be carried out during the autumn circulation. Because of several delays it was carried out from December 1992 until February 1993. General recommendations are made in Chapter 7, Phosphorus precipitation.

In both cases (iron hydroxy-oxide and iron chloride), the area to be treated by the addition of iron was restricted to the deeper parts of the lake. Iron deposited in shallow parts of the lake (erosion zone) would be transported to deeper parts during turbulent phases. In Lake Gross-Glienicker the limit of sedimentation lies between 3 and 4 m. Thus, only areas with water depths of more than 4 m were treated. Both preparations were distributed more or less equally over this area. The transects on the lake driven by the boat are shown in Figure 3.

Application of iron hydroxy-oxide

Iron is common in groundwater. In waterworks that use groundwater as a drinking water supply, iron hydroxy-oxide is formed during the water conditioning. This material can be added to lakes in order to increase their phosphorus-binding capacity. However, it has to be tested for phosphorus content prior to application as it may contain precipitated phosphorus. The molar quotient Fe/P of the iron hydroxy-oxide should be 20, but if necessary no lower than 10. The iron hydroxy-oxide is a wet, fine grained material which can be suspended in water and pumped to the lake. It forms an iron buffer inside the sediment. In Lake Gross-Glienicker 250 g Fe m^{-2} of lake area were used.

Application of iron chloride

Iron chloride is a very acid solution and has to be treated with care. In this case, another 250 g Fe m^{-2} of lake area were used. The pH of the 40% iron chloride solution is below one. After dilution with water, the Fe^{3+} ions are precipitated as oxides and hydroxides. This process releases additional protons by the hydrolysis of water. The protons have to be buffered by alkalinity. In Lake Gross-Glienicker, the addition of 112 mg l^{-1} Fe Cl_3 led to a drop of alkalinity from 1.96 to 0.36 mM (Figure 4). The pH of the water during the treatment with iron chloride reached a minimum of 6.26 (Figure 4).

With the iron treatment of the water body, i.e. precipitation of iron as oxides and

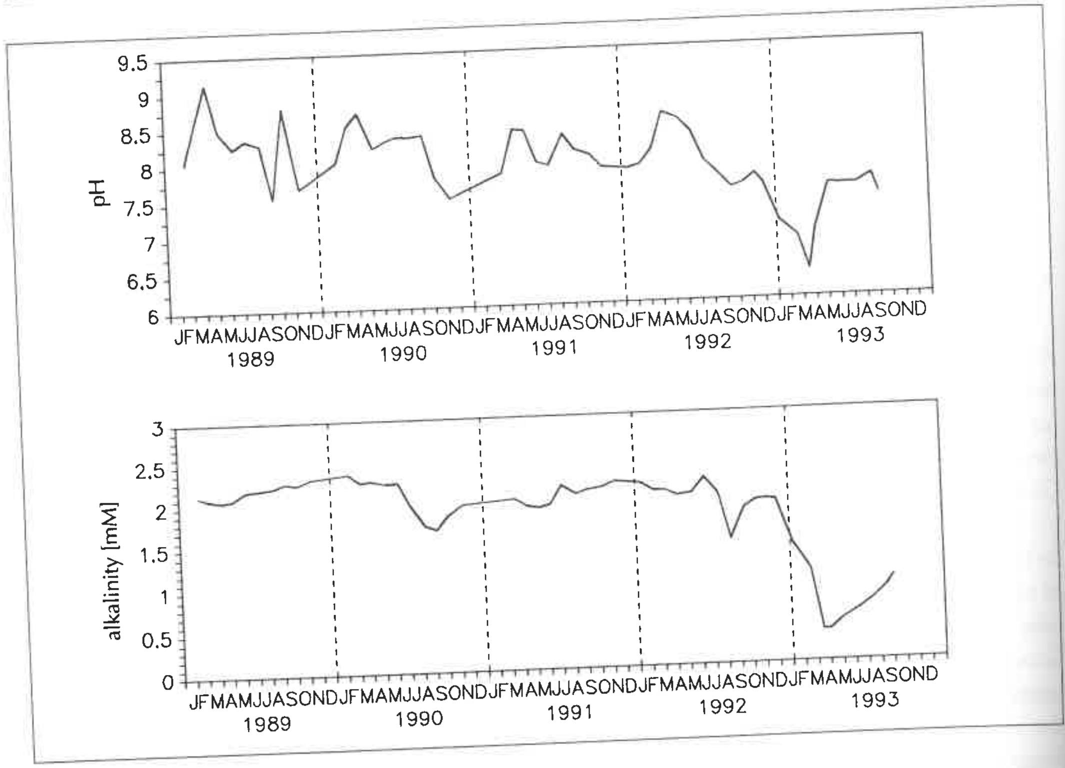


Figure 4. Chemical parameters (pH, alkalinity) – volume – weighted means.

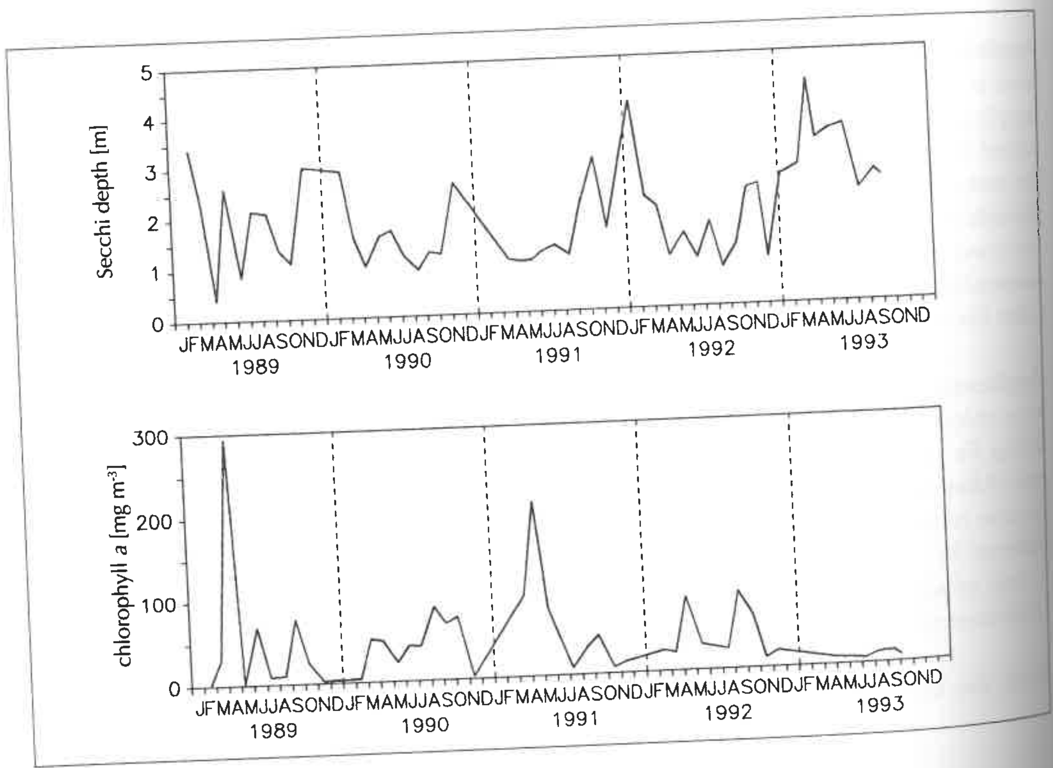


Figure 5. Physical and chemical parameters (Secchi depth, chlorophyll a) – volume – weighted means.

hydroxides and a subsequent binding of phosphorus to the sedimenting iron particles, the phosphorus concentration in the water decreased during the treatment phase and afterwards from about 0.5 mg P l^{-1} to 0.012 mg P l^{-1} (Figure 2). In 1993, the transparency of the water (Secchi depth) increased to values above 2 m during the summer. Due to diminished algal growth the chlorophyll *a* content of the water decreased to a mean value of 10 mg m^{-3} in the vegetation season 1993 (Figure 5).

Importance of the littoral zone and recommendations

In lakes with a long retention time, the littoral zone may have an effect on the nutrient flow in the lake. In Lake Gross-Glienicker, for example, phosphorus from the collection hollows may enter the lake. An additional release of phosphorus from the littoral sands can be caused by people bathing and stirring the sand with their feet. In one sample, a phosphorus concentration of 0.5 mg P l^{-1} was found in the interstitial water of the sand only 35 cm below the surface.

To prevent nutrient input by stirring of littoral sands by bathers, several recommendations were made: (i) artificial islands to be built to offer bathing facilities without disturbing the sand, (ii) a trail to be built near the shore line to allow the observation of littoral biotopes without damaging the sensitive macrophytes. These two measures are schematically drawn in Figure 6.

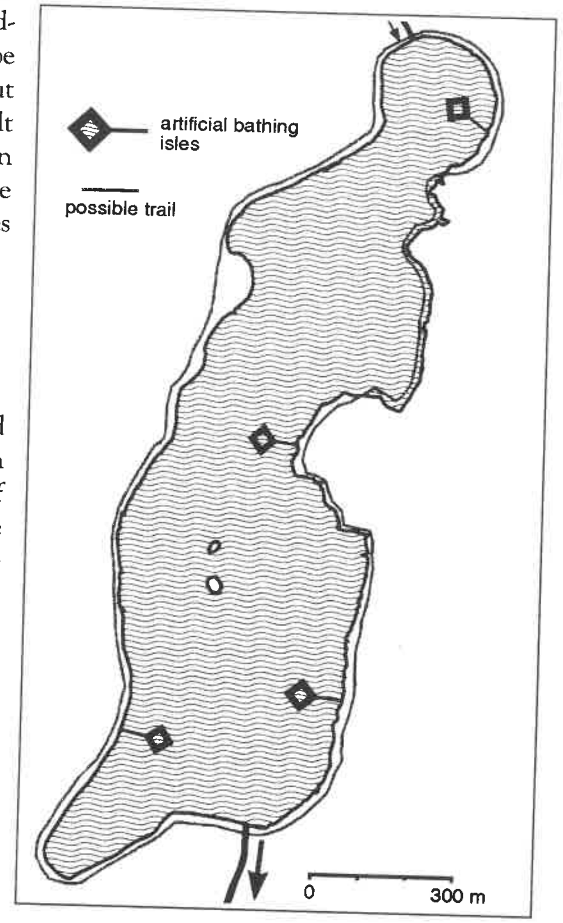


Figure 6. Proposal for artificial islands and 'observation' trail near the shore in Lake Gross-Glienicker.

Project evaluation

Until the end of 1993, the treatment seemed to be successful. During the whole vegetation period of 1993, low concentrations of phosphorus and chlorophyll *a* were maintained (Figures 2 and 5). During the following years the development of the lake will be observed and the following parameters monitored:

- hydrological parameters (inflow and outflow);
- development of stratification (oxygen and temperature profiles);
- chemical parameters (especially phosphorus and chloride); and

- biological response of the lake to decreased nutrient availability (phyto-plankton and littoral communities). Observations will be made every two weeks during the growing season.

We hope that in a few years the use of the hypolimnetic aerators will no longer be necessary and lake processes will stabilise by themselves.

Aeration of the hypolimnion as a tool for restoring eutrophic lakes

Bo Verner

Background

Oxygen conditions in lakes, especially in stratified lakes and reservoirs in a temperate climate, are highly dependant on morphometric features, such as the relation between epilimnetic and hypolimnetic volume, and the trophic status of the lake. The trophic status of many lakes has been altered through pollution by organic matter or nutrients, mainly phosphorus. Similarly, in man-made reservoirs, flooded organic material causes an increased oxygen demand.

Production processes in lakes lead to supersaturation of oxygen in the epilimnion and successively to equilibration through the water surface with the atmosphere. However, the sedimentation of the algae to the hypolimnion reduces oxygen in this strata by respiration processes. If the oxygen in a limited hypolimnion becomes depleted, serious anoxic conditions arise. Fermentation processes reduce both inorganic material, such as iron, manganese, nitrogen compounds, sulphate, and to some extent suitable organic matter to form methane. Higher living organisms cannot survive in such anoxic environments. The phosphorus binding capacities of iron compounds in the sediments become depleted, since iron is either dissolved to a larger extent in the divalent state or transformed to sulphide, thus releasing large amounts of phosphorus to the hypolimnetic water.

There have been several case studies where vertical mixing of the water column was carried out to avoid the adverse effects of anoxia. However, these mostly led to intensified production processes due to increased water temperature, a larger epilimnion, and fertilisation due to the direct contact of the productive layer with the sediments.

By the 1940s, some successful attempts had already been made to aerate the hypolimnion without disturbing the thermal stratification. Since then, aeration of the hypolimnion has been carried out in numerous lakes and reservoirs and is now frequently used for water management.

From a limnological point of view, aeration of the hypolimnion is the preferable tool for increasing breakdown efficiency in stratified lakes or reservoirs which are loaded with either organic matter or nutrients. The redox conditions obtained by aeration keep the sediment surface oxidised and prevent the recycling of nutrients from the sediments to the water. The limited oxidised layer of the sediment surface seals the sediment against transfer of nutrients through the interface and increases the capacity of phosphorus adsorption by converting iron sulphide in the sediments to iron hydroxides.

The construction of artificial reservoirs, for drinking-water supply or for the production of hydroelectric power, usually implies the inundation of topsoils rich in organic material thus demanding oxygen. The lack of oxygen reduces the redox conditions in these water bodies. Divalent iron and manganese are dissolved, in some cases even hydrogen sulphide is produced. The occurrence of these corrosive and toxic compounds degrades water quality with respect to network corrosion and increases the preparation cost for drinking-water. Hypolimnetic aeration is usually the chosen method of treatment in these cases.

Natural lakes

Hypolimnetic aeration has now been in use for several years and sufficient experience has been gained to evaluate the efficiency, benefits and costs of the various applications.

The first projects were undertaken with the Atlas Copco hypolimnetic aerator LIMNO (Figure 1) in highly eutrophic lakes which had received wastewater over a long period. The aim with the aeration was thus to improve their functioning and to avoid the worst inconvenience of stinking hydrogen sulphide.

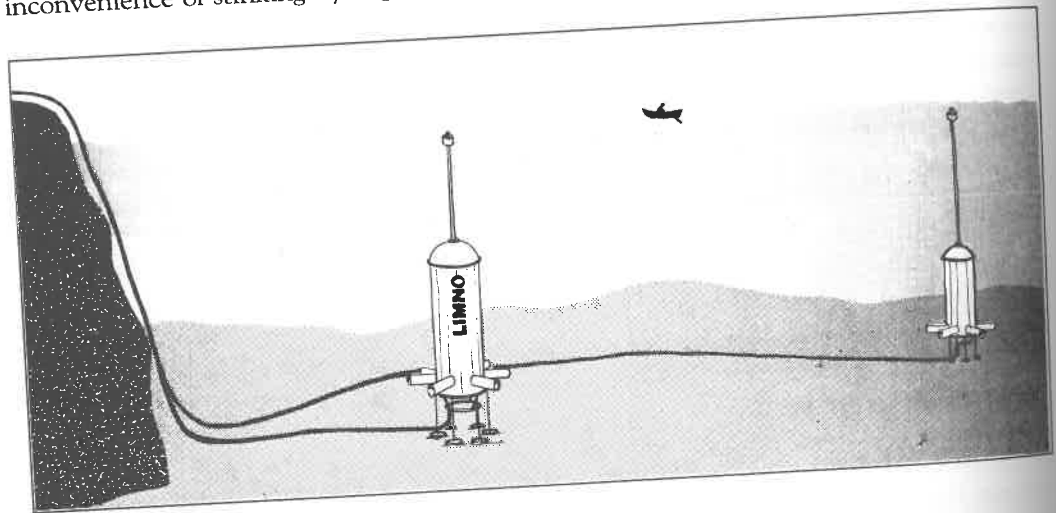


Figure 1. Installation of LIMNO aerators in highly eutrophic lakes to supply oxygen to the hypolimnion.

Lake Grebin, West Germany

The first LIMNO installation in 1971 was done at this eutrophic lake, which is about 1 km² in area with a maximum depth of 16 m. The LIMNO installation was part of a one year research project by Ohle. In addition, bentonit was added to the aerated water for adsorption and precipitation of phosphorus.

During the LIMNO aeration the oxygen depletion was successfully counteracted, and an oxygen saturation value above 50 % was achieved in the hypolimnion during the summer stagnation period. The phosphorus concentration in the water was considerably reduced. However, these effects were not permanent. The following year, without the use of LIMNO aeration, the situation was back to the previous oxygen depletion with the formation of hydrogen sulphide and high release of phosphorus from the sediment.

Lake Brunnsviken, Stockholm, Sweden

In 1972, four LIMNO units were installed in this lake which has a volume of about 10 Mm^3, an area of 1.5 km^2 and a maximum depth of 14 m. The situation in the lake had not improved several years after the diversion of the wastewater. Hydrogen sulphide concentrations of over 20 g m^{-3} were registered during both the winter and summer stagnation periods, and the offensive smell, when more than 20 tonnes of hydrogen sulphide were released was probably injurious to health. During eight years, 775 kg of oxygen was supplied to the lake for six months per year. In this way a positive oxygen balance was maintained each season.

In 1980, the lake outlet was widened and deepened resulting in an increased water exchange with the sea. Since then, a hypolimnetic draw-off has been built to increase the successive dilution of nutrients in the hypolimnion. As the compensating inflowing water has not been of the expected quality, the results of dilution were lower than expected. A combination LIMNO aeration and draw-off may be the final solution.

Lake Kolbotn, Oslo, Norway

This lake is 0.8 km^2 in area and has a maximum depth of 24 m. One LIMNO unit was installed in 1973 and has maintained aerobic conditions in the hypolimnion since then. In 1981, the unit came up to the water surface as the anchoring line broke off. Immediately, the situation reverted to that before the aeration started, with anaerobic conditions and high concentrations of hydrogen sulphide. The aerator unit was put back into service to continue assisting the receiver function.

Lago di Caldonazzo, Trento, Italy

This lake served for many years as a receiver of wastewater. The lake is 7 km^2 and has a maximum depth of 50 m. In the 1960s, the wastewater was diverted by means of ring-canalisation. However, the lake had been irrevocably damaged and did not recover by itself. Hypolimnetic aeration by means of six LIMNO units, adding 2 tonnes of oxygen per day to the water, was applied in 1974 to try to improve the water quality, which is so important for tourism.

In spite of the ring-canalisation, the external nutrient load to the lake is still high due to diffuse leakage into the lake. The aeration plant, therefore, marginally prevents anoxic conditions. This is, however, a considerable improvement compared to the situation before the aeration, when high concentrations of hydrogen sulphide were formed.

Lake Wacabuc, New York, USA

A research project, using LIMNO, was carried out in this lake in 1973 by Union Carbide Corporation. Lake Wacabuc has an area of 0.7 km² and a maximum depth of 13 m and is subject to eutrophication. Two units were installed to counteract the oxygen depletion. The lake home owner association has since then successfully operated the aeration system for more than 10 years avoiding anoxia development in the hypolimnion and permitting a quality, cold water fishing.

Lake Wessling, Austria

Another research project on hypolimnetic aeration using LIMNO was carried out by the Bavarian authorities. The lake area is about 0.5 km² and the maximum depth is 12 m. The aerator was installed in 1981 and run to obtain a dissolved oxygen concentration of not less than 6 mg l^{-1} in the hypolimnion. The immediate results were good, but the authorities are looking more on the long-term effects which will have to include extensive reduction of external load. This project will serve as a pilot study for further restoration plans of the Bavarian lakes.

Lake Södra Hörken, Sweden

Results from this 9.1 km² lake illustrate some typical effects of regulating the oxygen concentration by LIMNO aeration (data from Ahlgren, Institute of Limnology, Uppsala University, Sweden).

Lake Södra Hörken (Figure 2) was originally an oligotrophic lake in which sewage and industrial water from a mining industry was discharged into the northern bay, Grängesbergsviken. The bay is divided into basins of which the two innermost are 19 m and 14 m deep, respectively. They are separated by a ridge where the depth is only 6 m. The hypolimnetic parts of the basins are, therefore, separated from each other. An outflow of polluting water into the innermost, deepest basin of the bay, over a long period, greatly increased the concentration of phosphorus and nitrogen. The hypolimnetic oxygen deficiency was total, and heavy production of algae caused environmental problems. The main part of the external loading has been removed in recent years, but the internal loading is considerable and implies a threat to the lake as a whole.

Thanks to the morphology of the bay in which wastewater has been discharged, this acts as a buffer protecting the main water body of Lake Södra Hörken from direct pollution. The first deep basin, Djupudden, is the most severely polluted, affecting the second basin, Sandudden. The hypolimnion of Sandudden basin has also successively become anoxic and enriched in phosphorus and nitrogen.

