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The dynamics of Lake, Bog & Bay–Consequences of exposure to man related to final storage of spent nuclear fuel.

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THE DYNAMICS OF LAKE, BOG & BAY - CONSEQUENCES OF EXPOSURE TO MAN RELATED TO FINAL STORAGE OF SPENT NUCLEAR FUEL

Abstract

The natural ageing of the environment, much shorter than a possible contineous release from a repository, results in uncertainties as regards to consequences of exposure to man related to final storage of spent fuel.

The Phase I project summarizes the identification of such factors and parameters of predominant importance for an assessment of possible radiological consequences in a normally revolving biosphere.

Three major areas are described: geomorphology and residence time such as phenomena associated with the formation of lakes in Sweden formed by the latest inland ice, lake development in terms of uptake of different elements and radionuclides as a function of the chemical composition of the water in a recipient and some aspects of sedimentation in lakes and coastal waters.

The chemical environment is also studied in the evolution of a lake or a Baltic bay into farming land or peat land. The most important parameters to the behaviour of trace metal ions seem to be pH, Eh, ionic strength and content of complex formers.

Finally, the human impact during the evolution of a lake or a bay is discussed.

The study also includes recommendation for a reference field test area and for laboratory studies of the chemical properties of radionuclides in sediments in a Phase 2 project.

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THE DYNAMICS OF LAKE, BOG & BAY - CONSEQUENCES OF EXPOSURE TO MAN RELATED TO FINAL STORAGE OF SPENT NUCLEAR FUEL

BACKGROUND

With groundwater as a pathway radioactive material may reach the biosphere and man. The characteristics of the recipient of the biosphere is therefore of major importance for the dispersion of radionuclides.

The natural ageing of the environment - much shorter than a possible contineous release from a repository - results in uncertainties as regards to the assessment of possible radiological consequences.

The purpose of this initial study aims to define parameters of predominant importance concerning the transfer of radionuclides from groundwater to man and quantitively determine the changes.

Phase 1 of the project also includes recommendation for a reference field test area of Sibbo- and Trobbofjärden and laboratory studies of the characteristics of the chemical properties of radionuclides in sediment.

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To launch a project of this kind initiated some
brain-storming activities. Approaches were many
in order to include all scopes of ideas but
after having structured the major processes
affecting the evolution of lakes and given
priority parameters governing the various
development stages the point of interest focused
on three major areas of importance;

Geomorphology & Residence time	Lake Development - Flora & Fauna	Sedimentology: accumulation & resuspension

These three areas are described in the following
reports:

(1)   Lake and Coastal Zone Evolution during
        Postglacial Time (B Sundblad)

(2)   The Ageing of a Lake Changes in Plant
        and Animal Life. (P O Agnedal)

(3)   Aspects on Sedimenation in Lakes and
        Coastal Waters (S Evans)

The chemical environment is a contineous process
during the development of lakes and coastal
waters. The trace metal chemistry release into a
lake, river or sea is therefore treated in a
separate report:

(4)   Chemical Properties of Lake Sediments
        and Peat of Importance to Trace Metal
        Behaviour (K Andersson)

Finally, a fifth study describes some aspects on
the evolution effects on man by fertilization
and terrestrialization:

(5)   Human impact during the evolution of a
        lake or a bay (A-B Wilkens).

CONCLUSIONS OF REPORTS

Appendix 1. Lake and coastal zone evolution
during postglacial time (Björn Sundblad)

Most of the recent lakes in Sweden are
originally formed by the latest inland ice.
However, the lakes are occasional phenomena.
Many processes affect the evolution of the lake.
Some of these are discussed in this report.

The lake evolution goes through several stages,
where the importance of the different processes
will change.

The water is the carrier system for suspended
material, d_i solved substances, biomass etc.
Because of that it is important to know the
residence time during different stages of the
lake evolution.

The sedimentation is also dependent on the water
transport to and within the lake. The unknown
relation between sedimentation rate and residence
time must be solved, to be able to find the
transfer coefficients between water and sediment,
and their variations. The water transport also
affects the horizontal and vertical distribution
of the sedimentation.

Another important parameter is the stratification
of the lake, mainly the thermal stratification.