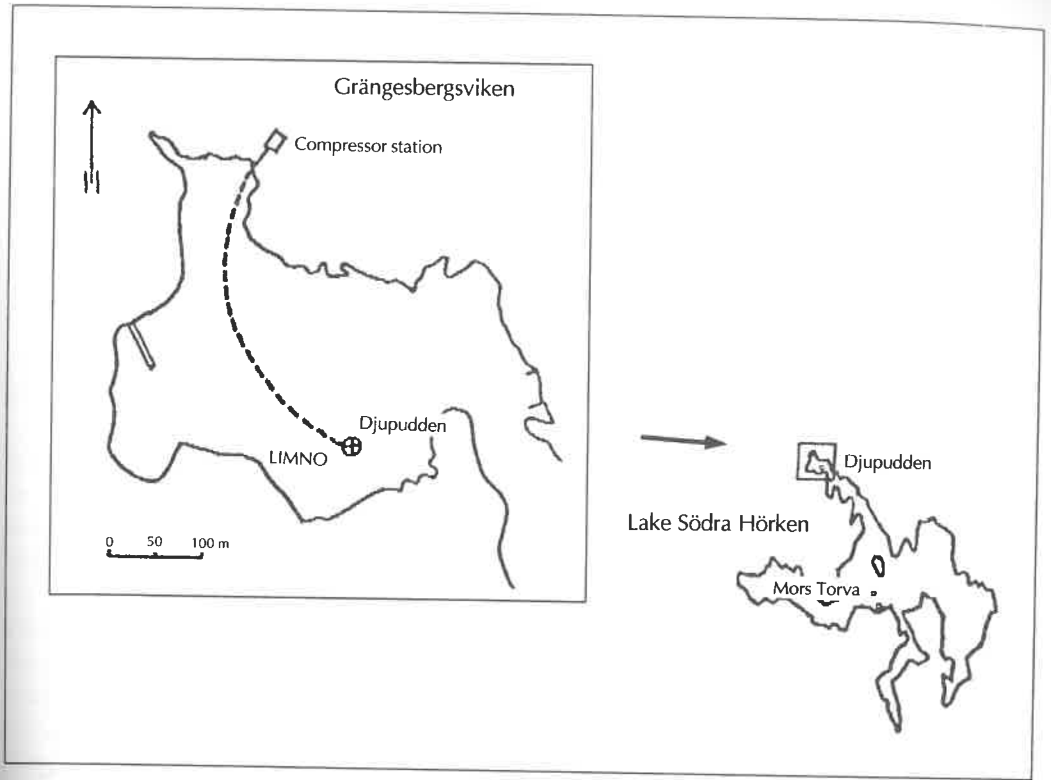


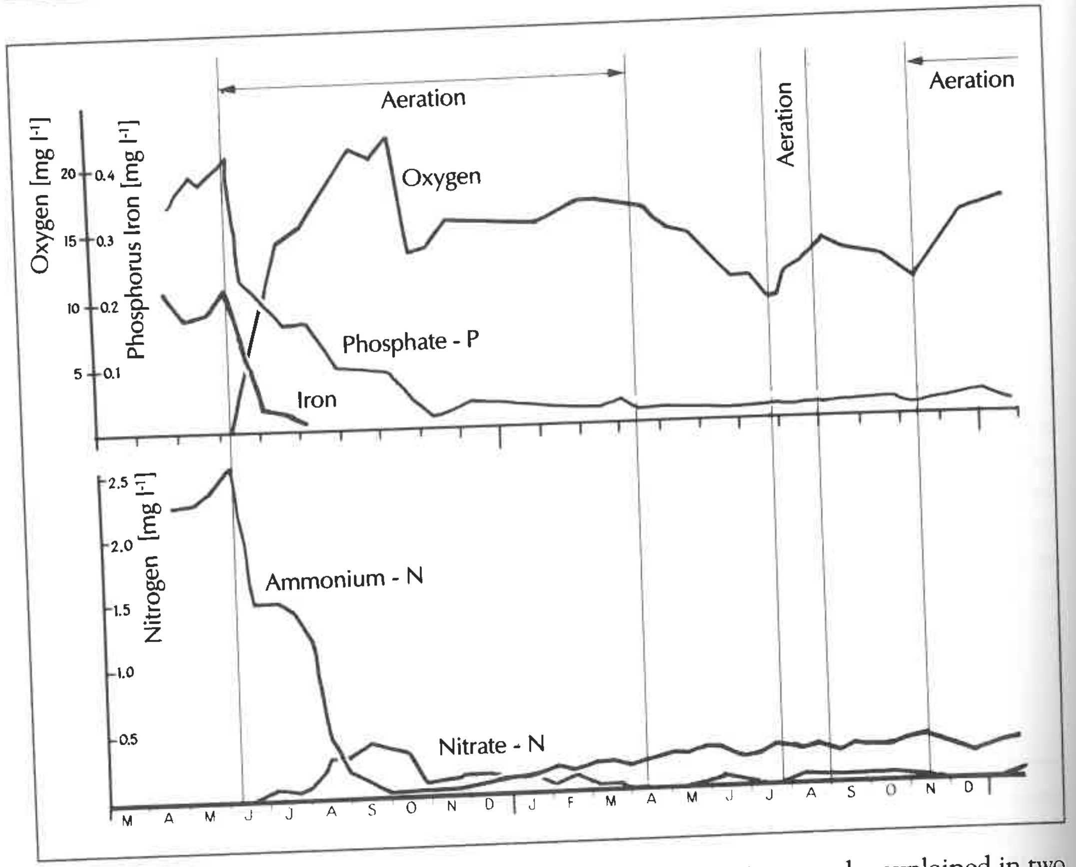
Figure 2. Lake Södra Hörken.

As it is of vital importance to protect the still oligotrophic main basin of Lake Södra Hörken, offering valuable environmental qualities, from degradation (total phosphorus concentration about 10 µg l^{-1}), a LIMNO hypolimnetic aerator was installed at Djupudden at a depth of 16 m. The size and morphology of the basin makes the effect of one single unit very efficient. The aerator maintained the hypolimnetic oxygen concentration of 15–20 mg l^{-1} with an air supply of 4 m^3 min^{-1}. It is therefore possible to reduce the amount of air to 1 m^3 and to run the aerator at intervals determined by the hypolimnetic need of oxygen to prevent the release of phosphorus from the sediment.

When the aerator was started, there was an immediate drop in total phosphorus from more than 0.4 to 0.05–0.025 mg l^{-1}. Immediately after the start, the phosphate concentration dropped very fast, at the same time as the concentration of iron was rapidly reduced. This initial reduction is supposed to be caused by the precipitation of ferric hydroxide with adsorbed phosphate. The slower decrease in phosphate concentration following the first rapid one probably depends on adsorption to the successively oxidised sediment surface.

The aeration also caused a drop in inorganic nitrogen from more than 2.0 to less than 0.3 mg l^{-1} (Figure 3). Before aeration started, nitrogen mainly occurred as ammonium ion in the hypolimnion. Aeration brought about a rapid reduction in the concentration of ammonium and a synchronous increase in that of nitrate. The reason why the increase in

Figure 3.
Lake Södra Hörken. Concentrations of oxygen, iron, phosphate-phosphorus, ammonium and nitrate-nitrogen at Djupudden (mean values for 12.5, 15 and 17.5m).



nitrate was considerably smaller than the reduction in ammonium can be explained in two ways: ammonium might have been adsorbed to the sediment surface, and nitrate may have been reduced through denitrification to molecular nitrogen.

Ahlgren is of the opinion that the first fast reduction was caused by chemical adsorption while the later, somewhat slower decrease was dependent on bacteria-mediated nitrification-denitrification processes (*cf.* Sediment treatment, Chapter 7). In the Djupudden basin, the biomass of algae and the chlorophyll content has decreased and a synchronous increase of transparency has been recorded. The relations between nutrient concentration and plankton development will be elucidated in further studies being carried out by Ahlgren.

The stratification in the Djupudden (innermost) basin is characterised by a high degree of stability because an ore flotation plant has been discharging its electrolyte-rich wastewater directly into the hypolimnion. However, water enriched with nutrients from the anoxic hypolimnion of this basin has successively passed over to the second basin. Thanks to the hypolimnetic aeration of the first basin, phosphorus is now largely trapped already in this part of the lake. Thereby the main water body is protected. In order to make the protection of Lake Södra Hörken still more effective, a second barrier against lakeward transport of phosphorus has been established. This has been done by the installation of a second LIMNO aerator, in the Sandudden basin.

This project shows how hypolimnetic aeration can be a useful tool to achieve:

- mineralisation of organic matter
- precipitation of phosphorus
- oxidation of the sediment surface and its recolonisation by bottom animals
- optimum conditions for fish to survive in the hypolimnion

Lake Tegel, Berlin, Germany

As a result of a high nutrient load this lake became eutrophic a long time ago with all the distinctive features: algal blooms, hypolimnetic oxygen depletion, fish-kills, and the formation of sapropel at the lake bottom.

Lake Tegel has a surface area of about 4 km^2, a volume of 32 Mm^3 and a maximum depth of 16 m. Situated in the centre of the metropolis it was once an outstanding recreational asset and is still used by many people for boating, fishing and swimming in spite of the deteriorated water quality.

However, the future of the lake is now secured, as restoration is well under way. In the near future, an advanced wastewater treatment plant including phosphorus elimination will enter into service.

To speed up this restoration, considering the high internal nutrient circulation in the former receiver, a LIMNO hypolimnetic aeration plant was installed in 1980. The system comprises 15 LIMNO units supplied, with an airflow of 1.07 m^3 s^{-1} at about 2.0 bar, by two Atlas Copco oil-free screw compressors mounted in a container placed at the shoreline. The compressors require 185 kW for an oxygenation capacity of 4500 kg per day. The installation is operated about 270 days per year. The total project costs were about US$ 1 m.

The aeration system has fully achieved the specified goal: a dissolved oxygen concentration higher than 4 mg l^{-1}. However, the reduction of nutrients due to fixation of phosphorus to the sediment has not yet been achieved. This was due to an underestimation of the oxygen consumption of the sediment due to higher external load, on which the size of the installation was decided. The Authority of Fisheries reports the return of salmonid fishes and crayfish to the lake and the recolonisation of the bottom sediment by chironomids assisting in the decomposition and mineralisation of the sediment.

Reservoirs

Another very important application of the hypolimnetic aeration is in drinking-water reservoirs. A high oxygen concentration of the raw water in the reservoir suppresses the releases of iron and manganese causing taste, colour and odour problems which are so difficult and costly to eliminate in the preparation plant. Considering the recent concern about the formation of trihalomethanes, a high oxygen concentration is equally important to be able to lower the chlorination demand. Below, some data are presented from three drinking-water reservoirs where hypolimnetic aeration, by means of the LIMNO method, has been applied for several years.

The Ennepetal reservoir, Hagen, Germany

This lake has an area of about 1 km^2, a maximum depth of 25 m and a volume of 10.6 Mm^3 and serves as a drinking-water reservoir. 23,000 m^3 of drinking-water are delivered a day or 8.4 million m^3 a year. In addition to the drinking-water at least 50,000 m^3 of water per day is necessary to ensure the recipient function for the little stream below the reservoir. The catchment area of 48 km^2 provides about 45 Mm^3 water per year. The water renewal time is two to three months.

The oxygen conditions in the Ennepetal reservoir become critical during summer stratification, since water in the tributary is of poor quality. The catchment area of the Ennepetal reservoir is largely used for farming and a small town (about 12,000 inhabitants) discharges water from a treatment plant to the reservoir. Until 1973, the mean phosphorus concentration entering from the treatment plant was about 10–15 mg l^{-1} or about 6 tonnes of P per year. Phosphorus elimination measures in the treatment plant were able to reduce the amount of phosphorus discharged to about 2.0–2.5 tonnes of P per year. Further phosphorus reductions are planned for the near future and the goal is to decrease the phosphorus discharge below 0.4 mg l^{-1}.

To meet the oxygen problems, the first LIMNO aerator was installed in 1976. The second unit was installed in 1981. At the same time the old piston compressor for the first LIMNO unit was replaced by three rotary screw compressors to be able to operate with different oxygenation capacities for different storage volumes and oxygen demands. The oxygenation capacity can be varied between 360 to 1,330 kg oxygen per day. The effect absorbed by the compressors depends on the water level in the reservoir as the air-pressure required is equal to that of the hydrostatic pressure of the LIMNO air-diffusor plus some very small pressure losses in the air-supply lines. The compressor capacity installed is also used to extend the natural circulation periods.

The alternatives to aeration are dramatically increased treatment costs, since an additional filtration step (active carbon) would be required to eliminate the fermentation products in cases of anoxic conditions in the hypolimnion. The trophic level of the Ennepetal reservoir is so high that sulphate reduction would occur immediately in the case of anoxia. Destratification sometimes used for a short period does not solve the problem, since it increases the productive layer and results in increased biomasses and temperatures unacceptable to users.

Data kindly supplied by the AVU water supply company from 1982-84 show the correlation between the oxygen concentration and the iron and manganese concentration of the reservoir water (Figure 4). During 1982 and 1983, the oxygenation capacity was not quite sufficient to suppress the manganese concentration.

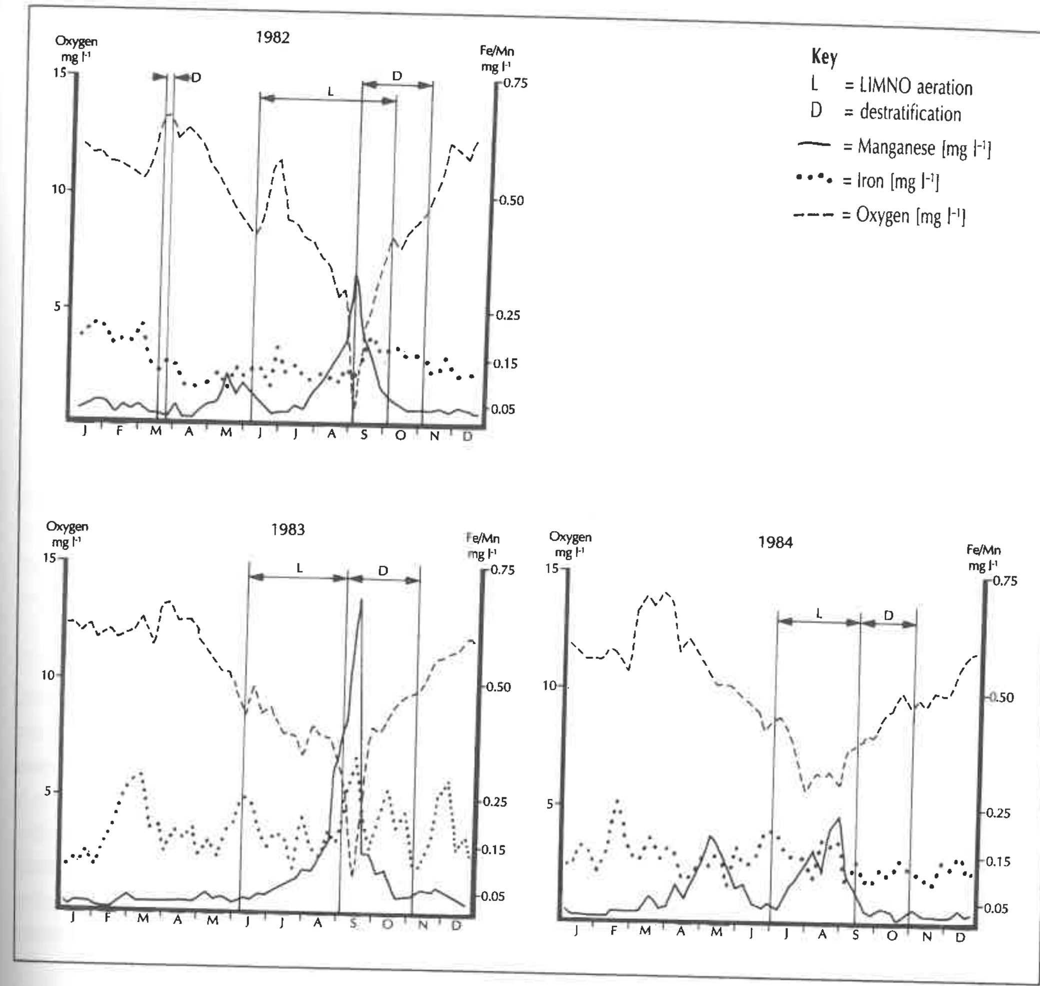


Figure 4. Ennepetal reservoir. Raw water quality in 1982, 1983 and 1984.

The Breitenbachtal reservoir, Siegen, Germany

This reservoir has served as a drinking-water supply since 1956. However, increased demand called for an extension of the storage volume. In 1975–79 the height of the dam was increased by 12.5 m, whereby the storage volume was enlarged from 2.6 to 7.8 Mm^3 with a maximum depth of 37 m. The drainage area of about 11.6 km^2 delivers about 9.6 Mm^3 run-off water to the reservoir and the maximum intake flow for drinking-water preparation is 26,000 m^3 per day.

For the increased storage capacity, hypolimnetic aeration was foreseen, and a LIMNO aerator was installed in August 1979 with an oxygenation capacity of 400 kg per day. The condition in the reservoir is oligotrophic and the aeration system is a preventative measure. The investment cost for the system, the LIMNO unit and the compressor, was US$ 40,000, installed and ready to operate. The record from the operation during 1981 shows that 120,000 kWh were needed for aeration.

A laboratory report for 1981 stated the onset of the summer stagnation at the end of April. Aeration was started on 22 June when the oxygen saturation in the bottom layer had decreased to 41 %. During aeration, mean saturation in the bottom layer was 60 %. The importance of aeration was demonstrated during a short maintenance stop of the compressor at the end of September, when the oxygen saturation decreased within a couple of days from 70 % to 20 % saturation.

During the whole period of aeration, manganese was present in concentrations of 0.1 mg l^{-1}. The incident in September, however, caused an increase of manganese to 0.78 mg l^{-1}. The oxygen demand in the reservoir was caused mainly by intensive production processes due to the mass development of the green algae *Cosmarium* during July and August. Up to 88 million cells per litre were obtained with a chlorophyll content of 75 mg m^{-3}, causing oxygen supersaturation of 185 % and pH values above 10.

Another result of aeration is the nitrification of the ammonia in the hypolimnion. About 1 mg l^{-1} NO_3-N was present in the hypolimnion while ammonia was lacking.

New LIMNO design

In 1982, the LIMNO aerator was redesigned (Figure 5). The earlier design in stiff polyester was changed to a one using flexible, heavy duty, polyester fabric coated in PVC. As no mould is required with this design the units can be manufactured to the customer's required size.

The relatively light and foldable material, with a density of about 1 kg m^{-2}, permits the transport and installation of bigger units than could be handled earlier. The less costly material and manufacturing, make it also favourable to apply lower flow velocities in the unit. These factors increase the contact time between the air and the water and increase the oxygen transfer efficiency.

The major increase in the efficiency of the new LIMNO design is the introduction of the secondary air diffusor, whereby a counterflow transfer is achieved. As this air diffusor is placed just above the water outlets, at the very lower part of the unit, more oxygen can be dissolved at this higher hydrostatic pressure.

For the overall efficiency of the aeration system the proper dimensioning of the regulating system and air supply lines is important. The operating air-pressure is the sum of the hydrostatic pressure at the level of the installed LIMNO plus a small pressure loss in the air supply lines. The transfer efficiency also depends on the specific compressor efficiency which, in turn, depends on the compressor design. The compressed air must be oil-free, which is why non-lubricated compressors are recommended.

Not only the operating cost (expressed as kg oxygen transferred per kWh), but also the capital cost must be considered for an aeration plant. It may be favourable to run at a somewhat lower transfer efficiency to reach a required oxygenation capacity (for details see

Chapter 7: Aeration). In each case, the minimum capital, operating and maintenance costs for the installation have to be calculated.

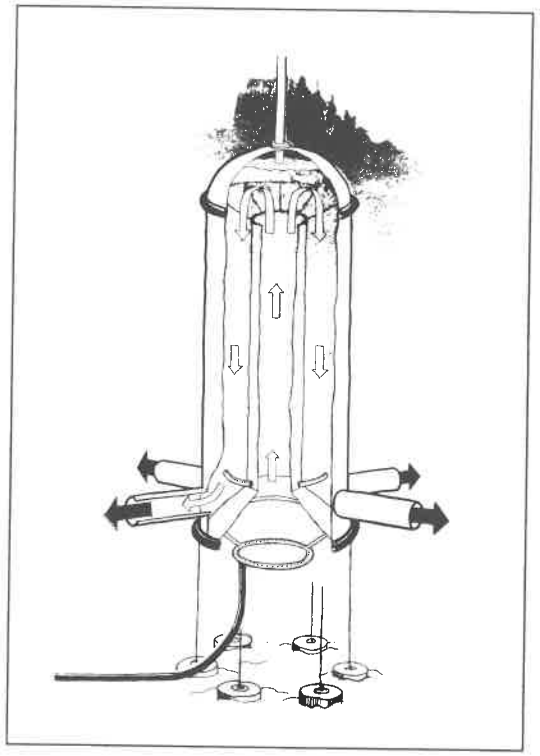


Figure 5.
The new LIMNO design.

Conclusions

The experience with hypolimnetic aeration shows that this method can be used for different purposes, as long as the preconditions are known and the mechanisms clear. General recommendations should always be complemented by special studies of the lake or reservoir in question. The costs for treatment can be classified as moderate in comparison to other measures, such as the use of pure oxygen or ozone.

The argument of lethal effects to fish, due to enrichment of dissolved molecular nitrogen in the water at the hydrostatic pressure in the level of the installed equipment, seems to be hypothetical. We have not observed this effect on fish, which also seems plausible when the movement pattern of fish is considered. Dissolved gas accumulation always occurs at the sediment water interface due to intensified respiration and denitrification processes during stagnation. Fish were observed in echograms in the hypolimnion at several installations, which would not have been the case under prevailing anoxic conditions.

Restoration of lakes through sediment removal – Lake Trummen, Sweden

Sven Björk

General description and type of disturbance

Lake Trummen (56°52'N, 14°50'E) at the town of Växjö in south-central Sweden (Figure 1) was a good example of the large group of polluted lakes which had deteriorated within and around urbanised areas. The originally oligotrophic, brown-water Lake Trummen (drainage area 12 km², lake area 1 km² and depth, until 1970, 2 m – Figure 2) was used for swimming and for water supply, to at least some degree, until the 1920s. Up to 1957–58, it was exploited as a recipient for sewage from the town of Växjö and from 1941 to 1957 for wastewater from a flax factory (an extensive survey concerning the cultural influence on the lake during the 19th and 20th centuries has been compiled by Lettevall (1969, 1977). The pollution became more and more severe and in the 1940s the hypertrophic ecosystem collapsed. After that there were regular fish-kills due to total oxygen depletion in winter and the summer transparency was depressed to 15 to 20 cm by blue-green algae (Cyanobacteria), especially *Microcystis*.

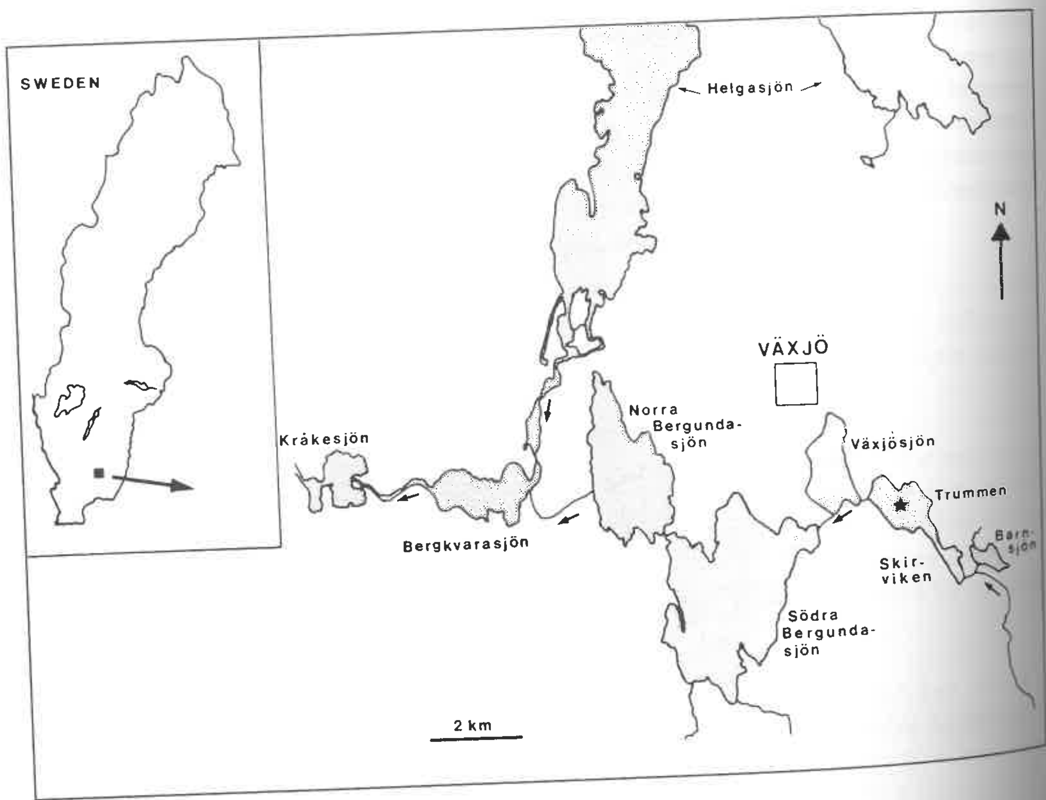


Figure 1. Location of Lake Trummen, Sweden. From Cronberg 1982.