A reduced vertical water exchange will for example affect the oxygen condition at the bottom.

At the coastal areas the transition from brackish water to fresh water is different from the pure lake systems for stratification, sedimentation etc. Because of that the transition period must be treated separately in the lake evolution. The Sibbofjärden-Trobbofjärden area are suitable to study these differences.

Appendix 2. The ageing of a lake - changes in plant and animal life (P O Agnedal)

The uptake of different elements and also of radioactive nuclides varies with the chemical composition of the water in a recipient. The uptake varies with different species of both plants and animals.

It is therefore important to have a good knowledge of conditions such as pH, redox-potential, nutrient contents, composition of flora and fauna in order to be able to calculate how much of an element will be accumulated by plants and animals and how much will be transported to the sediment after decomposition.

The uptake is also depending on the total production of organic material in the different trophic layers.

As has been reported the production can vary very much both during the year and in each trophic layer.

In an American lake the production of rooted vegetation was 50 kg dw/har per year and of phytoplankton 2560 kg dw/har. The fish catch varies from less than 5 kg/har up to 20 kg/har depending on the type of lake.

In order to be able to calculate the amount of an element bound in each trophic layer a knowledge of the production seems necessary.

It can therefore be deemed necessary to study the transport of some elements in the different lake types and also in the peat and from these studies try to draw conclusions of the behaviour of those radionuclides which might be of interest.

Comparably laboratory studies of the transport mechanisms should also be investigated step by step in laboratory experiments where the lake types are simulated.

Appendix 3. Aspects of sedimentation in lakes and coastal waters (S Evans)

Theoretical models of ecological processes in estuarine and lacustrine environments call for data suited to calculate sediment budgets in relation to space and time. Accurate long-term forecasts on sediment growtn and the subsequent accumulation of elements in the deposits laid down require knowledge about production rates and sources of particulate matter, sedimentation rates, affinity of elements to solids, mobility of elements when deposited on the sediment bed etc. Efforts to model sediment dynamics must be based on parameters based on field measurements together with data extrapolated from laboratory experiments.

The sediments are subjected to various short-term pertubations such as seasonal variations in the sedimentation rate, resuspension by near bottom currents and bioturbation due to the micro- and macrobiological activity occurring within the deposits. Consequently, the sediments from a defined layer only provide information integrated over longer periods. The resolution in time is highly governed by the sedimentation rate which varies from about 50 g m^{-2} in oligotrophic waters to about 3 000 g m^{-2} in eutrophic waters. Thus the measurement of the seasonal sedimentation rate is of crucial importance for the interpretation of the dynamics of the bottom boundary layer of a water body. Furthermore, the physical and biological factors disturbing the deposits laid down will to a large extent govern the chemical status of the sediment bed.

The interstitial water within the sediment layer serves as a carrier of elements between the sediments and the overlaying water. This will enable chemical species to migrate within the sediment as well as across the sediment-water interface. However, a limited number of physical sediment parameters such as water content, organic content, bulk density and grain size will generally be enough in order to make an accurate classification of the sediment. Most other physical sediment parameters like porosity, permeability and compaction may be defined in terms of these basic parameters.

Appendix 4. Chemical properties of lake sediments and peat of importance for trace metal behaviour (K Andersson)

In the evolution of a lake or a Baltic bay into farming land or peatland a wide range of

possible chemical conditions may be expected to occur within bottom sediments and in the peat.

There are some parameters in the interstitial water which are of large importance to the behaviour of trace metal ions. The most important seems to be pH, Eh, ionic strength and content of complex formers.

pH may vary from 3.5 to over 8. The low values are most likely to occur in peat and in acid soils, while pH values of 7 to 8 are the most common in sediments. Also in some types of peat pH up to 7 may occur. Since the possible sorption of cations usually is increasing with pH, especially for easily hydrolyzed ions, the systems with high pH may be expected to exhibit a lower mobility of trace cations.

The absolute value of the redox potential, Eh, is a quantity that is difficult (or impossible) to determine. A significant difference between oxic and anoxic conditions may although be observed. In sediments Eh changes may occur over the year, especially in the surface layer due to variations in the aereration and oxygen consumption of the overlaying lake water. Deep sediments and peats also tend to be depleted of oxygen. The behaviour of redox sensitive metals (e.g. Tc, U, Pu) may be determined by the redox conditions to a very large degree, while for others there is one predominant valence state and the redox conditions will only be of importance if they influence the properties of the solid or water chemistry.

The ionic strength is also varying over a wide range for the water types in question. The amount of dissolved ions may be from some mg/l

up to 16 000 mg/l. This means that the sorption
may vary very much for uncomplexed cations
depending on the type of water. Also the complexa-
tion is dependent of the ionic strength (stability
constants decrease with increasing ionic strength).

In all the water types discussed, the content of
potential complex formers may be quite high.
Among these there are both inorganic species
like phosphate, carbonate and sulphate and
organic compounds (humic substances). The
influence of complex formers may be due to
several mechanisms. They form complexes with
trace metals that may be both more and less
soluble than the uncomplexed ion. The complexes
may also be sorbed to a higher or lesser degree
than the uncomplexed ion. There may also be a
sorption of organic complexants on the mineral
surfaces that inhibit sorption of metal ions.
Complexation may occur with both macro and trace
components and competition has thus also to be
considered.

Complexation may occur with solid organic
substances. This may in some cases be virtually
irreversible.

The properties of the solid mineral phases
present are also of large importance to the
behaviour of trace metals. Metal ions may be
sorbed on the surfaces, causing both reversible
and irrebersible incorporation into the solid
phase. The interstitial water composition is
also influenced by the solid phases. The interac-
tions may however be slow, and equilibrium
conditions cannot always be expected.

Some general conclusions on the behaviour of
trace metals in peat may be drawn from

investigations of metal enrichments in nature.
Observations have been made on the difference in
the occurence of uranium and lead in peat
deposits. Also conclusions on the plant
availability have been drawn from this type of
investigations. The investigations show
enrichments of uranium in and lead in the bottom
layers of peat, while lead is enriched also on
the surface.

Radium seems to be stronger bound to the organic
solids than is uranium, and here the competition
with calcium is important.

Appendix 5. Human impact during the evolution of
a lake or a bay (A-B Wilkens)

The effects of human impact can as well
accelerate as counteract the development of a
lake or a bay, but most influences increase the
ageing rate during the different stages, for
instance:

- During the eutrophication: Fertilization
  of arable land in the catchment area
  may lead to nutrient losses with
  drainage and surface water, above all
  phosphorus and nitrogen, with increased
  eutrophication as consequence.

- Clearcutting of forests combined with
  fertilization, drainage and liming may
  give rapid release of nutrients to
  surrounding lakes.

- Ammonium containing fertilizers are
  under aerobic conditions enzymic
  oxidized to nitrate form leading to
  decrasing soil pH with increased
  solubility and mobility of cations,
  including heavy metals and radionuclides.

- Outlets of waste water from industries
  and households may accelerate the
  ageing rate of the recipient.

- Aquaculture may cause release of phosphorus and other pollutants in the lake water.

- Ditching of surrounding areas may lead to changes in the water balance in the recipient and in the water chemistry.

- In association with the terrestrialization: Drainage of a lake to get new agricultural soils leads to oxidation in the upper soil layers. Organic and inorganic sulphur may be oxidized to SO_4^{2-} and the soil will be acidified. The solubility and mobility of heavy metals and radionuclides will increase.

- More and more aerobic conditions in the upper layers of the soil leads to enzymic oxidation of nitrogen compounds to nitrate form and a lower pH as a consequence.

The release and transport of pollutants from sediment in organic soils may be influenced by for instance following factors:

- Oxidation processes and decomposition in organic soils due to drainage and cultivation.

- Uptake via roots of metal ions in connection to crop production and grazing.

- Release of pollutants in drainage water and transport to the recipient.

- Lowering of pH in soils due to acid rain may change the solubility and perhaps also the plant availability of metals.

- Use of peats and biomass in energy production may lead to dispersion of pollutants in the environment.

The effects mentioned above are examples of impacts that are common or may be expected today. Other human activities in the future may naturally lead to hitherto unknown effects.