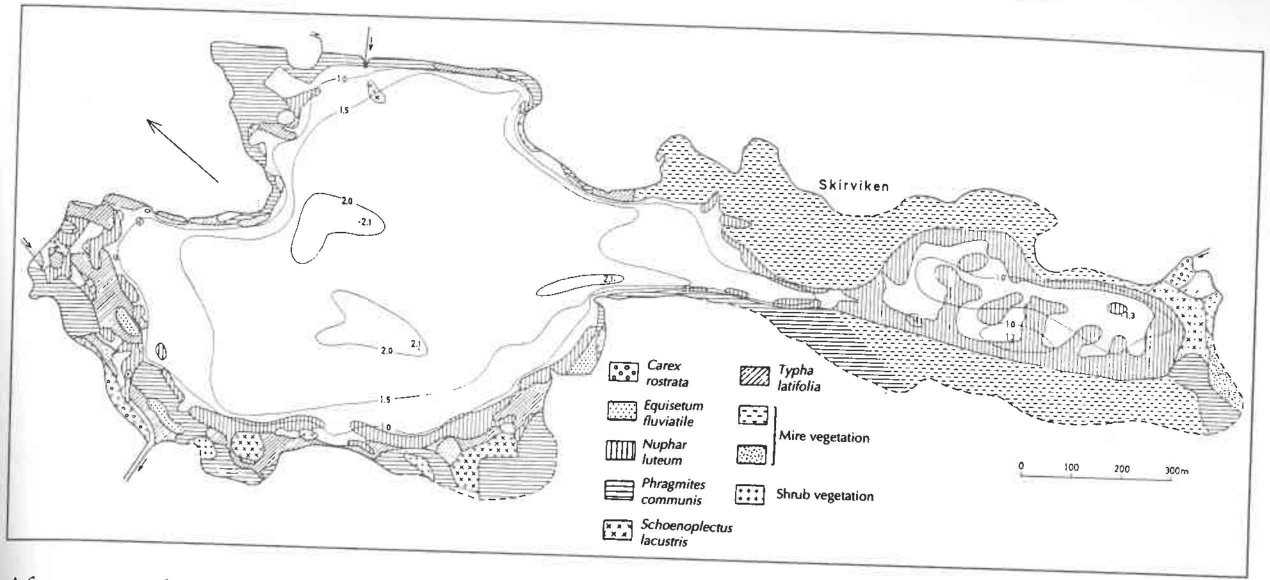


Figure 2. Lake Trummen. Map of vegetation and water depth (in m). From Persson 1969 and Digerfeldt 1972.

After sewage diversion and an end to industrial wastewater discharge in 1957–58, the lake did not recover, a very interesting but not surprising fact to limnologists and a severe problem for the town authorities. About 10 years after the diversion of wastewater from the lake, the summer transparency was still only 15 to 20 cm. At this point, the town authorities definitely had to face the Trummen problem when planning the urbanisation of the surroundings. Of course, the simple, narrow-minded technological solution of filling the lake was considered, but the authorities decided to follow a proposal to restore the lake. Later, the restoration project developed a very successful cooperation between limnologists and palaeolimnologists, other ecologists, politicians, technologists and administrators. The scientific pre-project investigations carried out as a teamwork during 1968–1970 were designed as a typical ecosystem-oriented study.

Pre-project investigations

In the intact, oligotrophic Lake Trummen, the sediment growth rate was between 0.2 and 0.4 mm per year, but because of the very high nutrient supply during the recipient period it increased to 8 mm (Digerfeldt 1972). Therefore, in the 1960s the sediment of Trummen showed a very characteristic stratification including a brown, well-consolidated gyttja overlaid by the 20 to 50 cm of black, loose deposits from the recipient period (Björk & Digerfeldt 1965). The irreversibility of the damage to Lake Trummen was due to the presence of that layer from which nutrients were being released (Bengtsson & Fleischer 1971, Bengtsson *et al.* 1975) and to which the plankton crop was being deposited after every vegetation period. Broad reedbeds, developed as a floating plaur zone, surrounded the lake (Figure 3).

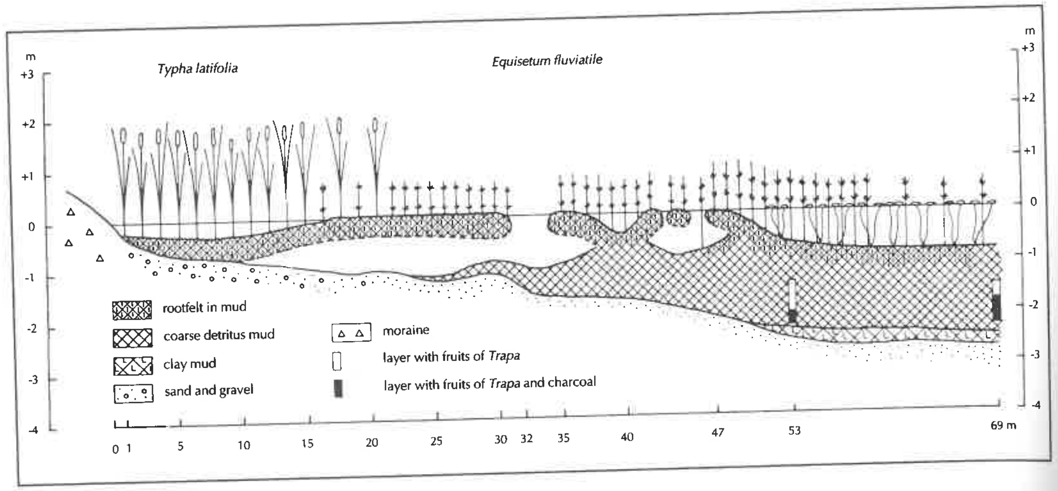


Figure 3. Lake Trummen. Section through a *Typha-Equisetum* plaur area. From Björk & Digerfeldt 1965.

Figure 4. Lake Trummen. Suction dredger. Photo: Sven Björk 1970.

Restoration

In 1970, the topmost half metre of sediment was removed by suction dredging, and in 1971, another half metre was dredged (Figure 4). Altogether about 400,000 m^3 of sediment (gyttja) were removed during nine working summer months. Furthermore, at least 200,000 m^3 of lake water were mixed with the pumped sediment. The limnologists requested a nozzle for dredging which would make it possible to suck in the sediment without making the lake water turbid, and with a minimum intake of lake water.

The sediment was pumped to simple, settling ponds (Figure 5) constructed in an abandoned, farming area from which the topsoil had first been removed. The run-off water from the settling ponds – a mixture of lake and interstitial water – was treated with aluminium sulphate in a simple plant (Figure 6) for precipitation of phosphate and suspended matter. Before restoration, the total phosphorus content of the lake water was about 600 µg l^{-1}. The phosphorus concentration of the water from the settling ponds was of the order of milligrams per litre before the treatment. After the precipitation, the total phosphorus content of the run-off water was about 30 µg l^{-1}.

The pre-project investigations comprised the growth test with the sediment to prove its suitability for agricultural and horticultural purposes. The pumped sediment was, after drying, sold by the town for parks, gardens, roadsides etc. (Figure 7), and the income financed the preparation of green areas around the lake. A separate overgrown bay was left as a waterfowl reservation (an observation tower was constructed on the shore), and an artificial island was built for the birds (Figure 8).

Overall evaluation

As was foreseen, the changes in the lake itself were dramatic (Cronberg 1982). The phosphorus and nitrogen concentrations decreased and the transparency increased (Figure 9). The blue-green blooms disappeared, and a plankton community with a higher diversity replaced the monocultures of *Microcystis*. The population of the freshwater mussel *Anodonta* was wiped out in the hypertrophic lake but recolonised the bottom immediately after dredging. The reappearance of the mussels means that the ecosystem became enriched with their function as efficient seston filtrators. The restoration caused such a change in the structure of the ecosystem that the lake recovered to a functioning unit, characterised by a balance between production and decomposition.

The lake became available for sport fishery, swimming, windsurfing and other forms of recreation within the urbanised area (Figure 10). Thus, only nine summer months were needed to transform the lake from an environmental problem to an environmental asset. The total cost was about 500,000 US$ (1971). The Lake Trummen restoration project was used for the training of professional limnologists and other ecologists (about 20 have been involved). It also served as a demonstration project for converting politicians and administrators to believers in ecology and preservation of nature for creation of sustainable, high quality environmental conditions.

In Trummen, the nutrient-rich sediments had been deposited over more or less the whole lake area (with the exception of the bay preserved as a waterfowl reservation). Therefore, it was necessary to dredge the extensive part of the lake. However, in other lakes, sediments of sewage-sludge character could be concentrated to restricted areas, and the restoration effect could be reached after dredging only part of the lake bottom.

Figure 5.
Lake Trummen. Settling ponds for dredged gytja. Photo: Sven Björk 1970.

Figure 6.
Lake Trummen. Plant for treatment (phosphate precipitation through addition of aluminium sulphate) of run-off water from the settling ponds. Photo: Sven Björk 1970.

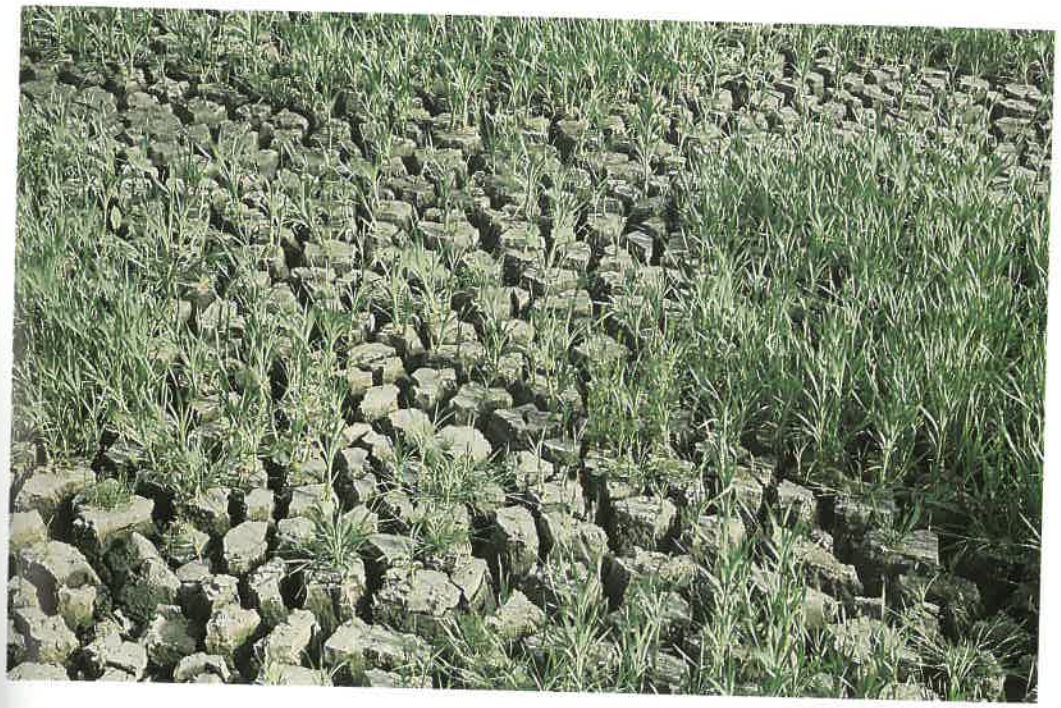


Figure 7.
Lake Trummen. Drying sediment in the settling pond. Photo: Sven Björk 1970.

Figure 8.
Lake Trummen. Artificially constructed island for waterfowl. Photo: Sven Björk 1979.

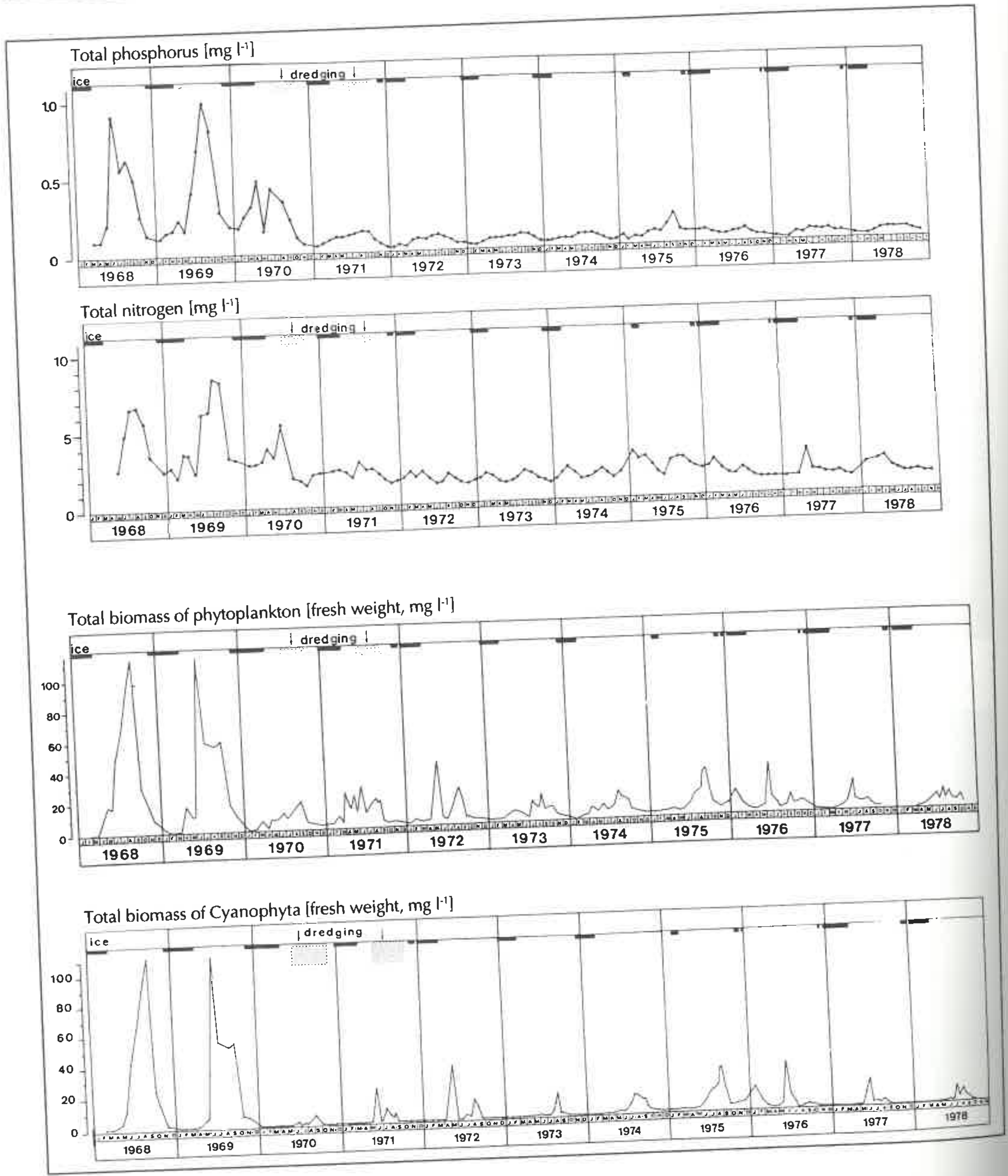


Figure 9. Lake Trummen. Concentrations of phosphorus and nitrogen, total biomass of phytoplankton and total biomass of blue-green algae 1968–1978. From Cronberg 1982.

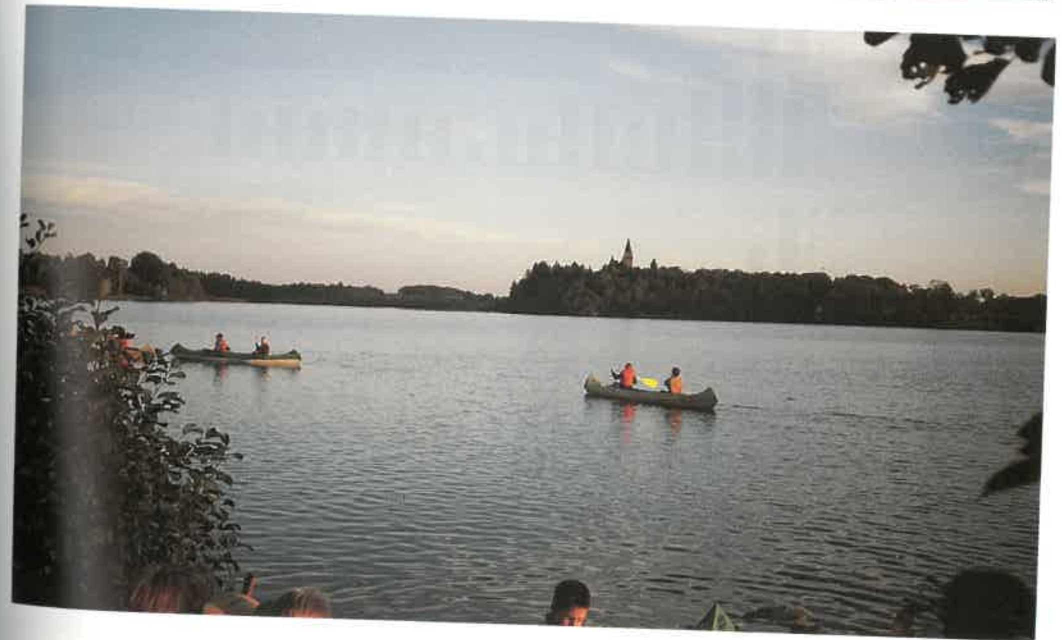
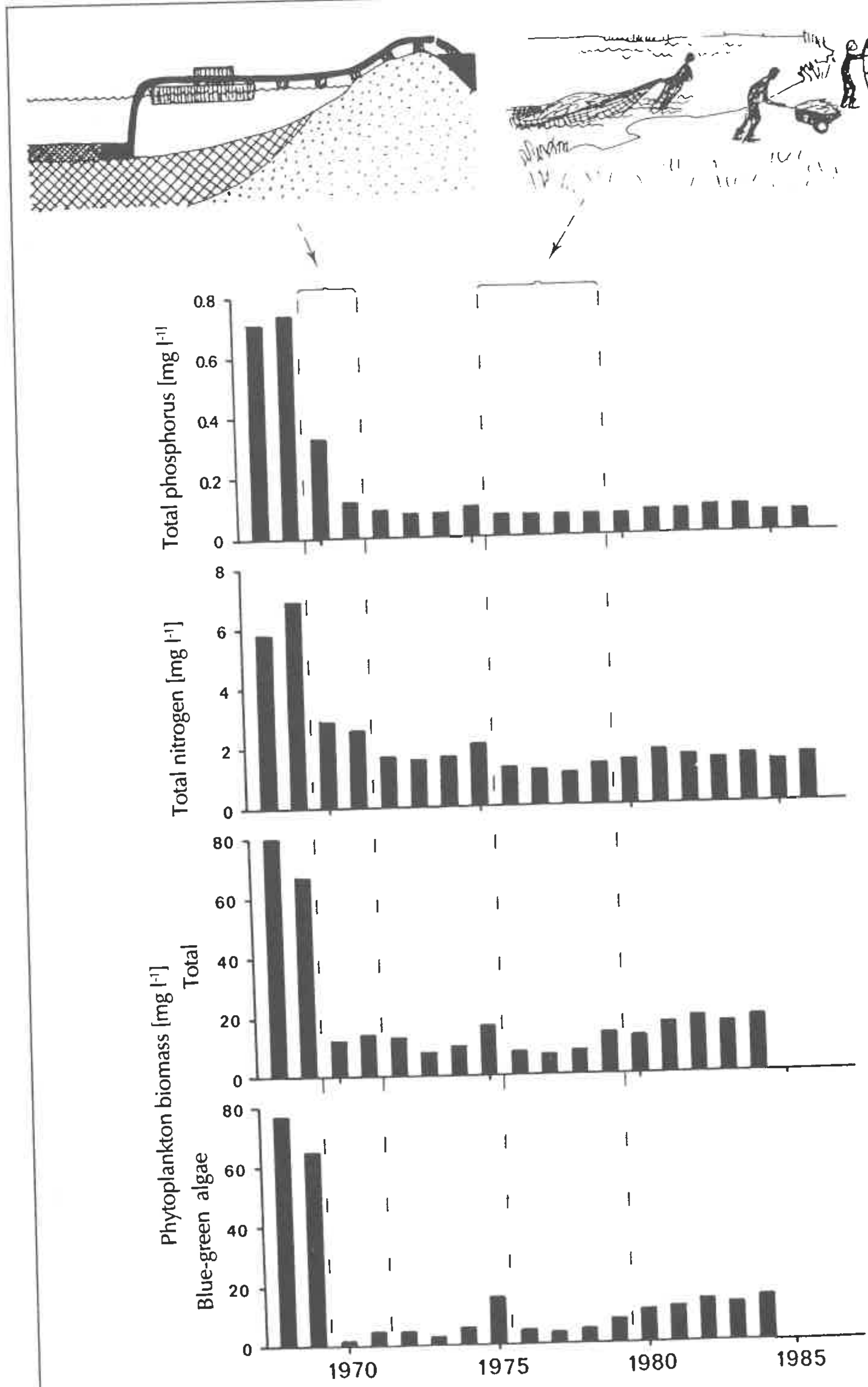


Figure 10. Lake Trummen before (above) and after (below) restoration through sediment-suction-dredging. Photo: Sven Björk 1970 (above) and 1989.