PHASE 2

BACKGROUND

The first stage of this study aimed to describe
the wide range of parameters, some of them with
greater importance than others, governing the
pathways of radionuclides from groundwater to
man. The initial project could be summerized as
the identification of factors and parameters of
predominant importance for an assessment of
possible radiological consequences in a normally
revolving biosphere.

Furthermore, the study included recommendations
for a reference field test area and for laboratory
studies of the chemical properties of radionuclides
in sediments.

PROJECT DESCRIPTION

In order to reach an more detailed understanding
of the consequences of the potential release of
radionuclides into the biosphere from an
underground repository, the phase 2 of the study
is suggested to focus on the quantification
evolution of lakes and coastal waters and their
sediment. The main purpose of the Phase 2
project is therefore to explain the transfer
mechanism and to identify possible accumulation
of radionuclides from groundwater to man during
different periods of the development of a
recipient area and quantitatively determine the
changes.

The proposed area of investigation is the
precipitation area of two outflows called Sibbo-
and Trobbofjärden, approximately 20 kilometers
east of Nyköping. The lake system in this area

represents typical lakes in the southern and middle region of the Swedish coastal and rural area.

The Trobbofjärden is a lake which was cut off from the sea some 30 years ago by human impact. By land uplift, the Sibbofjärden is today a fresh-brackish bay linked to the sea by a narrow passage. The transition of two separate systems from sea to lake can therefore be studied in a unique way related to time.

The proposed project is divided into three major categories:

1) <u>field studies,</u> including

   - sampling of sediments along profiles from surrounding solid ground to deep lake sediments in order to cover lake sediments of different age and character,

   - discharge measurements and water sampling of macro constituents and suspended material for estimation of material and water balance of the lake,

   - determination of the annual growth of sediment and its yearly cycle by using sedimentation traps at different sites and depths,

   - mapping of vegetation for the determination of biomass production.

2) <u>chemical studies,</u> comprising

   - determination of the pore water composition in the sediments. Parameters of importance to the mobility of trace metals such as pH, Eh, pS^{2x}, ionic strength and contents of complex formers will be determined,

   - characterization of the solid sediment phase. Measurement of organic content, cation exchange

capacity and in some cases surface
area and mineralogy,

- trace metal distribution between pore
water and solid to provide information
on the mobility of naturally occuring
trace metals in sediments. The data
may be used in predicting the
transport of radionuclides in sediments,

- the influence of organic complex
formers on the mobility of trace
metals. Fractionation according to
molecular weight of trace metal
complexes in the pore water gives
information on the influence of
organic complex formers on the
mobility.

3)   transport study of radionuclides in sediments

- in situ studies to verify the transport
of actinides and fission products in
sediment. Sediment cores are collected
in the recipient. The nuclide is
implanted at different depths in the
core. Thereafter these are transferred
back to the recipient. On subsequent
occasions the cores are recollected
and the distribution of the nuclide
within the core is recorded. The
migration is correlated to the physical
and chemical sediment properties.

TIME SCHEDULE

The Phase 2 project is planned to start up in
early 1985 and continue through to mid-86
according to the following preliminary schedule:

	1985	1986	1987
Field studies:			
Water balance	------------		
Material bal	------------		
Mapping of vegetation	---	---	
Sedimentation sampling	_____	_____	

Chemical studies:
Pore water
composition,
solid phase,
trace metal
distribution,
organic
formers   _____

Sedimentation studies:
Core sampling __   __

Progress of work will be reported regularly and discussions held in order to evaluate on-going results. A report of Phase 2 is planned for June 1986.

In a Phase 3 the material will be used for the validation and/or development of biosphere models.

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WILKENS A-B
Human impact during the evolution of a lake or a
bay
STUDSVIK/NW-84/769

Björn Sundblad

LAKE AND COASTAL ZONE EVOLUTION
DURING POSTGLACIAL TIME

ABSTRACT

The most important climatological and geomorpho-
logical processes governing the evolution of
lakes, from a physical point of view, are
discussed. The classification of the Swedish
lakes based on the area of the lakes and the
geomorphological regions is presented.

The coastal zone evolution, including land
uplift and examples from some areas along the
Swedish coast, is also presented. In one of
these areas, Sibbofjärden close to Studsvik, a
field survey has been carried out. The salinity,
pH, temperature and depths have been measured.