Figure 11. Lake Trummen. Results of nutrient reduction and intensive removal of fish on phosphorus and nitrogen concentrations and on phytoplankton development. From Björk 1988.



Management and monitoring programmes

After the highly significant nutrient reduction had been realised, intensive fishing during the period 1976–79 was carried out to reduce the populations of bream (*Abramis brama*) and roach (*Rutilus rutilus*) by 16 tonnes (190 kg ha^{-1}), while predators, pike (*Esox lucius*) and perch (*Perca fluviatilis*), were left (Andersson *et al.* 1978, Andersson 1985). There is hardly any doubt that a synchronous decrease in total phosphorus and nitrogen concentrations as well as in phytoplankton biomass was caused by this intensive selective fishing. It has, however, not been possible to obtain stability in these respects (Figure 11). The fish reduction procedure must be repeated and such a continuous management programme, in the case of Lake Trummen, has not been applicable as there is a considerable exchange of fish between the lake and its tributaries and outflow including the lakes located upstream and downstream.

In a eutrophicated lake, after reduction of, for example, bream and roach, the eventual appearance of a valuable food resource (cladocerans etc.) will not be left untouched and unexploited. It again opens up the possibilities for the prosperous development of consumers (fish) that will reproduce successfully until their biomass becomes the food resource of a higher consumer or is reduced through laborious fishing. The oscillations might be great in disturbed systems, but the base of the trophic pyramid will always remain broad and the top narrow. If a snapshot of biomass distribution in the open water should happen to show upside-down relations, these are of limited duration only.

It should be stressed, that Lake Trummen was restored by suction dredging *after* sewage and industrial wastewater had been diverted from the lake, and that the external loading could be considered as fairly normalised with respect to regional conditions. However, since the restoration, the surroundings have been urbanised, i.e. the character of the catchment area has changed considerably.

In the limnological management plan for the restored Lake Trummen, it was suggested to make arrangements for continuous and careful control of the water quality in the tributaries. Among other things, construction of basins for the reduction of nutrients, and the protection against oil spill from industrialised sections of the catchment were recommended. One of the tributaries was supplied with such a basin working for a limited period of time. However, generally speaking, the suggested arrangements have not been realised and put into practice.

Changes in the ecosystem caused by the restoration have been followed by a research programme covering the period 1968 to 1980. The lake is, furthermore, included in the routine environmental control programme executed by the local authorities. Ten to fifteen years after the restoration of Lake Trummen, there appeared indications of increased plankton turbidity caused by tiny blue-green algae (among these *Cyanodictyon imperfectum*) decreasing the transparency. In order to revert to the restored stage, it is necessary to reduce the external loading, including nutrient trapping at the mouths of the tributaries and, first of all, by minimising the losses of nutrients from the catchment area.

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Restoration of lakes through sediment removal – Vajgar fish pond, Czech Republic

Jan Pokorný and Václav Hauser

Background

In Central and Eastern Europe fish ponds, constructed mainly in the Middle Ages, are shallow water bodies ranging in size from less than a hectare to several hundred hectares. Situated mostly on sites of former marsh, swamp, bog or fen, or in original floodplains of small watercourses, the fish ponds and their littoral zones have become a specific type of wetlands (Dykyjová & Květ 1978). These fish ponds are an important part of the hydrological system in the landscape, serve as water purification systems and sediment traps, provide habitat for many plant and animal species, serve for recreation, but above all, dedicated for fish production.

The original oligo- or mesotrophic character of these fish ponds has changed to eutrophic and even hypertrophic conditions due to intensified management for fish production, agriculture practices in the catchment area, but also due to other uses of fish ponds such as wastewater recipients.

In the case of fish ponds suffering from continual internal nutrient loading, the removal of nutrient-rich sediment, a method developed for degraded shallow lakes, can be considered to be applied.

Vajgar and its catchment

Vajgar fish pond (*c.* 40 ha) is situated in the district town of Jindřichův Hradec (49°09'N, 15°01'E, altitude 476 m). The fish pond was part of the town fortification since the foundation of the castle. The first written documents about Vajgar fish pond come from the 13th century. The fish pond provided water, fish, acted as a recreational area, and served also as a recipient of the town's waste waters. The historical documentation of the town and Vajgar fish pond are deposited in the town archives and described for example by Teplý (1927).

Fish pond	water area ha	water volume thousands m^3
Zhejral	9.8	161
Karhov	21.9	385
Meziníěský	32.0	660
Komorník	56.3	1700
Hejtman	68.4	2052
Ratmírovský	78.0	1100
Krvavý	127.0	1300
Vajgar	36.0	400

Table 1. Hydrological data for important fish ponds in the Vajgar catchment area. (The data might be overestimated because of the silting-up of the fish ponds).

Until 1970, the town wastewater (BOD_5 200–250 mg l^{-1}, P_{tot} *c.* 5 mg l^{-1}, N_{tot} 50 mg l^{-1}) was discharged into Vajgar fish pond (about 15 l s^{-1}). In 1970, a wastewater treatment plant which treats about 10,900 m^3 per

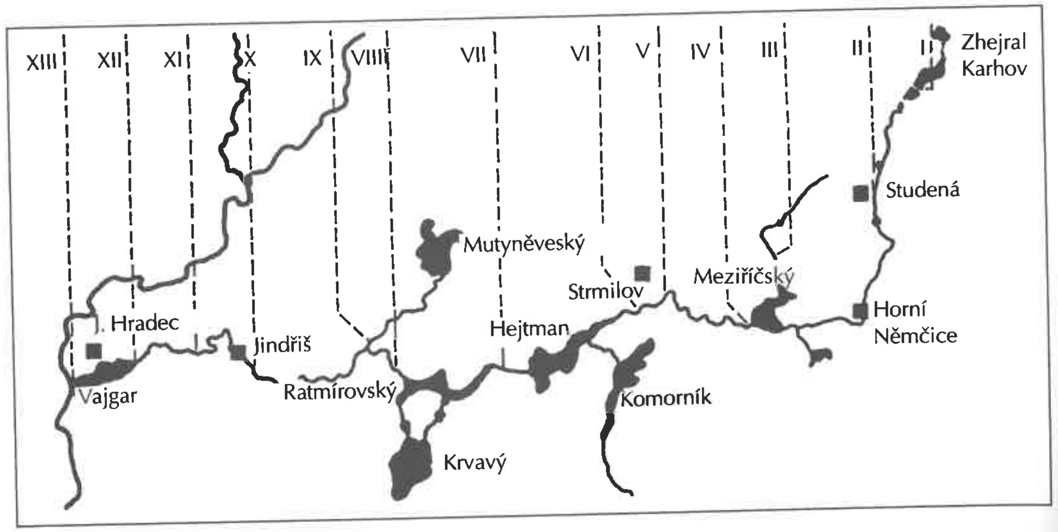

Figure 1.
The Vajgar catchment. Scheme of the fish pond system in the catchment of Studenský/Hamerský brooks and the sampling points (I to XIII).

day (126 l s^{-1}) was built. Unfortunately, four overflow rain water drainage outputs and about twelve individual septic tanks (which lay under the level of canalisation pipes) are still discharged into the fish pond.

Up to 1965, the fish pond was regularly emptied and fish was harvested. Then due to damage of outlet mechanisms and later also due to a small dam which was made between the so-called 'Small' and 'Large' Vajgar, the pond could not be drained.

Vajgar fish pond serves as a sedimentation basin of the Hamerský potok (brook) which brings water from the catchment. Hamerský brook (in its upper part called Studenský) rises from a spring 45 km north-east of Jindřichův Hradec near the hill Javořice (837 m altitude) in Českomoravská vrchovina and flows through the system of several fish ponds (Figure 1).

The catchment area of Studenský/Hamerský brook has about 250 km^2. The average annual precipitation (rainfall) in the 1980s was about 670 mm and the average temperature 7.3°C. The average water flow in Horní Pole was 0.064 $m^3 s^{-1}$ in the years between 1981–85, and 1.2 $m^3 s^{-1}$ in Hamerský brook at Oldřiš near Blažejov. The hydrological data of important fish ponds in the catchment are given in Table 1.

Pre-project investigations

Catchment area

Thirteen sampling sites (Figure 1 and Table 2) have been chosen in the catchments of Studenský and Hamerský brooks in order to identify and evaluate the sources of eutrophication (Stara *et al.* 1988). BOD_5, COD, NO_3, NH_4, PO_4, saprobity index, and trophical potential were measured. Marked differences between summer and winter values were found for nitrate, ammonium and phosphate. In general, the concentrations are lower in summer

Table 2.
Sampling points (I-XIII) in the catchment of Vajgar fish pond.

I.	Karhov fish pond was built in the spring water area of Studenský brook and together with Zhejral fish pond (above) they provide potable water to the abattoir and town of Studená. The water is of good quality only, because of low alkalinity, conditioning with calcium hydroxide and carbon dioxide is necessary.
II.	Šantl fish pond (1,3 ha) serves as the stabilisation pond of the wastewater treatment plant of Studená (monoblock treatment, two-steps activation and anaerobic stabilisation of sludge). About 850,000 m^3 of sewage water (about 500 metric tonnes of BOD_5 a year and 400 metric tonnes of insoluble material a year) from the abattoir and from the town are treated. About 1,600 inhabitants live in the small town of Studená. In the eighties, when samples were regularly taken, the wastewater from the town was partly discharged untreated into the fish pond.
III.	Dvorecký brook which brings water from a mainly agricultural landscape into Meziříčský fish pond.
IV.	Meziříčský fish pond is the first larger fish pond below the town of Studená and therefore served as the recipient of untreated sewage water from Studená until 1980, when the wastewater treatment plant was built. The brook flowing from Meziříčský fish pond is called Hamerský.
V.	Hamerský brook above the small town of Strmilov (about 1,800 inhabitants).
VI.	Hamerský brook below the town of Strmilov. Only part of the sewage water of Strmilov is treated. In total, about 70,000 m^3 per year is discharged from Strmilov town, only 18,000 m^3 per year of which is treated.
VII.	Hamerský brook under the fish pond Hejtman near the factory Strojobal Rozkoš. The factory produces metal lids for glass conserves. The wastewater is pre-treated in septic tanks.
VIII.	Hamerský brook below Ratmírovský fish pond, a well-known place of recreation.
IX.	Mutyněvský brook receives wastewater from the small village of Oldřiš (120 inhabitants) and flows into Hamerský brook. The distillery (potato processing and fermentation) discharge pre-treated wastewaters into the Hamerský brook (*c.* 4 tonnes y^{-1} of BOD_5 and 18 tonnes y^{-1} of insoluble material).
X.	Hamerský brook above the old mill in Jindřiš. In Jindřišské valley (Jindřišské údolí) the brook has a status of trout water quality.
XI.	Hamerský brook below the village of Jindřiš near a cattle stable.
XII.	Hamerský brook at the inlet to Vajgar fish pond in Jindřichův Hradec.
XIII.	Hamerský brook at the outlet from Vajgar fish pond.

when nutrients are bound in the organic matter of micro-organisms, algae, etc. The concentration of ammonium under ice was 4.1 mg l^{-1} and it became lower than 1 mg l^{-1} in summer. The highest load of nutrients was measured below the town of Studená (locality II) where the average values of ammonium were about 30 mg l^{-1}, for PO_4 about 16 mg l^{-1}, COD_{Cr} about 160 mg l^{-1}, and BOD_5 about 32 mg l^{-1}. Downstream, due to self-purification, the concentration of nitrate decreases to about 20%, the concentration of ammonium decreases by about one order of magnitude, and the concentration of phosphates decreases by about two orders of magnitude. The organic matter decomposes slower – to about 30%.

The values of trophic potential and values of saprobity index show similar trends as the values of nutrient concentrations. The biological analysis showed water quality measured as saprobity index according to Sládeček *et al.* (1989), between beta- and alfa- meso-saprobity, the water in Šantl fish pond was polysaprobic with dominance of *Sphaerotilus* and *Oscillatoria*.

In general, the high nutrient level in these catchment results from agricultural run-off, communal waters discharge, fish pond fertilising, and from fish pond sediments which

Figure 2.
Seasonal course of chlorophyll a concentration in Vajgar fish pond from January 1990 till July 1991.

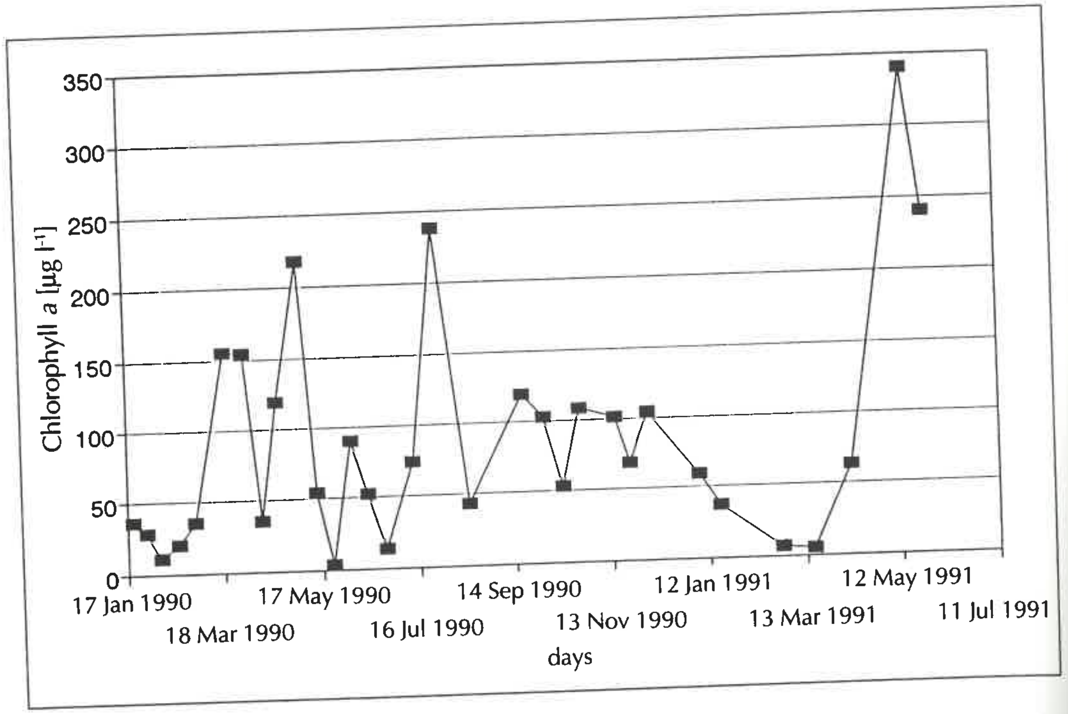

Figure 3.
Seasonal course of total phosphorus concentrations in inflow and outflow of the Vajgar fish pond in 1991.

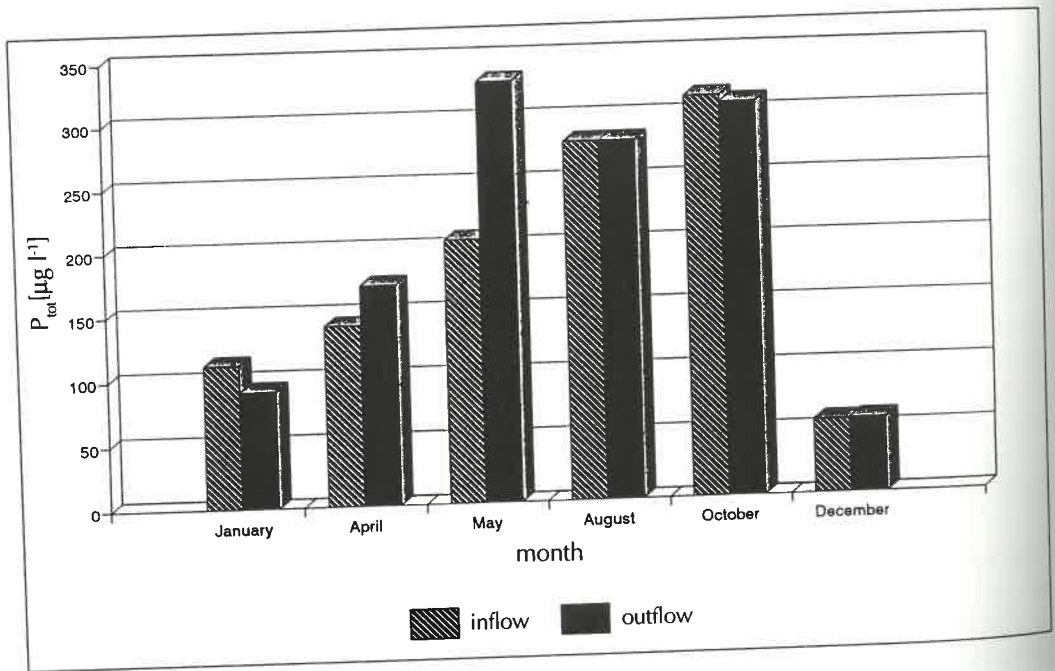

Figure 4.
Sediment core with distinctive grey and black layers of sediment.

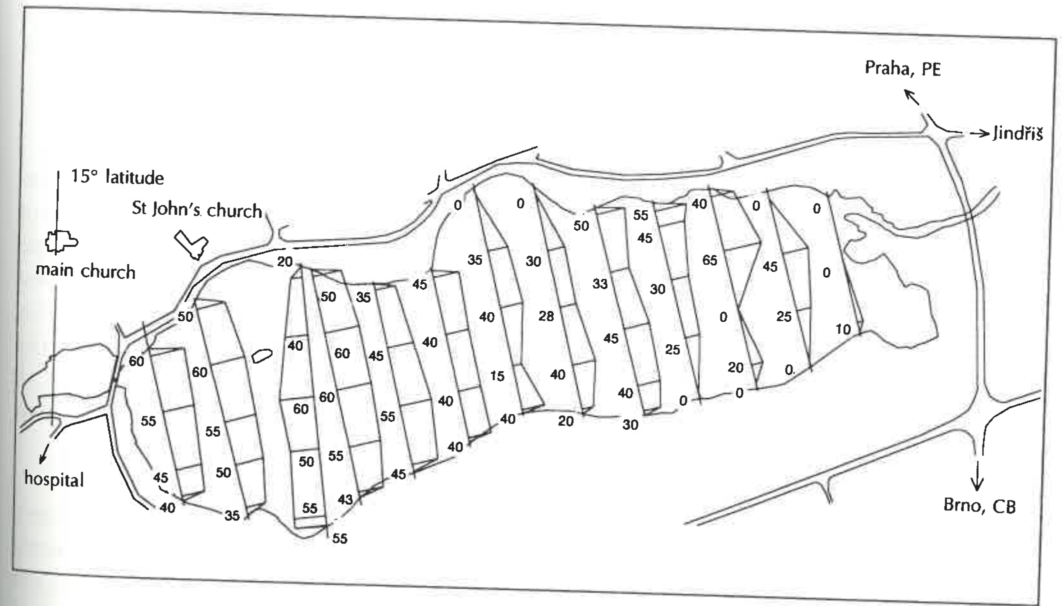

Figure 5.
Thickness (cm) of black sediment in the Vajgar fish pond.

release nutrients to the water. The results of the chemical analyses of nutrients correspond with the test of trophical potential: 570 mg l^{-1} and 860 mg l^{-1} (locality II), and 60 mg l^{-1} and 195 mg l^{-1} (locality VIII–XIII), in July and October 1985, respectively.

Vajgar fish pond

Water quality

Mass development of *Stephanodiscus hantzschi*, indicator of eutrophic waters, in the summer of 1985 was typical for Vajgar fish pond as well as for the Hamerský brook and fish ponds up to Mezířičský fish pond. Water blooms of *Microcystis* occurred regularly in summer and caused environmental problems – a smell of degrading blue-greens and the impossibility of using the pond for recreation.

The seasonal course of chlorophyll *a* concentration in Vajgar fish pond from January 1990 till July 1991, before sediment removal, is shown in Figure 2 (Kummel & Pechar, unpub. data). Seasonal courses of the total phosphorus budget in the inlet and outlet of Vajgar fish pond (Figure 3) have shown that the fish pond served as a phosphorus source in late spring. At that time, the sediment begins to warm up, organic matter starts to decompose removing oxygen from the bottom and, thus, allowing the release of phosphorus from the sediment. This observation confirmed that in-lake restoration measures have to be undertaken should the nutrient loading be reduced.

Zooplankton and benthos were also studied in order to compare the status before and after the restoration.

Stratigraphical study of sediment

Sediment cores (Figure 4) were taken in order to measure the thickness of black (nutrient-rich) sediment which should be removed from the pond. Map of the black sediment thickness were made for the whole pond using data obtained on transects 100 m apart and at 50 m distances along each transect (Figure 5). In the centre of the pond, the deepest possible core was taken in order to try to reach the original bottom of the valley basin. This approximately 2.4 m deep core was divided into 42 five-centimetre segments and the following analysis made for each segment: dry matter, organic matter, Al, As, B, Bi, Ca, Cd, Co, Cu, Cr, Fe, Hg, K, Mg, Mn, Mo, Na, Ni, P, Pb, S, Se, Sr, Ti, V, W, Zn, and pollen analysis (selected parameters in Figure 6).

The release of phosphorus from the upper black layer of sediment and from the lower grey layer (both total P and phosphate) were tested in laboratory under aerobic and anaerobic conditions (Figure 7). The dating of the whole profile was made by ^{14}C, the dating of upper layers was measured by ^{134}Cs and by ^{137}Cs dating. The dating by ^{14}C showed that the original bottom was flooded about 1,000 years ago. The rate of oxygen consumed by the different layers of sediment was also studied and could be correlated to organic matter content (Figure 8). The dry matter content in the vertical sediment profile is shown in Figure 9. The pH, alkalinity and phosphate in the interstitial water were measured and microbiological

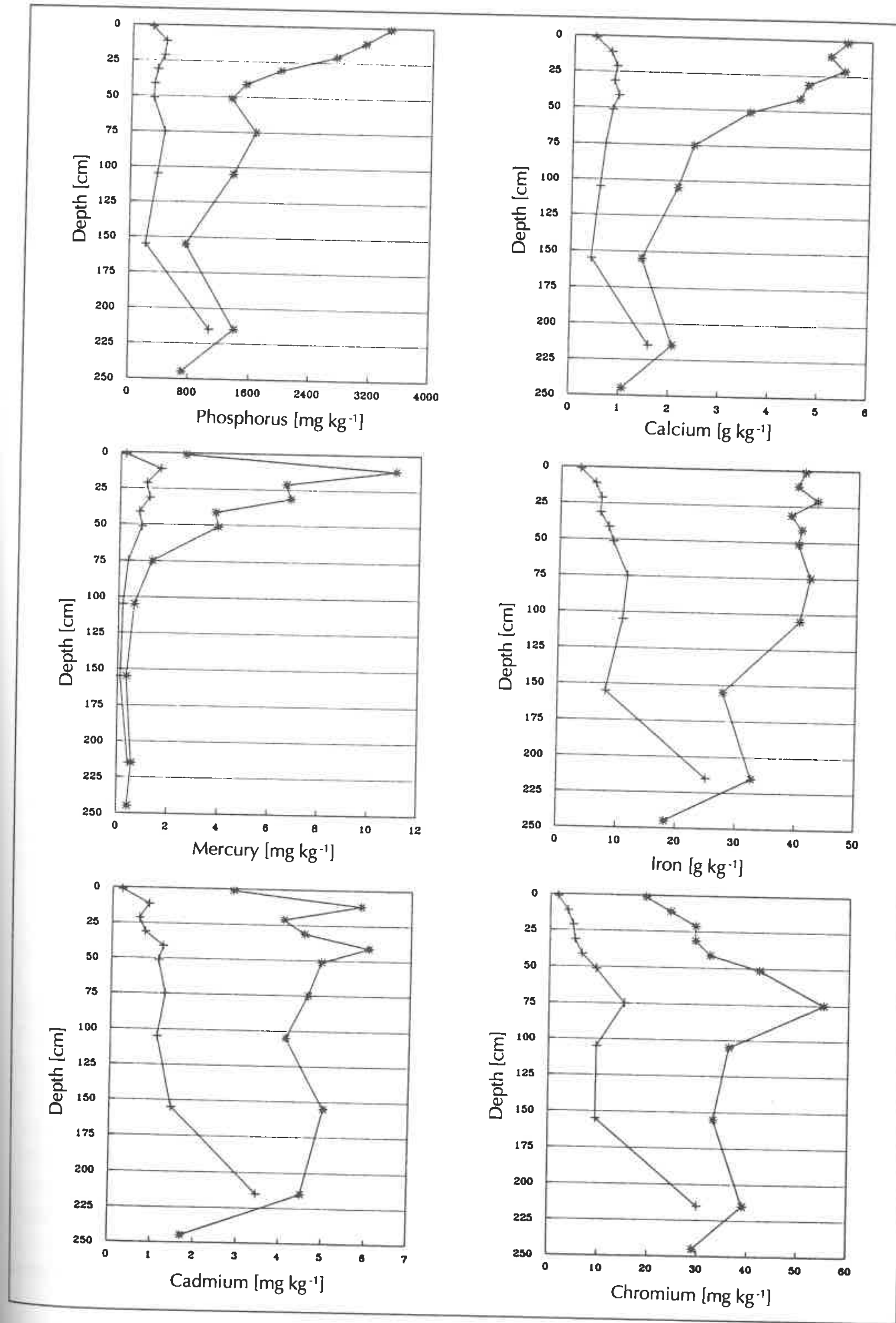

Figure 6. Vertical profile of concentrations of selected elements in fresh (+) and dry (✻) mass sediment of Vajgar fish pond.

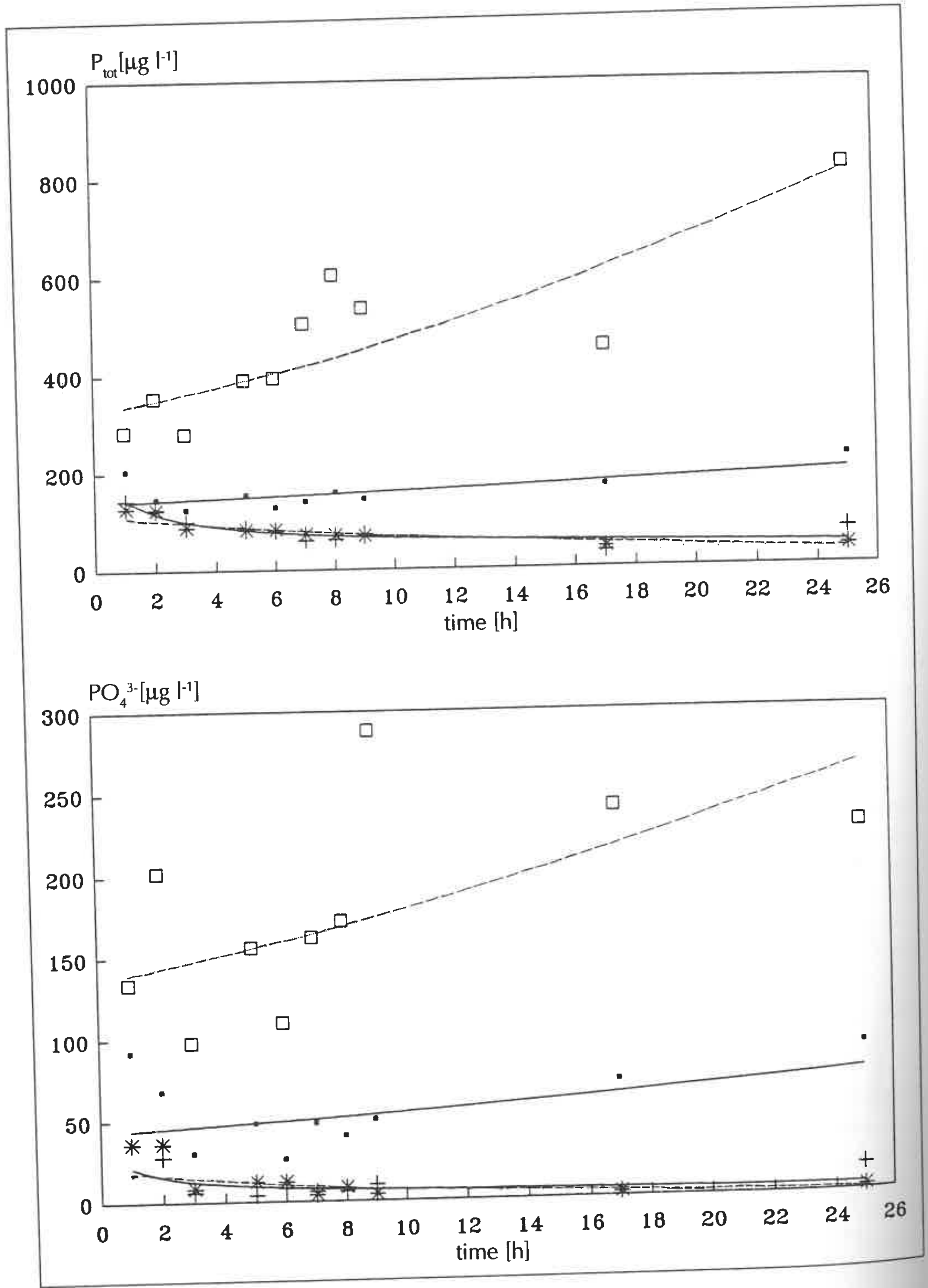

Figure 7. The release of phosphorus (P_{tot} and PO_4^{3-}) from the upper (black) layer (□, •) and lower (grey) layer (✳, +) of sediment from the Vajgar fish pond under aerobic (•, +) and anaerobic (□, ✳) conditions.

tests were undertaken. The analyses showed that the sediment could be safely applied to fields for agricultural use.

Restoration

The sediment accumulated in the pond served as an uncontrolled source of nutrients. Therefore, attempts were made to find a technique of sediment removal convenient for the situation in Vajgar fish pond. In 1986, military pontoons carried a suction-hopper container lorry which sucked the sediment out and transported it by a conveyor-belt to where it was de-watered using different types of precipitants. The de-watering of sediment on the conveyor-belt itself would be possible but expensive. In 1987, an old type of suction-dredger was tested (Czechoslovak-made suction dredger SB 20). The dredger was able to pump sediment onto the bank of the pond. However, the sediment was very diluted (less than 4% of dry matter), the dredger made holes in the bottom which could cause problems during fish-harvesting, the maximum length of transport and capacity of the dredger was low, and systematic dredging of the bottom surface was very difficult.

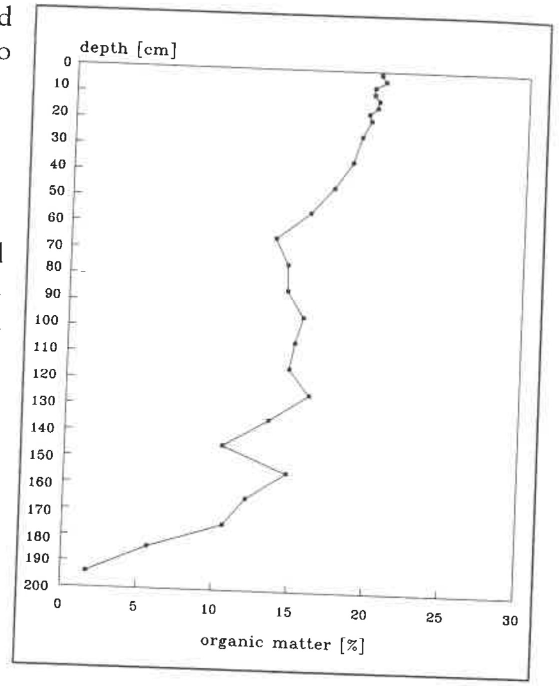

Figure 8. The organic matter content in the sediment profile of Vajgar fish pond, in October 1990.

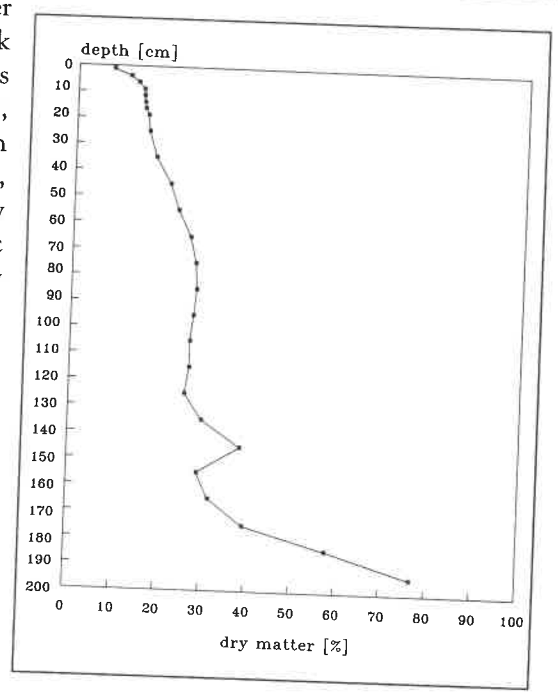

Figure 9. The dry matter content in the sediment profile of Vajgar fish pond, in October 1990.

New technology

From the beginning, the restoration of Lake Trummen (see previous case study and e.g. Björk 1988) inspired the restoration activity of Vajgar. However, the excessive water pumped by old, conventional suction-dredgers during sediment removal constituted a general problem. Hence, there was the need for a suction dredger which would respect the demands of limnologists (for details see Chapter 7, Sediment removal). The prototype of such a dredger, designed in Sweden, was constructed for the sediment removal in Vajgar (technical parameters are given in Table 3).

Figure 10.
Vajgar fish pond. Suction-dredger. Photo Jan Pokorný 1991.

Table 3.
Basic technical data of the suction-dredger.

Measurements	length: 13.50 m height: 2.80 m width: 5.20 m weight: 18.5 t draft: 0.80 m	
Nozzle	width: 1.2 m height: 0.80 m efficiency: 2–4 m³ min⁻¹ pump: 40 kW, Turo, (Swiss) pumping distance: 1 km pumping height: 30 m	
Main engine	Stationary diesel engine Volvo Penta	
Hydraulics	two independent hydraulic systems: 1. pump 2. movement of nozzle depth and steering, etc.	medium: biodegradable oil (BioSafe).
Measuring instruments	angle transmitters (angle of wires) incremental decoders (relative length of wires, speed) absolute decoder (absolute length of wires) densitometer (density of the sediment)	flowmeter (flow rate, volume) load cell (tension of pulling wires) echolot (depth)
Electronics	microcomputer of industrial standard (Satt-Control, Sweden) programmable from Personal Computer, terminal regime allowed, two programme loops (BASIC, pseudoassembler), eight independent programmable regulators, 64 counters, 64 timers, battery-backed programme and memory registers content, display, keyboard 3 digital/analog input/output interface cards interfaces to hydraulic valves, relays	
Automatic or manual regime (regulated values)	speed (typically 2 m min⁻¹) flow (typically 2 m³ min⁻¹) depth of the nozzle	

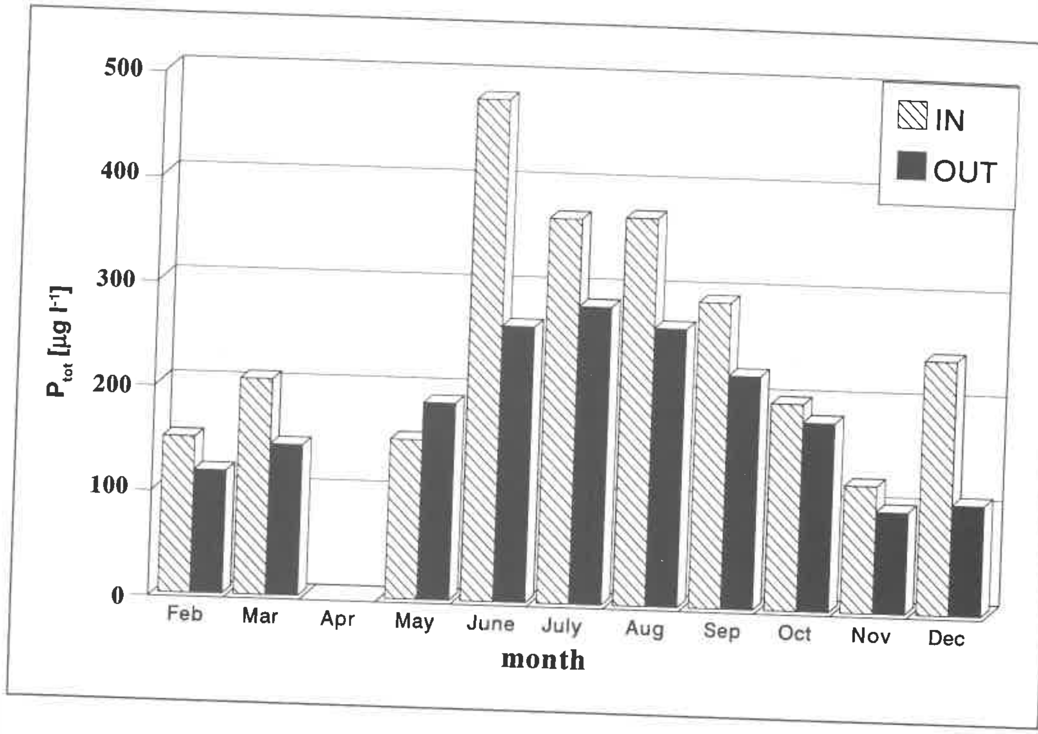

Figure 11.
Seasonal course of total phosphorus concentrations in inflow and outflow of the Vajgar fish pond in 1993.

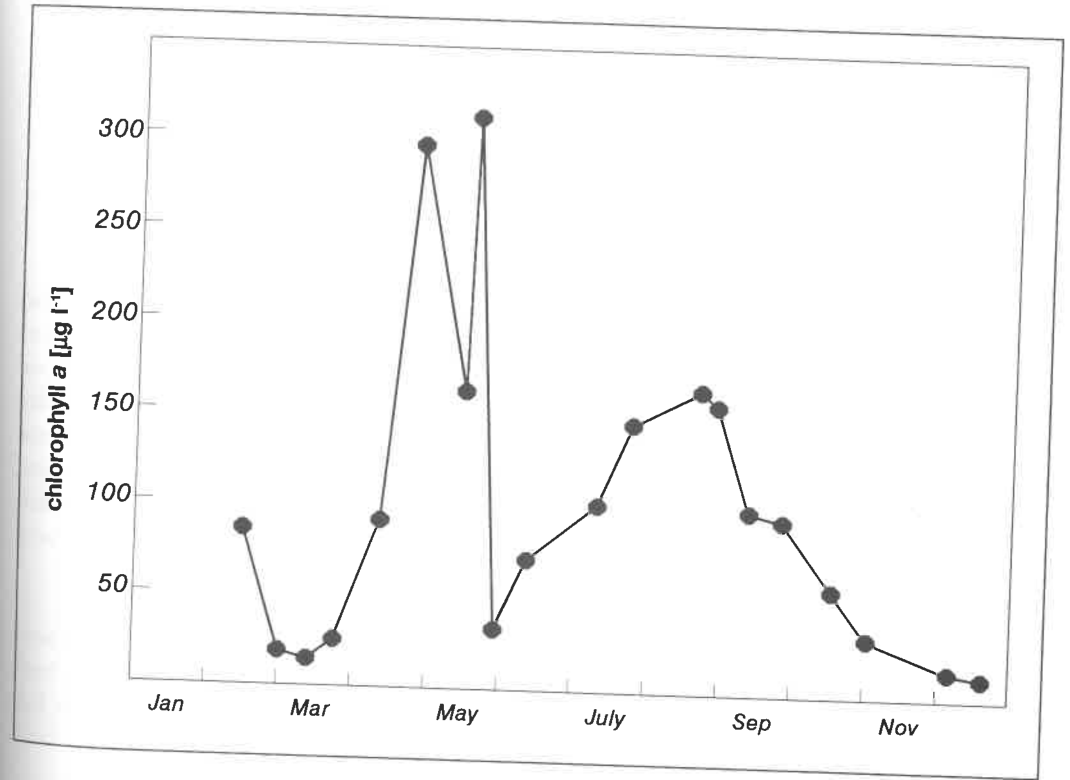

Figure 12.
Seasonal course of chlorophyll *a* concentrations in Vajgar fish pond in 1993.

The sediment-pumping started in August 1991 (Figure 10). By the end of 1992, about 330,000 m^3 of sediment had been pumped out from the pond and transported in pipes for about 2.5 km to seven settling lagoons. From 1994, the sediment will be used for agricultural purposes.

The town of Jindřichův Hradec supported by the district office financed the restoration works, the Environmental Ministry subsidised the sediment removal and scientific work. The whole cost was *c.* 850,000 US$.

The sledge suction-dredger designed in Sweden (Industrikonstruktioner AB) and produced in cooperation with Czech and Swedish firms was used. Functioning automatically, the selective removal of sediment with a minimum of extraneous water are the main advantages of this suction-dredger.

Situation after restoration

In April 1993, Vajgar fish pond was partly drained in order to remove the 'temporary' dam placed between Large and Small Vajgar and to allow the complete drainage of Vajgar fish pond again. In the second half of April, the fish pond was refilled with water and stocked with heavier carp (*Cyprinus carpio*, 1–2 kg, about 250 kg ha^{-1}). These carp were added to the original fish stock in order to disturb and mix the rest of the black sediment.

Water transparency (Secchi depth), chlorophyll *a* content, algal species composition, and phosphorus budget were monitored regularly during the whole of 1993. Vajgar fish pond was acting as a phosphorus trap during the whole year except for May when the pond was not completely filled with water. The output P budget mean value (one week's daily samples per month) was lower than the input P value, and the difference between inlet and outlet concentrations of total phosphorus in 1993 (Figure 11) were evidently higher than in 1991 (Figure 3).

There is still a high load of nutrients from the catchment area entering the Vajgar fish pond via Hamerský brook; an even higher load of phosphorus was recorded in 1993, in comparison to 1991. These sources now need to be controlled. Sewage treatment plants do not remove phosphorus satisfactorily and two further problems remain. Firstly, the run-off from the agricultural catchment needs to be controlled. Secondly, the fish ponds of the upper catchment area need to be managed in a manner that the quantity of fertilisation used in a fish pond is in accordance to the measured level of nutrients in the fish pond and the size of its fish stock. This is a more 'natural' management (Faina, *pers. comm.*, Fott *et al.* 1980, Pokorný *et al.* 1994) and would reduce the excessive nutrient outflow from these fish ponds.

Monitoring of Vajgar fish pond is still continuing. In the first year after the sediment removal, the chlorophyll *a* concentration increased up to *c.* 160 µg l^{-1} (Figure 12). The high values of chlorophyll in April and May are due to low water level and relatively high fish stock at high nutrient discharge from the catchment. No blooms of *Microcystis* have been registered even though *Microcystis* cells were transported from the upper pond into Vajgar. Other minor blooms occurred briefly. Swimming is again possible.

The measures are now being taken to decrease the external loading and to implement a fish stock control aimed at reducing the amount of small white fish – predator of *Daphnia*. The measures have been taken to stop the discharge of several individual septic tanks. The improvement of water quality in the whole catchment area will be a long process which requires a lot of understanding among local authorities, farmers, and fish pond managers, and which may even require changes in the tax system. Some possible solutions of how to incorporate ecological consideration into catchment development are suggested by Ripl *et al.* in Chapter 3. Only the catchment-wide redevelopment approach can ensure that the restoration effort in Vajgar and the results achieved will be sustainable in the long term.

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Treatment of overgrown shallow lakes – macrophyte control: Lake Hornborga, Sweden

Sven Björk

Background

In order to get access to arable land, the water level of a great number of wetlands has been lowered. However, the aeration of the peaty and organic sediment soils made available for agriculture resulted in rapid subsidence due to increased mineralisation. The subsidence often amounts to 1 to 2 cm per year in temperate latitudes and to about 4 cm per year in tropical countries. Therefore, after some decades, the water level lowering of shallow lakes nearly always created problems. In many cases the remaining lakes have become overgrown by macrophytes and the obtained arable land impossible to cultivate, being too wet due to subsidence soil. Some of the lowered, shallow lakes used to be outstanding waterfowl biotopes and efforts are now being made to restore them to their former ornithological value, to get back fishing-waters and to recreate the qualities of the landscape in general.

Among the numerous lakes degraded through water level lowering in Sweden, Lake Hornborga (in Swedish, Hornborgasjön, location 58°29'N, 13°34'E, Figure 1) in the province of Västergötland, is the best known. This is partly because the lowering was a legal scandal and a big economical failure, and partly because extensive basic investigations have been made there, concerning the possibilities to restore this type of shallow, drained lakes. The eco-technical restoration methods elaborated at Lake Hornborga were applied there already in the 1960s (Björk 1972).