Rough estimates of the time scale for lake
evolution from present lake to dry land have
been done.

Estimates of the present residence time for
different hydrological regions and lake types
are also included in the study.

TABLE OF CONTENTS

1   INTRODUCTION

The landscape is continuously changing. Large areas of the present landscape have formerly been at the bottoms of lakes and seas.

Because of the land uplift new land areas will be formed. Coastal bays can become lakes in the future.

A lake will be changed in many ways during its lifetime. There are many processes governing these changes, including geomorphological, climatological, biological and hydrological processes and human impact.

The above mentioned processes contribute to the terrestrialization of a lake, which in most cases is a natural process. The lake normally ages slowly by silting up, whereby the vegetation migrates out into the lake. This process is slow in the early stages, but it usually accelerates during the evolution.

Under natural conditions the ageing process takes thousands of years. However, nutrients from households, industries and farming, are transported to the lakes and the process is accelerated by the human impact. The nutrients can also be brought to the lake by cultivation, draining and clear-cutting within the catchment area.

Processes at the bottom and within the uppermost layer of the sediments are essential for the terrestrialization of a lake. Furthermore, the sediments are often the greatest reservoir of radionuclides which have been brought to the water recipient. Thus in the future the

sediments formed earlier can be brought into
cultivation and the radionuclides can begin to
circulate in the biosphere again.

2   PROCESSES GOVERNING LAKE EVOLUTION

2.1   Climatological Processes

From a climatological point of view it is mainly
the air temperature and the precipitation that
affect the level of a lake.

A humid cold climate means a high discharge into
a lake and little evaporation. As a result of
these conditions the lake level will be high.
The opposite, a dry warm climate, leads to
decreasing lake levels because of high evapo-
transpiration and a low discharge.

There are examples of both these extremes of
climatic type during the post-glacial period.
About 9000 years BP, in the Ancylus era of the
Baltic, the climate was warm and dry. During
this dry period some lakes dried out completely.
Others lost their outlet and because of the high
evaporation they were enriched in salts and
became brackish.

After the Ancylus era the Littorina Sea period
began. The temperature was still high, about
$2-3^{\circ}C$ above the summer temperatures nowadays.
However the climate became more humid and the
discharge increased, so that the lake levels
rose again.

These two periods are examples of variations
over a long time span. However, the duration of
the fluctuation can be much shorter, for example
over decades.

2.2   Geomorphological Processes

Geomorphology is the science of forms of the
earth, their origin and development.

A lake is usually formed in a fast way by
tectonic movements in the earth crust, or as a
result of the effect of inland ice on the
landscape. On the other hand the ageing is a
slow process. However, human impact can accele-
rate the conversion of open water to dry land.

The so called exogenous forces erode the earth's
surface. The main exogenous processes are
(Lundegårdh, 1964):

-        mechanical and chemical weathering
-        fluvial activity by running water
-        glacial activity by ice
-        eolian activity by the wind
-        marine activity by the sea
-        biogenous activity by man and the
         "organic world"

All these processes act in the same direction,
i.e. to level the landscape. The water is the
transport system and this means that large
quantities of the eroded material will be
transported to the lakes and the sea.

3   PRESENT STATUS OF LAKES IN SWEDEN -
         A GEOMORPHOLOGICAL REVIEW

3.1   Type of lake

A geomorphological lake register for Sweden has
been set up (Håkansson et al 1983). About 1000
lakes are included in the register.

A lake can be classified in many ways, for
example according to some of its morphometric
data such as volume, area, depth etc. In a
report (Håkansson 1978) about Lake Vänern many
morphometric parameters are defined and de-
scribed.

In the geomorphological lake register three
morphometric base parameters have been used.
They are lake area (A), lake volume (V) and
maximum depth (D_{max}). From these parameters it
is possible to deduce other expressions, which
describe the character of a lake.

Håkansson has used the following morphometric
parameters:

-        mean depth, $D_{med} = \dfrac{V}{A}$ (3-1)

-        volume development, $VD = \dfrac{3 \times D_{med}}{D_{max}}$ (3-2)

-        the potential area of erosion and tran-
         sport bottoms, A_{-ET} (% of the total
         lake area).

The form of the lake (the relative hypsographic
curve) is also defined by Håkansson. In Figure 1
five different lake types are represented.

Table 1 gives among other things the relation between the different classes and the volume development.

A very convex lake has shallow shores and only a small part of it is deep. The opposite lake type, concave, has steep shores and a large area within the same depth interval.

These different shore conditions are of great importance for sedimentation-resuspension, i.e. they may determine whether the lake has an erosion-transportation bottom or an accumulation bottom. In addition the depth conditions are very important for the biomass development.

Table 1. Terminology and classes for the different lake types (from Håkansson 1981)

Name		Probability %	Classes	Volume development
Very convex	VCx	6.545	f(-3)-f(-1.5)	0.05-0.33
Convex	Cx	24.170	f(-1.5)-f(-0.5)	0.33-0.67
Slightly convex	SCx	38.300	f(-0.5)-f(0.5)	0.67-1.0
Linear	L	24.170	f(0.5)-f(1.5)	1.00-1.33
Concave	C	6.545	f(1.5)-f(3)	1.33-2.0

The area of erosion and transport bottoms (A_{-ET}) has been calculated from the following formula

$$A_{-ET} = 100 - A_{-ACK} = 25 \times \sqrt{A/D_{med}}$$
$$\times 41^{0.061 \times D_{med}/\sqrt{A}} \qquad (3-3)$$

where A_{ACK} is the ackumulation area (%)

If $\sqrt{A/D_{med}}$ = DR (dynamic ratio) eq (3-3) will become

$$A_{-ET} = 25 \times DR \times 41^{0.061/DR} \qquad (3-4)$$

3.2   Geomorphological regions

The Nordic countries have been divided into 18
geomorphological regions (Rudberg 1968). Twelve
of the eighteen regions are of interest in
Sweden. From a cluster analysis (Håkansson 1983)
it is clear that several of the regions show
morphometric similarities. It is possible to
obtain five lake classes, see Table 2.

Table 2. Lake classes

Lake class	Type of lake
1	Lakes in flat terrain
2	Lakes in hilly terrain
3	Lakes in very hilly terrain
4	Lakes in mountainous terrain
5	Lakes in fjeld terrain.

The different regions are shown in Figure 2
(Håkansson 1983).

A table is also included in the Figure 2,
showing the most probable volumes, mean depths
and erosion-transportation bottoms for lakes
having an area of 3 km^2. It is evident that the
mean depth differs a lot between the five
regions.

The lake classes have also been divided into
three different area classes:
-        A0 $A < 1\ km^2$
-        A1 $1 < A < 10\ km^2$
-        A2 $A > 10\ km^2$

The distribution among the area classes (A0-A2)
is roughly 60, 30 and 10 percent. The lake
classes are dominated by class 3. About 70

percent of the 1060 lakes are found in this class.

To exemplify the distribution of lake parameters the class A0/3 has been chosen; statistics for this class are presented in Table 3.

Table 3 Geomorphological parameters of area class A0,-lake class 3

	Area km^2	Volume km^3	Max depth m	Mean depth m	Volume development
Mean	0.237	0.00119	10.6	4.2	1.33
Standard deviation	0.249	0.00197	7.3	2.5	0.43

From Håkansson 1983.

Correlationcoefficients have been calculated for the different classes. These coefficients for class A0/3 are found in the table below.

Table 4 Correlation coefficients for class A0/3

Mean depth - Max depth	0.83
Area - Volume	0.77
Mean depth - Volume	0.72
Max depth - Volume	0.64
Max depth - Volume development	-0.49
Area - Max depth	0.34
Area - Mean depth	0.32

From Håkansson 1983.

These values are representative for most of the classes.

4   COASTAL ZONE EVOLUTION

4.1   Sea level variations

An inland ice will by its weight press down the
earth's crust. When the ice melts the crust
begins to return to the original level before
the ice age. This movement of the crust is
called isostatic land uplift.

The inland ice contains great quantities of
water in solid form. It has been calculated that
during the last ice age about 35 to 40 km^3 more
water than nowadays was retained (Lundegårdh
1970). This is equivalent to a lowering of the
sea level of 90 to 100 m. When the ice melted
the sea level rose again. As long as melting
inland ice exists, this so-called eustatic
effect will diminish the regression of the sea
level. There have been periods in southern
Sweden when this effect has dominated and the
shores have been submerged. This is called
transgression of the sea level, i.e. the opposite
of regression.