Causes of degradation

Until man interfered with the well-organised complex of components that functioned within the Lake Hornborga ecosystem, it maintained a rather high degree of productivity without suffering from rapid ageing. This is quite remarkable for a lake of this size (30 km^2), shallowness and high trophic level. The lake used to have a maximum depth of 3 metres, but most of the lake was much shallower. A constellation of factors such as rapid water renewal, well-situated inlets and outlets (Figure 1), strong water- and strong ice movements, provided a good system for transporting matter out of the lake and prevented the lake from becoming overgrown. The emergent vegetation was mainly restricted to wind-protected shore areas. The organic matter of the richly developed submerged vegetation easily decayed and was transported out of the lake, while the precipitated calcium carbonate particles covering these plants settled to the bottom. Before degradation following the water level

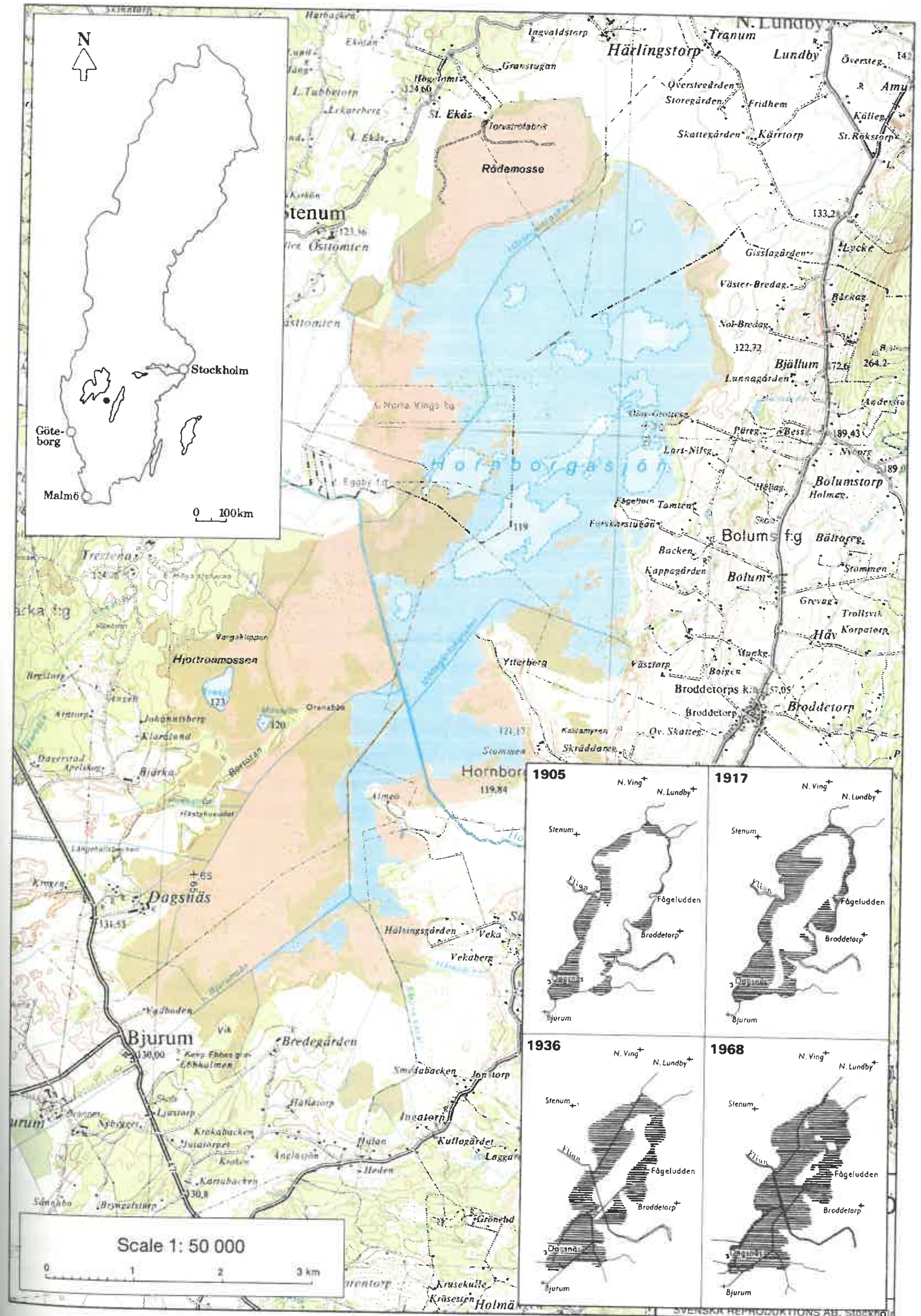


Figure 1. Lake Hornborga (Hornborgasjön), Sweden. Location; topographic map from 1967–69; and recent overgrowth by emergent vegetation (1905, 1917, 1936, 1968).

lowering and finally, in 1933, complete drainage, Lake Hornborga ranked as one of the most valuable waterfowl lakes in north-western Europe.

Since 1802, the lake has been lowered five times in attempts to obtain arable land. The last big failure, in 1932–33, resulted in a bottom that was drained in the summer, and which in a considerable part of the lake consisted of lake marl. A hilly land area of about 616 km^2 drains into the lake. This area needs Lake Hornborga as a reservoir to catch rainwater and melted snow that rush down the hills to the plain below.

In 1954, the water level was slightly raised in a part (12 km^2) of the lake that was (Figure 1) diked-in. This could be looked upon as a large-scale field experiment. As described below, it revealed, amongst other things, the necessity of removing the macrophytic vegetation and of restoring a clean sediment bottom in areas intended to be re-created and preserved as open water with submerged vegetation.

During the 1930s until the middle of the 1960s, the Hornborga area went through the typical development for lowered, shallow lakes and their drained surroundings. The lake area became overgrown by emergent vegetation. In fact, the slight raising of the water level in 1954, to a maximum depth of *c.* 80 cm in the diked-in portion, considerably improved the environmental conditions for common reed which had colonised the lake bottom. Huge masses of coarse detritus accumulated and filled in the lake basin (Figures 2 and 3). At the same time, the surrounding organic soil areas subsided. Altogether this resulted in fairly severe flooding and it became impossible to cultivate the successively wetter organic soils.

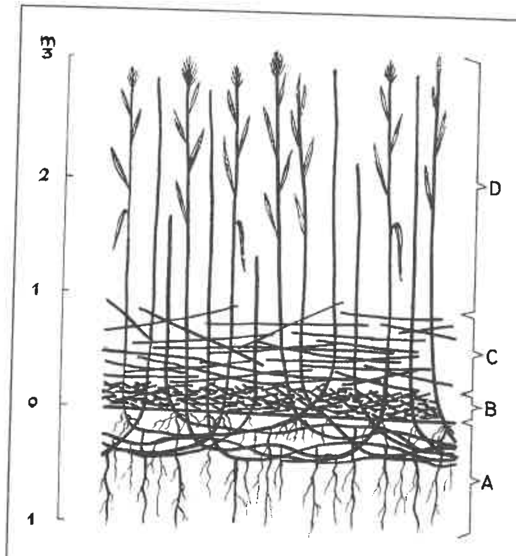
Lake Hornborga lost its value as an outstanding waterfowl habitat because the lake ecosystem's structure and function were completely destroyed. Monocultures of common reed (*Phragmites australis*, Figure 9) and slender-tufted sedge (*Carex acuta*), nearly covered the whole lake area in 1967. Mixed plant communities including living and dead willow bushes (*Salix* spp.) made large areas almost impenetrable.

Basic research, arguments and directives for restoration

The legal course of events, including the long series of illegal ingredients, leading to the destruction of the great natural asset of Lake Hornborga, was revealed by Swanberg (1959a, 1959b, 1968, 1971) who elucidated the very serious economic and ecological consequences of the drainage project. Thanks to Swanberg's basic research and persevering, convincing argumentation, it was successively realised by all parties concerned that the drainage project was a multidimensional failure that had severely hit both nature and culture. The Swedish Government decided to investigate the possibilities to restore Lake Hornborga. According to governmental directives, investigations to determine whether or not the lake could still be restored and whether restoration would be sufficiently permanent were to be carried out (Vilborg 1973). In the case that restoration would be theoretically possible, methods to attain this goal were to be elaborated.

Restoration goals

The ecological goal for most restoration projects in lowered and overgrown lakes is to create an open water area and a mosaic of open water and emergent/submerged vegetation. The restored lake/wetland should be brought to a stage of permanence where no future extensive management programme is needed. For responsible, long-term planning of wetland restoration, management and protection, this is of the utmost importance.



Key
A = roots and rhizomes. B = very coarse *Phragmites* detritus. C = horizontally deposited stems of several shoot generations together with basal parts of broken stems. D = vertical living stems and several generations of dead stems.

Figure 2. Lake Hornborga. Distribution of living and dead material in stands of *Phragmites*. At the start of the restoration of the bottoms overgrown by *Phragmites*, the thickness of the B and C layers amounted to *c.* 75 cm. From Björk 1972.

Shallow lake ecosystems are not at all static but highly dynamic systems and can suffer from a very rapid ageing processes. After water level lowering, as well as after an insufficient increase in water levels as a means of 'restoration', such lakes usually pass through a transient period of flourishing birdlife. Sometimes, as in the Lake Hornborga case, it is still difficult to convince the general public that this period is of just short duration and that ecologically realistic measures to counteract the rapid ageing must be taken in order to give the ecosystem sustainable character. When planning for restoration of such systems to sustainable units – according to man's time-scale – it is necessary to apply an holistic approach not only in space, i.e. to comprise the whole ecosystem including its catchment area, but also in time, i.e. to allow for the factors which have an influence on the speed of the ageing processes.

Field investigations

The field investigations started in 1967 and were carried out by a team of ornithologists (under the leadership of Dr. P.O. Swanberg, 1980), limnologists, technologists, hydrologists, economists and agriculturalists. In 1968, the lake and its subsiding surrounding areas were mapped using aerial photographs to obtain contour maps with a 25 cm interval. Following the governmental directives, the necessary limnological restorative measures and developmental prognoses were elaborated, based on the specific conditions in the degraded Lake Hornborga itself. As the ecological properties in every lake/wetland ecosystem are unique, a tailor-made restoration plan has to be designed for each individual project.

One year of limnological studies made it quite clear – theoretically – that Lake Hornborga could be restored (Björk 1972). The most serious problem following the lowering of water levels in shallow lakes is the development of a root-felt in the upper sediment layer and the accumulation of coarse plant material (detritus) produced by highly productive macrophytic vegetation.

Figure 3. Lake Hornborga. Accumulation of coarse detritus in the 11km² large stand of *Phragmites* in the northern half of the lake. Photo: Sven Björk 1969.

With clear evidence, the results of the limnological investigations demonstrated that before the decisive step for the restoration of Lake Hornborga – i.e. the raising of the water level – was to be taken, the bottom had to be treated in order to get rid of the coarse plant material and the root-felt. In 1968, large-scale field experiments were begun in order to develop the technical methods necessary to counteract the otherwise irreversible damage, i.e. the huge masses of accumulated detritus as well as the root-felt of the emergent macrophytes had to be removed (Figures 2 and 3). If the water level is to be raised over areas overgrown by sedge (*Carex*), reed (*Phragmites*), bulrush (*Schoenoplectus*) etc., the gas (mainly methane) produced in the bottom accumulates in and beneath the root-felt. After a period of time the root-felt would, therefore, float to the surface and would soon be overgrown by new vegetation. This process leads to plaur formation (floating stands of emergent macrophytes).

Project design

The basic limnological investigations carried out in all parts of Lake Hornborga included both the inflowing streams and the outflow, and the well-documented changes in the ecosystem following the drainage. Bottom levels, sediment, peat, water and flora were studied and the results were synthesised with findings from the groups responsible for ornithology, hydrology etc. The ecological restoration plan for Lake Hornborga was presented in 1973 (Plan/73, *cf.* Björk 1972, Swanberg 1972, Wilborg 1973).

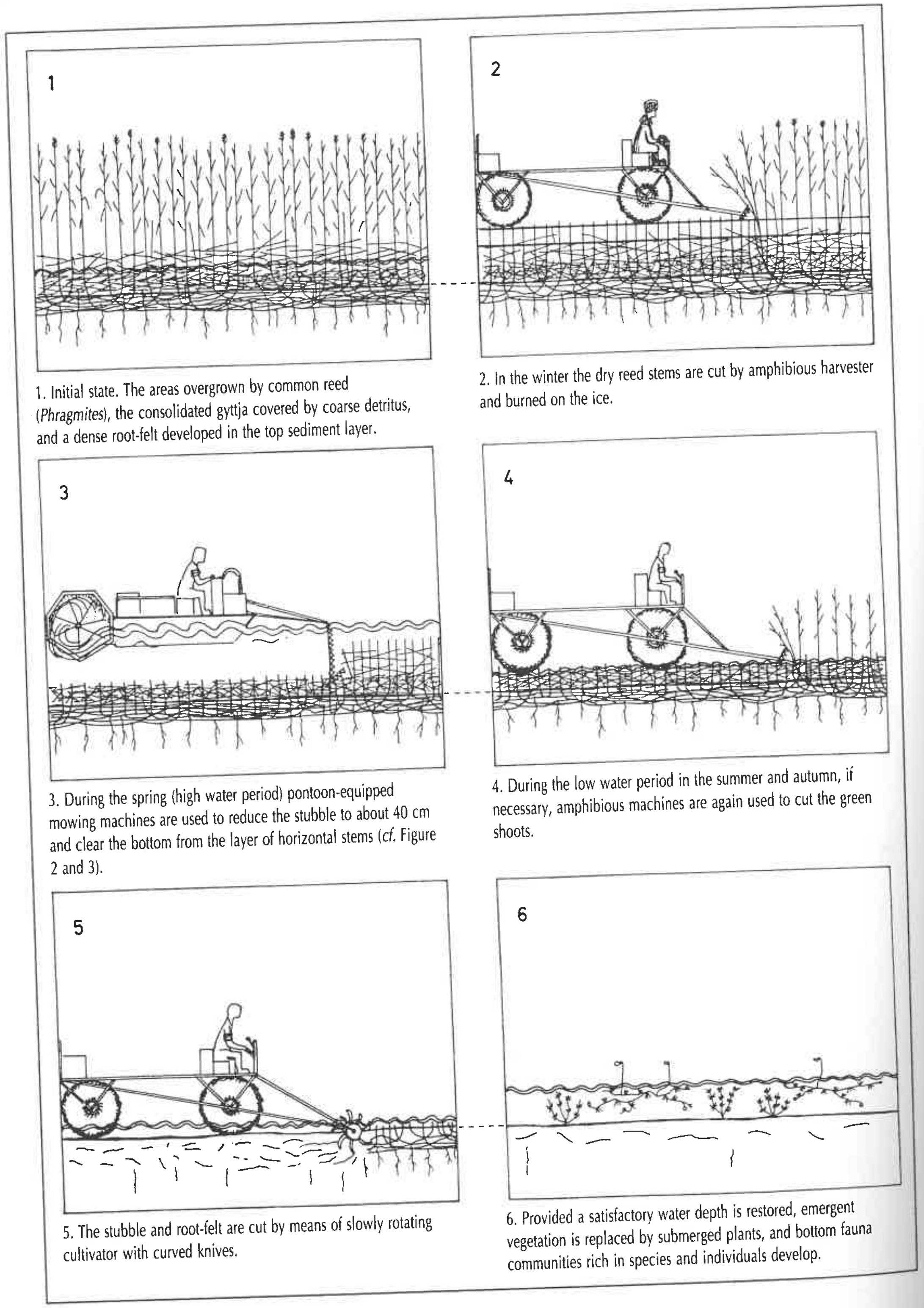
In Lake Hornborga, as elsewhere, the sedge root-felt poses a problem, as it is thick, resistant and for economical reasons impossible to remove from the very large areas covered by it (*c*. 18 km²). In small water bodies, the floating root-felt can be removed by means of a dragline, or cut up, towed to the shore and removed. The reed root-felt can easily be cut by amphibious rotavators that were constructed for the Lake Hornborga project (Figure 4).

Following the governmental directives, the limnological project goal for Lake Hornborga was to transform the reed (*Phragmites*) areas to open water (about 11 km²) and to keep emergent vegetation in the area covered by sedge (*Carex*) (about 18 km²). When the water level is raised, the sedge root-felt will float to the water surface and be recolonised by reed (*Phragmites*) and bulrush (*Schoenoplectus*). Before raising the water level, the tough sedge root-felt can be removed from small areas by means of amphibious excavators. After raising the water level these parts will be preserved as open waters surrounded by plaur vegetation and attractive for birds.

The procedure to replace emergent vegetation by submerged vegetation is illustrated in Figure 5. During the field experiment in the years 1968–1972, convincing results for a successful restoration using the above series of methods were obtained. For cutting macrophytic vegetation and destroying the root-felt, prototypes of amphibious and pontoon machines were constructed (Figure 4). The use of the constructed equipment is schematically shown in Figure 5, according to dry and flooded conditions characteristic for Lake Hornborga. During dry periods in late summer and early autumn it was sometimes possible to quickly clean several km² from accumulated reed detritus by means of fire (Figure 6), especially in

Figure 4. Lake Hornborga. Prototype of rotavator for removal of the stubble mat and rhizome layer of *Phragmites*. The slowly moving knives cut the rhizomes in long, easily floating pieces. Photo: Sven Björk 1972.

Figure 5. The Lake Hornborga restoration project. From Björk, 1972.



1. Initial state. The areas overgrown by common reed (*Phragmites*), the consolidated gyttja covered by coarse detritus, and a dense root-felt developed in the top sediment layer.

2. In the winter the dry reed stems are cut by amphibious harvester and burned on the ice.

3. During the spring (high water period) pontoon-equipped mowing machines are used to reduce the stubble to about 40 cm and clear the bottom from the layer of horizontal stems (*cf.* Figure 2 and 3).

4. During the low water period in the summer and autumn, if necessary, amphibious machines are again used to cut the green shoots.

5. The stubble and root-felt are cut by means of slowly rotating cultivator with curved knives.

6. Provided a satisfactory water depth is restored, emergent vegetation is replaced by submerged plants, and bottom fauna communities rich in species and individuals develop.

the sections of the lake where willow bushes prevented the immediate use of machines. However, the fire does not destroy the root systems. Therefore, the productivity of the reed becomes higher when the fire has improved the environmental conditions through removal of the layers of settled detritus and dry stems which decrease both temperature and light at the bottom (see Chapter 7, Macrophyte control – Figure 3).

It is usually most practical to start the removal of the macrophytic vegetation along the wind-and wave-exposed shores. During periods of high water level, the cut and loosened plant material is transported to the exposed shore. During low water periods it is possible to burn it on the shore or to collect and compost it. In small scale projects, it is practical to carry out the cutting and root destruction in sub-areas surrounded by floating booms. All the loosened material is dragged within the boom to the shore and removed from the lake by means of a conveyer belt. With the procedure as described in Figure 5, enormous masses of coarse detritus deposited in the stubble mat was loosened and together with the rhizomes and roots transported by the spring high water to the shores along the wind-exposed portions of the lake (Figure 7) where they were burned in the summer.

When the detritus was removed, the consolidated original gyttja (mainly lake marl) again became the bottom. If covered by sufficiently deep water, the reed monoculture is replaced by submerged vegetation, and a rich bottom fauna, microbenthos and periphyton develop. The biotope changes documented already during the experimental period resulted in a very obvious improvement in the waterfowl fauna (Figure 8).

The restoration of bottom areas primarily overgrown by common reed (*Phragmites*) is an easy procedure by means of the technical equipment developed for the Lake Hornborga project. Even if the root-felt of bulrush (*Schoenoplectus*) and sedge (*Carex*) could be cut by rotavators, the main part of the loosened material would remain on the bottom, stationary or partly drifting. In case of common reed, the internodes of big pieces of stems and rhizomes are filled with gas, providing them with an excellent buoyancy (Figure 7). It is imperative that the rotavators cut the rhizomes in big pieces. Machines making a slurry of both stems and rhizomes are destructive because the buoyancy of the material gets lost and the bottom is not cleaned but becomes covered by a layer of coarse, partly drifting particles.

After treatment of the bottom for an area of 11 km^2 as described, and after the manual removal of trees and especially bushes along the shores and in the bays of the former lake, the restoration plan presented in 1973 (Plan/73) recommended the raising of the water level in two steps, first by *c.* 1 m and then, after about 5 to 10 years by a further 0.5 m. This procedure was recommended in order to get the wind-exposed shores worked up, cleaned and washed by the waves in a natural way. The primary raising of the water level should be sudden and big enough to prevent both successive recolonisation of emergent species, mainly *Phragmites*, from the littoral zone of vegetation and survival of initial stands in open shallow water areas. Before the lake degraded completely, *Schoenoplectus* was the characteristic species. After a restoration according to Plan/73, this species would again develop outside *Phragmites* stands along the shores and within plaur areas. With the distribution of the emergent vegetation under control by means of water depth, wave and

Figure 6.
Lake Hornborga. Burning of *Phragmites* reedbeds. Photo: P.O. Swanberg 1970.

Figure 7.
Lake Hornborga. *Phragmites* rhizome pieces after cutting with rotavators. Photo: Sven Björk 1969.

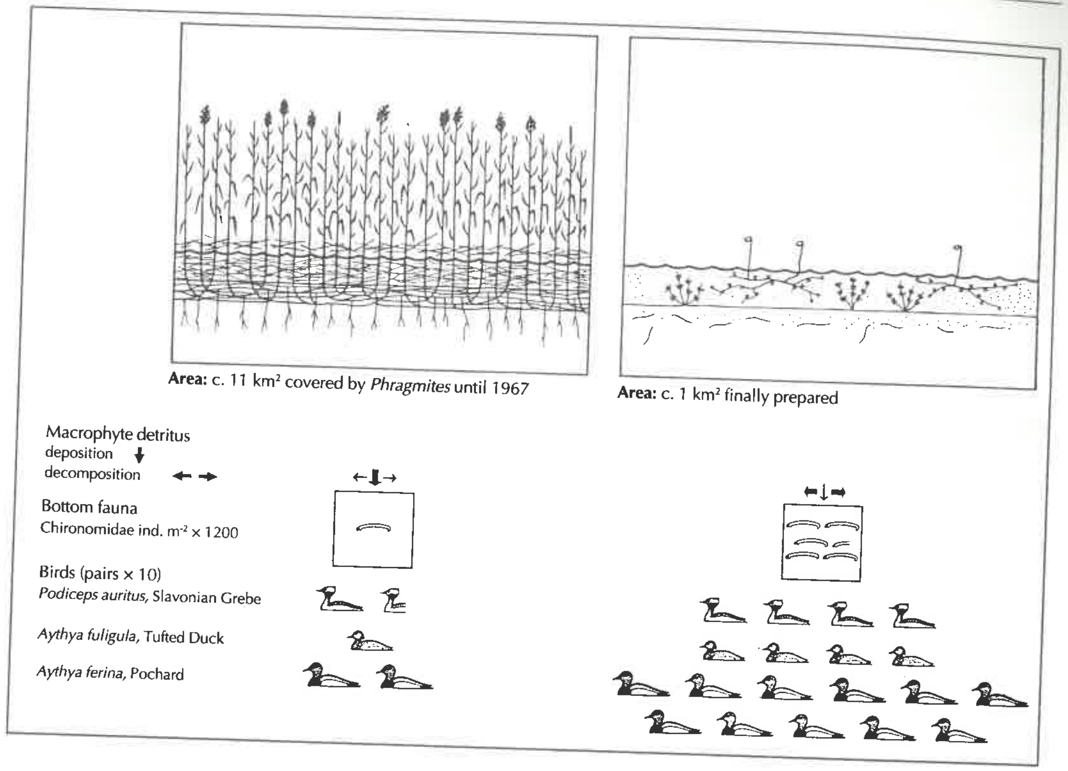


Figure 8.
Lake Hornborga before and after demonstration experiments for directing the primary production from emergent to submerged vegetation. Water level not yet raised. Comparison between conditions in 1965 and 1971. Data from H. Berggren and P.O. Swanberg. From Björk 1972.

especially ice action, the sediment bottom cleaned from coarse detritus could be largely covered by underwater vegetation (Figure 9).