Because of these two opposite effects it is more
appropriate to talk about shore displacement
instead of land uplift. The shore displacement
has been studied in many places along the
Swedish coast. The displacement has been highest
in Angermanland, around 285 meters. The shore
displacement was very rapid in the beginning,
around 10 cm/year (Miller 1982). However, the
displacement decreased to about 2.5 cm/year
during the Litorina stage of the Baltic, i.e.
around 4000-7000 years B.P. During the last 3000
years the shore displacement has been around
1 cm/year. This present value decreases to

around 0.5 cm/year in the Stockholm area and is
around zero in the southern part of Sweden.

It has been possible to use varved lake sediments
to study the development of a lake from the time
when it was isolated from the sea (Segerström,
Renberg 1982). In the Umeå area the time of
isolation has been obtained from sediment
studies. For example Sarsjön, about 40 km
north-west of Umeå, was isolated from the sea
about 9000 years ago. Its height above sea level
is nowadays 177 meters. Thus the mean shore
displacement is around 2 cm/year.

There are certain conditions that must prevail
to get the varved lake sediments. The changing
seasons provide the necessary conditions to
obtain varves with different thickness and
colour. No stirring due to bioturbation or wave
motion in the sediments may occur.

These still conditions at the bottom can be
obtained in productive lakes with depths exceeding
6-7 meters (Segerström, Renberg 1982).

4.2   The Sibbofjärden and Trobbofjärden area

In the vicinity of Studsvik there are two
enclosed seas, Sibbofjärden and Trobbofjärden.
They are typical examples of the development of
archipelago areas in Sweden.

4.2.1   Description of the Sibbofjärden
Sibbofjärden is nowadays an enclosed sea, but
over the next few hundred years it will be
isolated from the sea and become a lake with
fresh water. Its threshold at the outlet is
around 1-1.5 m below the mean sea level.

The "lake" area is 5.6 km^2 and the drainage area
is 83 km^2. The area of arable land is 25 percent
of the total area. This part of the drainage
area is the former sea bottom of the bay. The
calculated "lake" volume is 2.6 x 10^7 m^3 and the
mean depth is almost 5 m. In the table below
some morphometric data are presented.

These data are calculated according to Håkansson
(1978).

Table 5   Morphometric data of Sibbofjärden

Parameter

Area	5.6 km^2
Volume	0.026 km^3
Mean depth	4.7 m
Maximum depth	11 m
Relative depth	0.4 m
Maximum length	6.0 km
Mean width	0.9 km
Area of erosion and transportion bottom	20.8 %
Dynamic ratio	0.5
Volume development	1.3

4.2.2   Residence time for Sibbofjärden
The total volume of Sibbofjärden is 0.027 km^3. A
specific discharge of 5.5 l/s km^2 (Tryselius
1971) and a catchment area of 83 km^2 give a
theoretical residence time of 1.7 years.

However, the bay is connected with the sea, i.e.
it has a water exchange through the outlet. A
rough estimate of this exchange gives a value of
3 x 10^7 m^3/y. This is equivalent to a residence
time of around 285 days. The two sources of
water exchange for the bay give together a total
residence time of around 200 days.

To obtain the exchange through the outlet the water fluctuations and their duration and amplitude (reduced) in Tvären has been used.

To get a better understanding of the water exchange, especially through the outlet it is necessary to measure the flux by current meter during some representative periods.

4.2.3   Field survey in March 1984

In March a survey in the areas of Sibbofjärden and Trobbofjärden was performed, see Figure 3. The following parameters were measured at seven places in Sibbofjärden and at one place each in Trobbofjärden and Rundbosjön respectively;

- depth
- conductivity (salinity)
- pH
- temperature
- ice thickness

In addition, sediment cores were taken at the same places.

The depth in Sibbofjärden varied between 2 and 5.5 m. In order to obtain a more exact measure of the volume it would be necessary to take soundings over the whole bay.