According to Plan/73 lowland areas adjacent to Lake Hornborga should not be cut off from the restored lake as shallow shores and vast wetlands also belonged to the most characteristic features of the former lake (Vilborg 1973). Plan/73 recommended the construction of 1) the compulsory dike at the outlet for making it possible to raise and regulate the water level and, in addition to that 2) short dikes for protection of still arable land against high-water floods, part of these landwards from large plaur areas (Figure 10).

Implementation

In 1977, the limnological restoration plan of 1973 was unanimously accepted by.the Riksdag (Parliament) and in 1982, the Water Court and the Government granted permission to raise the water level, in two steps, by 1.4 m. The Swedish Environmental Agency was responsible for the implementation of the plan. Under their management, dikes, finally reaching a total length of 25 km, were projected around the lake and along tributaries. In addition to these, efforts were also made to get permission for construction of the Hertzman dike for reclamation of one of the lake's broad bays (Figure 10). The latter was, however, rejected by the Government. Because geological investigations had not been made at the suggested location of the dikes, the risks for leakage had not been assessed. Pleading the

Figure 9. Lake Hornborga. Demonstration of restoration methods and recolonisation of treated areas. Photos: Sven Björk.

1967: initial state.

1970: treated area.

1973: after cutting the reed, coarse detritus, stubble, rhizomes and root-felt have to be cleaned from the bottom.

1989: without raising the water level the treated area is recolonised by *Schoenoplectus, Phragmites* and *Typha*.

argument that depths greater than 0.4 m is a 'waste' of water with regard to the environmental demands of many duck species, together with that of the leakage risks, the Environmental Agency changed the plan for the restoration of Lake Hornborga. The projected dikes (with the exception of the one at the outlet) were abolished and the water was to be raised by only c. 0.8 m, with the first water rise by 0.2 m in 1993. The future development of the ecosystem will serve as an illustrative example of the importance in restorative planning of an holistic view in both space and time.

The further the ageing processes (filling in with sediment, peat and coarse detritus) have advanced in shallow lake basins, the more difficult it is to restore them by raising the water level. Ultimately, it becomes impossible. If the lake should be restored in a sustainable way, then that part not covered by revegetated plaur and bottom-rooted emergent plants must be large and deep enough to give the ecosystem the functional longevity as stressed in the original governmental directives for a restoration of Lake Hornborga, and not just give it a temporary appearance as an attractive waterfowl habitat. The open water should act as the 'heart and/or kidney' of the whole area, constituting an integrated part of the complex system, functionally powerful enough to preserve the vitality after restoration. Persistence

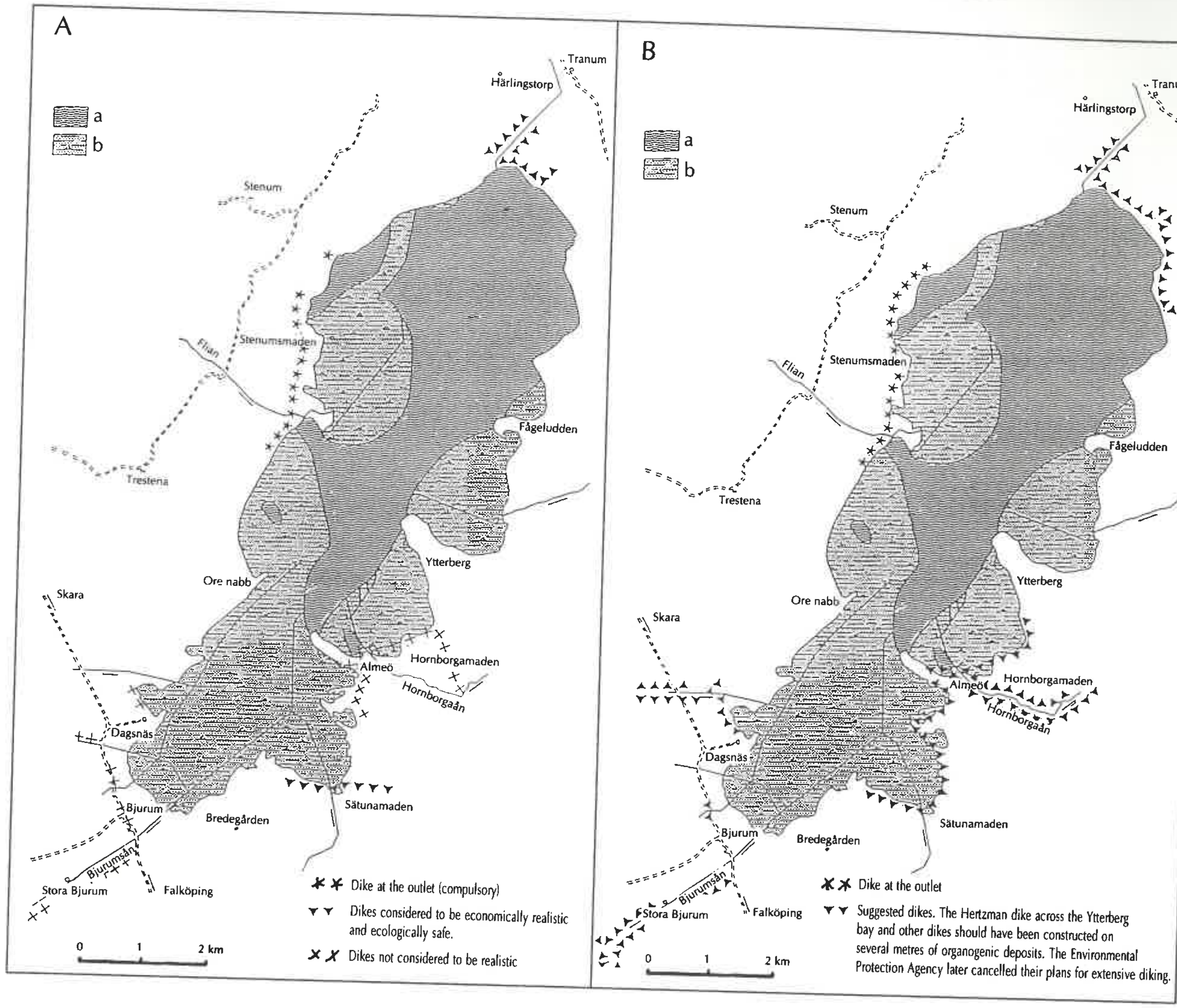


Figure 10. Lake Hornborga. Distribution of open water (a) with submerged and emergent vegetation and areas predetermined as plaur (b) after raising the water level according to restoration Plan/73. Map A: location of dikes according to Plan 73. Map B: dikes according to the Environmental Protection Agency, Stockholm:

of the results would, otherwise, be a matter of decades, not centuries. Every rise in the water level would, of course, cause a flourishing period for waterfowl, but the lower the rise the shorter the period.

It must not be forgotten that even a limnologically correctly restored Lake Hornborga is severely damaged by the former water level lowering and that the restored system must be given a structural and functional capacity to withstand the increased nutrient loading. It should, amongst other things, have the capacity to metabolise nutrient-rich water from the agricultural catchment area (the 1954 diked-in lake was supplied with such water for only a short spring period) as well as anoxic water pressed out from the plaur sections during periods of decreasing water level. In this connection, chain reactions, including periodic oxygen deficiency and nutrient release, will appear and will affect animal survival and fertilisation of downstream waters. Within the open water area, the bottom should consist of sediment cleaned from root-felt and coarse detritus. Furthermore, the open water should be large enough to enable the formation of blue ice capable of pressing and pushing to keep exposed shores open as the ice did in the lake before its degradation.

The evolution of shallow lakes/wetlands is, in several respects, dependent on the conditions in the extreme years. Most important are the effects of dry periods with low water levels, especially if these effects in regulated lakes cannot be compensated by the effects of extreme high water periods. In dry years, the artificially shallow lake overgrown by emergent vegetation will suffer from extra water losses due to intensive transpiration from the plant cover. Within the dry littoral zone, plants will transpire groundwater which, otherwise, would have supplied the lake. In addition to this, agricultural irrigation may reduce the supply of water during dry seasons. The summarising effect of low water level in extreme years is a lakeward expansion of emergent macrophytic vegetation, resulting in increased production of accumulating coarse detritus.

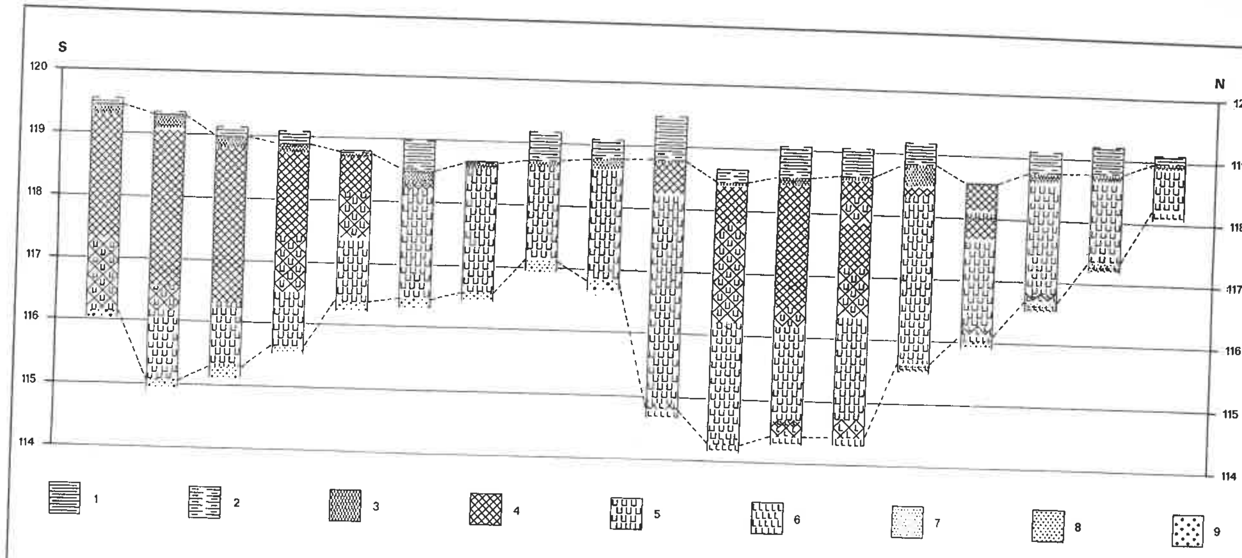
The Plan/73 for the restoration of Lake Hornborga also included measures to secure an even bottom in order to prevent sites from the accumulation of detritus and development of stands of emergent plants. Dikes along the drainage canals (Figure 1) had, therefore, to be levelled as part of the efforts to restore, as much as possible, the former wind-induced regime for in-lake water and ice movements.

Methods to increase water depth

When the water depth of a lake becomes too shallow, there are in general three possibilities to increase it: to raise the water level, to lower the bottom, or to combine these two measures. Under all circumstances the bottom has to be treated in order to get rid of the accumulated coarse plant material and the root-felt, a procedure which also results in a lowering of the bottom level. In Lake Hornborga, the removal of the root-felt and the dense stubble mat according to the Plan/73 would have lowered the bottom by *c.* 40 cm.

Lake Hornborga is situated in an area where the bedrock, in some parts, consists of limestone. The characteristic sediment deposit in the northern section of the lake is lake marl, i.e. calcium carbonate precipitated as a result of the photosynthetic activity of a rich submersed vegetation. The marl is of high quality and excellent for the liming of acidified areas instead of spreading finely-ground limestone/chalk, obtained from quarries, and crushed and ground to fine fractions. In Sweden, where inland waters in oligotrophic areas are suffering from acidification, the liming of acidified lakes, brooks and catchment areas is state-financed (*c.* US$11 million per year in the middle of the 1980s).

In lowered and completely drained lakes with lake marl as the characteristic sediment, modern automatic extraction methods have opened up possibilities for efficient sediment-mining aimed at the restoration of the lowered lakes, and the treatment of acidified land and waters. For Lake Hornborga, situated on the geological borderline between limestone



Key
1 = horizontally deposited stems of *Phragmites* etc.; 2 = coarse detritus; 3 = sediment with roots and rhizomes; 4 = detritus gyttja; 5 = lake marl; 6 = clay; 7 = silt; 8 = sand; 9 = gravel.

Figure 11. Lake Hornborga. Stratigraphical section from north (N) to south (S). Figures on both side of the diagram denote level above the sea (m). The upper dashed line connects the upper levels for consolidated gyttja and the lower line the upper levels for minerogenic soils.

and gneiss, lake marl is a characteristic sediment (Figure 11), and immediately southwest of the lake is the area of Sweden where the soils and waters are most severely affected by acidification. At the start of the Hornborga project, in the mid-1960s, acidification was not yet identified as an environmental problem and large-scale liming as a remedial measure was, of course, not introduced until later.

As pointed out at the beginning of this chapter, the ageing of shallow, productive lowland lakes is caused by the deposition of sediment and peat, produced in the ecosystem. In Lake Hornborga, the sediment in that part of the lake possible to restore to open water, consists of thick, homogeneous deposits of high-quality lake marl. The removal of the top layer of marl would not have changed the character of the bottom. The increase in water depth would, on the contrary, have had a real rejuvenating effect on the system. Thus, after liming had been given such a high economical priority in environmental protection financed by the state, there would have been an excellent opportunity to investigate this innovative alternative for reaching the goal of a sustainable Lake Hornborga through the combination of lowering the bottom and raising the water level, although this possibility did not appear until after half-time in the 28-year old project. Within the lake and wetland restoration-research sector there is an urgent need for the collection of experience of ecologically realistic, and economically sound, methods applied in demonstration projects.

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Food web management in reservoirs

Josef Matěna and Vojtěch Vyhnálek

Overview

More than 40 drinking water supply reservoirs in the Czech Republic have been managed since 1978 with the aim of reducing the numbers of planktivorous fish and so to reduce the phytoplankton biomass. Nine of these reservoirs were investigated in 1991 in order to check the efficiency of this management. The criteria used were the relationship between the total phosphorus concentration and the phytoplankton biomass, the species composition of zooplankton and the share of large (>0.7 mm) species of *Daphnia*. The results can be briefly summarised – that no positive effect of food web management measures was observed in any of the nine reservoirs investigated (Matěna, unpubl. data). Our explanation is that the reduction of planktivorous fish was insufficient. These results were confirmed by the qualitative investigation of the fish stock (Vostradovský, unpubl. data).

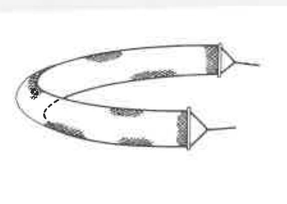
In the Czech Republic, a lot of experience on the effect of fish stock on the whole ecosystem has been gathered from studies on fish ponds (Fott *et al.* 1980). All these results confirm that the positive effects of food web management can be more easily achieved in smaller, fully controlled, shallow water bodies and can be maintained there for a longer time.

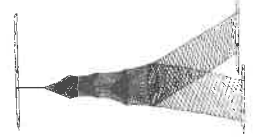
The two large-scale field experiments described below aim to illustrate the application of food web management in reservoirs and varied effects attained there due to the complexity of this method as described in Chapters 5 and 7 (Food web relations and Food web management).

Římov Reservoir, Czech Republic

The best documented example of the development of fish stock in a reservoir is the Římov reservoir, which was intensively investigated since its impoundment in 1979. This reservoir, located about 140 km south of Prague, has a maximum depth of 43.9 m, surface area of 210 ha, and retention time of 90 days. The three dominant fish species – perch (*Perca fluviatilis*), roach (*Rutilus rutilus*) and common bream (*Abramis brama*) – account for more than 90% of both total numbers and biomass of fish. An extensive programme was established, aimed at the reduction of planktivorous fish during 1985–1989. The application of fyke-nets (see Table 1) proved to be very effective in the elimination of perch during their spawning period (Kubečka 1992). Shore seining was used to reduce adult cyprinids. While daytime shore seining is often inefficient, night-time seining remains one of the simplest and cost-effective methods (Kubečka 1993a). Probably the most effective method of reducing the populations of dominant cyprinids (roach, common bream) was the lowering of the water level in the reservoir after the mass spawning of both species. The spawning takes place mainly in shallow water near the shore and, thus, after the water level decreases, a substantial

Technique		Catch (kg)/stock (individuals)	Effect*
Seining and fyke nets		24,000	20 %
Stocking of piscivorous species	pike perch (*Stizostedion lucioperca*) 1+	170,000	
	pike perch 2+	320	
	pike perch S	75	
	pike (*Esox lucius*) 1+	22,000	
	pike 2+	45	10%
	wels (*Silurus glanis*) 1+	1,300	
	wels 2+	480	
	asp (*Aspius aspius*) 1+	40,000	
Water level decrease (0.3 – 0.5 m) after spawning of cyprinids			70%


Beach seine without bag – used for fishing at shallow littoral stations – an active gear


Fyke net – used mainly for removal of perch – a passive gear

Key

* Effect represents the contribution of a given technique to the total reduction of fish biomass
1+ one year old fingerling
2+ two years old fingerling
S spawner

Table 1. The effectiveness of measures to reduce the stock of planktivorous fish in Římov reservoir (fish biomass decreased from 500–600 kg ha^{-1} in 1982–1985 to 100–150 kg ha^{-1} in 1988–1991). Kubečka, unpubl. data.

proportion of eggs dry out. Moreover, the reservoir was intensively stocked with piscivorous fish species – pike perch (*Stizostedion lucioperca*), pike (*Esox lucius*), asp (*Aspius aspius*) and wels (*Silurus glanis*). As a result, the total fish stock decreased from about 600 kg per ha to about 100 kg per ha (Figure 1).

The effectiveness of all the measures applied to reduce the stock of planktivorous fish is given in Table 1. At the same time a gradual shift in species composition was observed. Perch formed about 75% in terms of numbers in 1985, but by 1992 its share had gradually dropped to 3%. On the other hand, the proportion of cyprinids increased to 60% (roach) and 30% (common bream). This phenomenon seems to occur regularly in newly impounded reservoirs (Kubečka 1993b). The decrease of perch was, most likely, accelerated by the selective fyke-net catching. The results of food web management in Římov reservoir can be summarised as follows:

- No effect of the reduced fish biomass on the total zooplankton biomass was observed (Figure 2).
- No change in the species composition of the zooplankton was achieved. The small filtrator, *Daphnia galeata*, remained dominant and was not replaced by the larger *Daphnia pulicaria*, as expected.
- A distinct effect on the size structure of zooplankton was observed (Seďa *et al.* 1989). The share of large daphniids (retained on a mesh-size of 0.71 mm) increased from 1% to 10% (Figure 3). This result can be regarded as the only positive effect (*cf.* Chapter 5) of the food web management experiment in Římov reservoir.

- No effect of the reduced fish biomass was observed on the further (phytoplankton) trophic level. The seasonal averages of phytoplankton biomass remained almost unchanged between 1985–1991 (Figure 4). The observed, between-year fluctuations of phytoplankton biomass were most likely related to differences in water discharge (Komárková 1993).

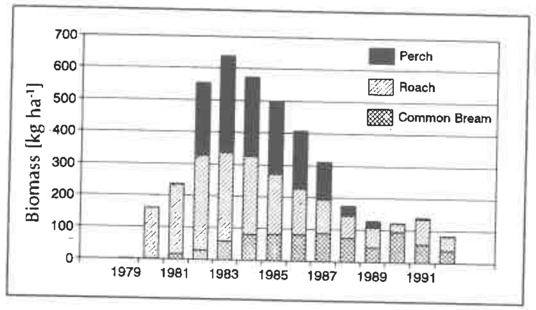


Figure 1. Biomass of planktivorous fish in Římov Reservoir during 1979–1992.

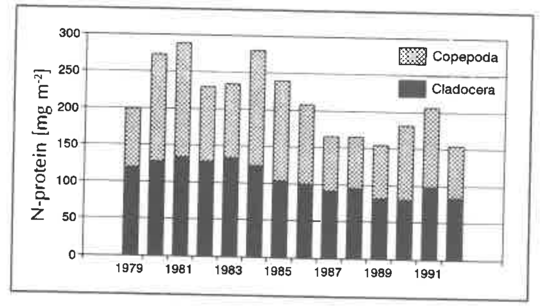


Figure 2. Biomass of zooplankton (Cladocera, Copepoda) expressed as concentration of protein nitrogen under 1 m^2 of water surface in Římov Reservoir during 1979–1992.

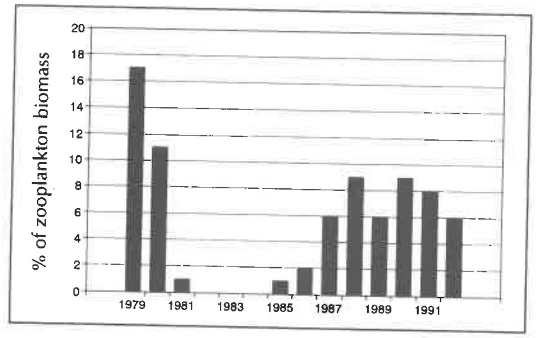


Figure 3. Share of Cladocera retained on a 0.71 mm mesh in total zooplankton biomass in Římov Reservoir during 1979–1992.

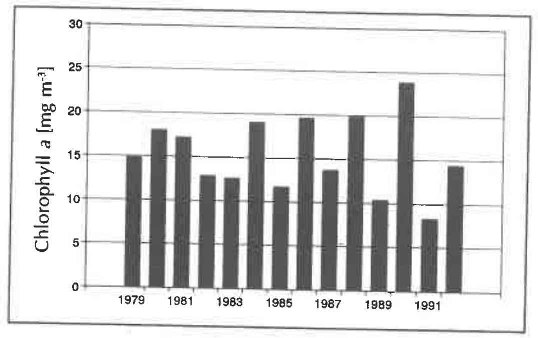


Figure 4. Concentration of chlorophyll *a* in upper 4 m of water column in Římov reservoir during 1979–1992.

Conclusions

It seems that the critical biomass of fish stock for obtaining a positive effect through food web management lies lower than the 100 kg ha^{-1} achieved in Římov reservoir. According to some data from the literature, it might even be lower than 50 kg ha^{-1} (Mills & Forney 1983, Walker 1989). From this point of view the results from London reservoirs (see p. 99), where the very low biomass of fish stock corresponds well with the occurrence of large daphniids and low phytoplankton biomass in spite of the high level of nutrients, are of particular interest. Nevertheless, it is necessary to point out the further management measure – continuous mixing of the whole water column – that contributed substantially to the positive results obtained there.

Hubenov Reservoir, Czech Republic

Hubenov reservoir, located about 100 km east of Prague, has a maximum depth of 17.5 ha, surface area of 51.6 m, and retention time of 167 days. The reservoir, immediately after impoundment in 1972, was stocked with salmonids – brown (*Salmo trutta*) and rainbow trout (*Oncorhynchus naykis*). Thus, a delay of the massive development of planktivorous species (mainly small perch, roach and common bream) was achieved. The population of salmonids declined, however, quite quickly; the last trout were observed in 1978. Their mortality was caused mainly by diseases.

Table 2. Selected hydrobiological parameters (seasonal averages) during periods with different fish stock in Hubenov reservoir. After Hrbáček *et al.* 1978, 1986.

Period	Zooplankton N protein mg dm^{-2}	Cladocera >0.71 mm %	Secci depth m	Chl *a* µg.l^{-1}	Total P µg.l^{-1}
1975–1979	1.65	24.1	3.8	6.5	19.9
1980–1983	1.26	16.5	3.3	13.7	17.6

Note: Period 1975–1979 = low fish biomass; period 1980–1983 = high fish biomass.

Subsequently, planktivorous species became dominant due to the propagation of perch and later also roach (Hrbáček *et al.* 1986). At the same time the fish biomass increased. Unfortunately, no exact data about the fish biomass are available.

Results

The years 1975–1979 represent the period with a low biomass of fish (period of successful food web management) while the following years 1979–1983 represent the period with a high biomass of fish. During the period of low fish stock, the zooplankton biomass, the ratio of large daphniids to total zooplankton, and the water transparency were higher (Table 2). The biomass of phytoplankton (measured as chlorophyll *a* concentration) was about half of that at high fish stock levels, with a nearly unchanged concentration of the limiting nutrient (phosphorus). At low fish stock, filtratory zooplankton, dominated by the large species *Daphnia pulicaria*, effectively controlled the development of phytoplankton. The effective food web management is indicated by the fact that the concentration of chlorophyll *a* lies outside the 95% confidence limits of the Dillon-Rigler's relationship (Dillon & Rigler 1974, Hrbáček *et al.* 1978), being about half of that predicted according to the phosphorus concentration.

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APPENDIX A

Glossary

ACIDIFICATION – Decline of pH and alkalinity in groundwater and inland water bodies since pre-industrial times, primarily due to emissions of sulphur oxides, nitrogen oxides and ammonia to the air or as a result of industrialised forestry.

ALGAL BLOOM – Seasonal appearance of dense phytoplankton colouring the water.

ANOXIC – Void of molecular oxygen.

BENTHOS – The community of organisms associated with the bottom of a water body. Benthic plants and animals occur on or in the submerged substrate.

BIOCOENOSIS – The community of organisms inhabiting a biotope.

BIOTOPE – 'Place of life'. The totality of the environmental conditions under which a biocoenosis exists.

BOTTOM-UP RELATIONS – The series of basic dependence in the food web: of plant nutrients for the production of plants, and plants for plant-eating organisms (primary consumers), and successively higher levels of consumers (secondary, tertiary consumers). *Cf.* Top-down relations.

CATCHMENT AREA – In this book, the denomination of the area from which water is drained to a lake, wetland, etc. Drainage basin and watershed are sometimes given the same meaning.

CIRCULATION PERIOD – Cool, temperate, deeper lakes are characterised by total circulation of the water column during two periods of the year, in spring and autumn. In summer and winter they are thermally stratified. *Cf.* epilimnion, metalimnion and hypolimnion.

DENITRIFICATION – The bacterial reduction of nitrate to molecular nitrogen (nitrogen gas).

ECOSYSTEM – A natural unit which includes both living and non-living components, forming a system characterised by the interaction and recycling of matter. Lakes and wetlands are good examples of such functional units.

EMERGENT MACROPHYTES – Aquatic plants that are rooted in the bottom and with leaves and reproductive organs rising above the water surface.

EPILIMNION – The upper, warm water layer in deeper lakes with summer stratification. *Cf.* circulation period, metalimnion and hypolimnion.

EUTROPHIC – Water bodies with a high supply of nutrients and therefore characterised by high organic production. *Cf.* oligotrophic.

EUTROPHICATION – The enrichment of water bodies with nutrients, such as phosphorus and nitrogen, causing an accelerated productivity.

EXTERNAL NUTRIENT LOADING – The supply of nutrients to a water body from the catchment area.

FLOATING-LEAVED MACROPHYTES – Aquatic plants rooted in the bottom with submersed and floating leaves. Reproductive organs are floating or aerial.

FOOD WEB MANAGEMENT – Efforts to govern the structure and function of an aquatic ecosystem through reduction or favouring of selected species or groups of species.

GROUNDWATER RECHARGE – Infiltration of rainwater and surface water to water-bearing soil and bedrock layers.

GYTTJA – Fine to coarse lake sediment with high content of organic matter, also including inorganic precipitations and mineral particles.

HOLISM – Emphasising the value of a comprehensive view. In ecological research and management a HOLISTIC APPROACH means to adopt a comprehensive view on, for example, a lake ecosystem as a whole and its dependence on external and internal factors and processes.

HYPOLIMNION – The lower, cold water layer in deeper lakes with summer stratification. *Cf.* circulation period, epilimnion and metalimnion.

INTERNAL NUTRIENT LOADING – The supply of nutrients from the sediment to the water column of a lake, most typically occurring in connection with oxygen depletion at the water-sediment interface.

LAKE MARL – White to grey lake sediment of precipitated calcium carbonate, poor in organic matter.

LEACHING – The washing of ions out of soil and bedrock by surface water and groundwater.

LENTIC – Standing water; living in lake pond, etc. *Cf.* lotic.

LIMNOLOGY – The science of inland water ecosystems.

LITTORAL ZONE – Extends from the shore to a water depth where the light is barely sufficient for rooted aquatic plants to grow. *Cf.* pelagic zone.

LOTIC – Running water; living in brook or river. *Cf.* lentic.

METALIMNION – The layer of water of vertically, rapidly changing temperature in lakes during summer stagnation, above which is the epilimnion and below is the hypolimnion. Syn. Thermocline. *Cf.* circulation period, epilimnion and hypolimnion.

MATTER LOSS – Irreversible flow of charges, i.e. dissolved ions (base cations and anions) via surface waters to the sea, measured as water flow in the river multiplied by matter concentration measured as conductivity.

OLIGOTROPHIC – Water bodies with a low supply of nutrients and therefore characterised by low organic production. *Cf.* eutrophic.

PELAGIC ZONE – The open water outside the littoral zone in lakes. *Cf.* littoral zone.

PERIPHYTON – Community of micro-organisms that live attached to submerged substrate (e.g. rocks, aquatic plants etc.).

PLAUR – Stand of aquatic macrophytes with the root-felt floating at the water surface. Plaur formations develop when gas bubbles collect to such a degree in and beneath the root-felt that this is loosened from the bottom and raised to the water surface.

PROFUNDAL – The bottom zone of a lake outside the littoral zone with macrophytes, i.e. beneath the pelagic zone of open water.

REDEVELOPMENT – Activities aiming at a general upgrading of the environment according to given goals for the sustainable use and utilisation of ecosystems.

RESTORATION – Measures taken in ecosystems, like lakes and wetlands, to adjust the structure and govern the function of a specific damaged system. In practical environmental protection and management, the meaning of the term restoration often implies the re-establishment of an ecosystem – according to local or regional interests – for those purposes for which it used to be suitable before its man-made degradation.

RHIZOSPHERE – The soil surrounding the root systems of plants.

SEDIMENT – Deposits of minerogenic and organogenic particles separated from water or air, often settled in layers of sorted fractions.

SHALLOW LAKE – Common designation of a water body characterised by its littoral organism communities. *Cf.* littoral zone.

STAGNATION PERIOD – A period of thermal stratification in lakes; summer stagnation and winter stagnation. The opposite is the periods of total circulation in cool temperate lakes.

STRATIGRAPHY – Description of the arrangement and type of layers (strata) in soil (e.g. sediment) and bed-rock.

SUBMERGED (SUBMERSED) MACROPHYTES – Aquatic plants, rooted or rootless, whose vegetative parts are under water. Reproductive organs are aerial, floating or submersed.

SUSTAINABLE ECOSYSTEM – An ecosystem characterised by the natural development of structure and function and which does not degrade because of man's interference. The term sustainable came into common use through the man-centred work of the World Commission on Environment and Development 'Our Common Future' (UN 1987). According to this commission 'Sustainable development is development that meets the needs of the present without compromising the ability of future generations to meet their own needs'.

THERMOCLINE – *Cf.* metalimnion.

TOP-DOWN RELATIONS – Influence exerted by higher trophic level on lower levels in the food web. *Cf.* Bottom-up relations.

WETLAND – Wetlands are biotopes, or sections of the landscape, whose general appearance and habitat, as well as their plant and animal life, are, for at least part of the year, to an essential degree characterised by the factor water.

APPENDIX B

Biography of authors

SVEN BJÖRK, born in 1927, studied biology and geology and obtained his Ph.D. in limnology from Lund University. During the late 1950s and 1960s, he devoted a lot of time to regio-limnological studies and the large variation among lake ecosystems. As the pollution of inland waters became more and more serious in Sweden in the 1950s, he was asked to join the Technical Research Council for training in water pollution research. A systematic survey of the limnological conditions in drainage systems in south-east Sweden revealed the serious misuse of inland waters which had been severely hit by water level lowering and pollution. The combination of limnological training and insight into and contact with the technical sector inspired him to pursue research within the field of lake restoration, i.e. to investigate the possibilities to treat and govern the development of damaged lake ecosystems. The first plan to restore a lake was presented in 1966 and the definition of the term 'lake restoration' was published in 1967. As senior lecturer and, since 1968, professor of limnology at Lund University, he has organised and been the leader of teams of ecologists collaborating with technologists on the restoration and management of lakes, wetlands and running waters in Sweden and abroad (in Europe, Tunisia, Brazil, Iraq, Iran, Colombia, Jamaica, China). Lake Trummen, in Sweden, was the first project to demonstrate how to restore a shallow lake irreversibly damaged through pollution. Prof. Björk also developed the methods and designed the plan (Plan/73) for sustainable restoration of the drained Lake Hornborga. Demonstration projects and several publications illustrate the excellent opportunities for basic ecosystem research and training of limnologists involved in the restoration activities of water bodies. In his contributions to limnology, Sven Björk has strongly emphasised a holistic approach to the study of ecosystems, with a special interest in ecosystem development in both time and space. His list of publications include about 110 papers. He is a member of the Royal Swedish Academy of Sciences, Knight of the Royal Order of the Northern Star, appointed Doctor *honoris causa* at the University of Munich, and got the Bruno H. Schubert Prize at Frankfurt a.M. in 1993. *Present address:* Källadal, Gislöv, S-27200 Simrishamn, Sweden.

VÁCLAV HAUSER graduated in electronics at the Technical University in Prague (1983). He is currently a research assistant at the Institute of Botany at Třeboň. Ing. Hauser was responsible for digital mapping of the sediment in the Vajgar restoration project and supervised the suction dredging. *Present address:* Institute of Botany, Dukelská 145, 379 82 Třeboň, Czech Republic.

JAROSLAV HRBÁČEK, born in 1921, graduated with a thesis on the ecology and physiology of waterbeetles (Hydrophilidae). In 1948, he started as assistant to the Chair of Zoology at the Faculty of Natural Sciences of Charles University in Prague. With a group of students he studied oxbows and pools in the inundation area of the river Elbe and, later, investigated

fish ponds in South Bohemia. These studies resulted in papers on the relations between fishstock, invertebrates and phytoplankton, which are referred to in many textbooks. In 1958, he became head of the Hydrobiological Laboratory at the Biological Institute of the Czechoslovak Academy of Sciences. From that time, he has made complex studies on reservoirs, then a new type of water body in the former Czechoslovakia, and in particular on the effects of a cascade of reservoirs on water quality and the resultant changes in the biota. Present studies of Dr. Hrbáček are concerned with the biota in the inundation area of the river Lužnice in South Bohemia. *Present address:* Hekrova 820, 149 00 Praha 4, Czech Republic.

JOSEF MATĚNA graduated in limnology at Charles University in Prague (1975) and acquired his Ph.D. in limnology from Charles University in 1983. His professional experience includes three years at the Fisheries and Hydrobiology Research Institute at Vodňany, and six years at the State Fishery Enterprise in České Budějovice. In 1988, he joined the Hydrobiological Institute, Academy of Sciences of the Czech Republic, where he is head of the Department of Production Processes since 1991. His main fields of research have been the ecology of macrozoobenthos, taxonomy and ecology of Chironomidae, and fish ecology in reservoirs. Current research interests are the feeding biology of freshwater fish, and biotic interrelations between fish and lower trophic levels (zooplankton, phytoplankton) of freshwater ecosystems. *Present address:* Institute of Hydrobiology, Na sádkách 7, 370 05 České Budějovice, Czech Republic.

JAN POKORNÝ graduated in biology and chemistry at Charles University in Prague (1969). His expertise lies in the photosynthesis of water macrophytes and their functioning in eutrophic water bodies. Dr Pokorný became involved in restoration activities of degraded wetland ecosystems and initiated restoration works in eutrophicated fish ponds in South Bohemia. In cooperation with Lund University he designed and supervised the restoration work in Vajgar (in Jindřichův Hradec) and stimulated the development of suction dredging technology. Since 1992, Dr Pokorný has been head of the Section of Plant Ecology at the Institute of Botany at Třeboň. *Present address:* Institute of Botany, Dukelská 145, 379 82 Třeboň, Czech Republic.

STEVE RIDGILL obtained his B.Sc. in botany and zoology from University College of Wales, Aberystwyth, U.K. in 1983. With a strong background in physics and mathematics from previous studies at Imperial College, London, he has combined this interest with biological systems, ecology and information systems. For the last six years he has worked as a research scientist at the The Wildfowl & Wetlands Trust, Slimbridge, UK. *Present address:* IWRB, Slimbridge, Gloucester, GL2 7BX, United Kingdom.

WILHELM RIPL, born in 1937, started his university studies with technical chemistry in Vienna. He then moved to the University of Lund, Sweden where his scientific interest focused on limnology. As a member of the research teams dealing with restoration of Lake Trummen in Sweden, Lac de Tunis in Tunisia, and Lagoa Rodrigo de Freitas in Rio de Janeiro, he was fascinated by the possibilities to govern lake ecosystems to reach defined goals. The first restoration method developed by himself – later named the Riplox method

– was applied as a whole-lake experiment in a hypertrophicated Swedish lake. The project resulted in a lake suitable for swimming and a doctoral thesis presented in Lund in 1978. Dr Ripl designed several of his own lake restoration projects in Sweden before, in 1979, he was appointed Professor of Limnology at the Berlin Technical University and established a team of young limnologists working within the restoration sector of limnology. Results from basic research especially on dynamic processes at the sediment-water interface inspired him to further improvement of in-lake measures for control of internal loading in lakes. In recent years, he has stressed the importance of a comprehensive view of the landscape and has, in interdisciplinary projects, devoted much interest on restoration and management of whole catchment areas. The overall objective is, first of all, to define and evaluate parameters determining a sustainable development and stability of the landscape, where an ecologically sound management of the water cycle constitutes the keystone. *Present address:* Technical University, Institute of Limnology, Hellriegerstr. 6, G-14 195 Berlin 33, Germany.

KAREL ŠIMEK graduated in limnology at Charles University in Prague (1979), from where he received his Ph.D. in 1986. He is currently head of the Department of Biological Self-Purification Processes and Water Chemistry, at the Hydrobiological Institute in České Budějovice, Czech Republic. His main fields of specialisation are microbial ecology including bacteria-protozoan interactions, and limnology with regard to carbon flow through microbial food webs to higher trophic levels. His current research deals with the factors regulating bacterial abundance, production and activity in aquatic ecosystems, and protozoan bacterivory and algivory. *Present address:* Institute of Hydrobiology, Na sádkách 7, 370 05 České Budějovice, Czech Republic.

BO VERNER, limnotechnologist. After technical studies and service in the Royal Swedish Navy, he was, in 1964, employed as an engineer at the Atlas Copco Central Laboratories of Physics in Stockholm. In close cooperation with hydrologists, limnologists and environmentalists, Bo Verner has invented and developed a wide range of technical equipment for the management, restoration and protection of lakes, running and coastal waters. These technological achievements include the development of hypolimnetic aerators, the technical facilities for the Riplox and Contracid methods (treatment of acidified lake's sediment with sodium carbonate), pneumatic barriers preventing oil spillages and siltation, devices for the prevention of ice formation and for preventing of jelly fish from entering water intakes. His inventions have been introduced worldwide, and as an ecotechnologist bridging the gap between ecological theory and practical application, he has cooperated in redevelopment and restoration projects in Europe, Asia, Africa and the Americas. During 1972–1984, he worked at Atlas Copco, Antwerp; since the late 1980s, he has run his own company. *Present address:* Bo Verner AB, Frejgatan 46A, S-11326 Stockholm, Sweden.

VOJTĚCH VYHNÁLEK graduated in limnology from Charles University in Prague (1979), where he also obtained a Ph.D. in 1987. He is currently scientific secretary at the Hydrobiological Institute in České Budějovice, Czech Republic. He has specialised in phytoplankton ecology, especially nutrients-phytoplankton-zooplankton interactions, phosphorus dynamics in lakes and reservoirs, and eutrophication. His current research interests are with regards to factors controlling algal populations in freshwater ecosystems

– their growth, limitation, and losses. *Present address:* Institute of Hydrobiology, Na sádkách 7, 370 05 České Budějovice, Czech Republic.

KLAUS-DIETER WOLTER obtained his diploma in biology in 1986. His diploma work concerned the physical, chemical and bacteriological processes in lake sediments. During his thesis (1986–1991) Wolter was dealing with palaeolimnological methods, especially diatom analysis and the interpretation of physical, chemical and biological parameters. During this work, he gained an impression of the human impact on ecosystems and the importance of long time-scale processes, as well as a synthesis for the interpretation of processes in their catchment areas and lakes. Since 1991, he has been mainly working on lake restoration, water management, and ecological concepts. Central points in this work are the holistic view to ecosystems as shown by the ETR-model and the conclusions which can be drawn from this model for ecological and water management. *Present address:* Association for Water Management, Frauenstr. 6, G-12207 Berlin 45, Germany.

EDITOR

MARTINA EISELTOVÁ graduated in agronomy (1988) at the Agriculture University in Prague, and in limnology (1990) at Charles University, Prague. During her cooperation with the Institute of Botany at Třeboň, she studied the impact of eutrophication on aquatic vegetation, especially the development and ecophysiology of filamentous algae in highly eutrophic fish ponds in the Třeboň Biosphere Reserve. Since 1991, she has held the position of training coordinator at the International Waterfowl and Wetlands Research Bureau, based in Slimbridge, UK. She has been responsible for the establishment of a coordinated wetland management training programme, primarily in Central and Eastern Europe. Under this programme, she has organised several training courses for wetland researchers and managers, dealing with the restoration of degraded lakes, rivers and their riparian zones, and emphasising the need for sustainable development of the landscape as a whole. Compilation and editing of the present publication has been completed as a component of the training programme. *Present address:* IWRB, Slimbridge, Gloucester, GL2 7BX, United Kingdom.

APPENDIX C

Companies involved in lake restoration

AUSTRIA

Aquaterra-Consult-Gesellschaft m.b.H.
Rothkirchgasse 4/4/20
A-1120 Wien
Tel: +43 1 813 1967
Fax: +43 1 813 1967
Planning and accomplishment of restoration projects.

CZECH REPUBLIC

Institute of Botany
Dr Jan Pokorný
Dukelská 145
379 82 Třeboň
Tel: +42 333 2343
Fax: +42 333 2391
Consultancy and planning of restoration projects.

Envi Ltd
Dukelská 145
379 82 Třeboň
Tel: +42 333 2343
Fax: +42 333 2391
Precision suction dredging.

DENMARK

Seiga Harvester Co.
Mr. Gösta Larsson
Funkiavej 45
DK-2300 Köpenhamn S
Tel: +45 31 554265
Sale of amphibious vehicles and equipment for lake and wetland restoration.

J.S.P. Maskinfabrik
Mr Jörgen Pedersen
Sverigesvej 6
DK-8450 Hammel
Tel: +45 86 969511
Design, construction and production of amphibious vehicles and equipment for lake and wetland restoration.

FINLAND

Lännen Engineering Oy
SF-27820 Iso-Vimma
Tel: +358 38 3961
Fax: +358 38 3963122
Construction and production of multipurpose (dredging etc.) machines for rehabilitation of lakes and wetlands.

GERMANY

Gesellschaft für Gewässerbewirtschaftung mbH
Frauenstrasse 6
G-12207 Berlin
Tel: +49 30 76881 40
Fax: +49 30 76881 30
Planning and accomplishment of restoration projects.

SWEDEN

Limnoteknik AB
Mr Emil Cronqvist
Västergatan 33
S-274 00 Skurup
Tel: +46 411 405 95
Design and construction of machines for lake and wetland restoration and management (control of macrophyte vegetation, etc.), accomplishment of restoration projects.

Skanska AB
S-211 02 Malmö
Tel: +46 40 144000
Dredging of lakes.

Bo Verner AB
Frejgatan 46 A
S-113 26 Stockholm
Tel: +46 8 327525
Fax: +46 8 590 88320
Planning and accomplishment of lake restoration projects.