The variation of the conductivity with depth and distance from the outlet is shown in Figure 4. In the surface water (0-2.5 m) below the ice a halocline exists. This means that the water below the halocline is brackish with increasing conductance towards the bottom.

In the fresh surface water a slight decrease with increasing distance from the outlet can be

seen. The opposite situation is valid for the
bottom layers. The bottom value is around
650 mS/m. This can be compared to the value of
725 mS/m outside Sibbofjärden.

The pH varies between 5.60-7.25. The highest
values are obtained at the bottom. No special
gradient is found. In the vicinity of the inlet
at Björksund the pH is somewhat higher in the
surface layer. This depends on the discharge
from arable land areas, which usually have a
higher pH.

The temperature varies between 0.4 to 2.2°C in
Sibbofjärden from the surface to the bottom.

The temperature in the northern part, Sundhälla-
fjärden, deviates from that of Sibbofjärden
proper, with a higher temperature at the bottom.
This is around 4°C.

The ice thickness was around 40 cm on the whole
of Sibbofjärden.

One measuring point in the southern part of
Trobbofjärden was included in the survey to
compare the two enclosed seas. In 1957 a dam was
built at the outlet of Trobbofjärden to regulate
the inflow from the Baltic. The whole of Trobbo-
fjärden was converted to a freshwater reservoir
over a few years.

An additional site was chosen in Rundbosjön, a
lake within the catchment area of Trobbofjärden.
It was isolated from the sea around the Viking
Age, i.e. about 1000 years ago.

The specific conductance in Trobbofjärden varied
between 21 and 32 mS/m, from the surface to the

bottom at 2 m. In Rundbosjön the corresponding values were 16 to 24 mS/m at 8 m depth.

The pH values were around 6 for both lakes.

The temperature at the bottom (2 m) of Trobbo-fjärden was $2.6^{\circ}C$ and in Rundbosjön at 8 m depth it was $5.0^{\circ}C$.

4.2.4   Field survey in June 1984

A survey in the beginning of June 1984 showed that Sibbofjärden had a marked temperature stratification. The thermocline was very strong between 5 to 6 meters, about $8^{\circ}C$ per meter. A vertical section from the deepest part of the "lake" is shown in Figure 5.

The halocline was found between 5 and 7 meters, see Figure 5. In the epilimnion (surface water) the conductivity was just below 500 mS/m. These values are comparable with values found at the 2 meter level during the winter survey. Below 7 meters the conductivity was around 700 mS/m, comparable with the conductivity found outside Sibbofjärden at Helgö.

From the two surveys it is evident that the salinity (conductivity) in the bottom layer is the same as in the sea outside Sibbofjärden. Further, there is no marked horizontal stratifi-cation of the salinity. The salinity is about the same at the outlet as in the northernmost part of the enclosed sea. This is valid for the epiliminion as well as for the hypolimnion (the bottom water below the thermocline).

Depth measurements gave a maximum depth of around 11 meters.

4.3   Other coastal areas of interest

Two areas will be mentioned here, Utålskedjan in
Uppland and a coastal area close to Umeå. There
are of course other sites that can be of interest.
However, these two are examples of the develop-
ment of coastal areas from brackish sea to
freshwater lakes.

Utålskedjan is a chain of lakes with increasing
salinity towards the Baltic (Ingmar and Willén,
1980). A comprehensive material covering topo-
graphy, sediment, vegetation, hydrology, water
chemistry etc was collected during two periods,
1953-61 and 1967-75.

The innermost part of the catchment area was
isolated during the Middleages. Some lakes are
freshwater systems for the greater part of the
year. The thresholds are below the high-water
level, which means that brackish water can enter
the lakes.

The different sediment layers originating from
the brackish and freshwater periods respectively
have been studied. The marine clay gyttja with a
low content of organic material (10-15 %) can be
distinguished from clay gyttja with a higher
organic content. The depths of the sediments are
roughly between 5-10 m. The sedimentation
velocity in the deepest parts is 3-7 mm/year.

In the Umeå area the varved sediments have been
studied, as mentioned earlier (Segerström,
Renberg 1982). The consequences of the land up-
lift on the vegetation development in the
coastal zone in the vicinity of Umeå have also
been studied (Ericsson 1982).

5   THE TIME SCALE OF LAKE EVOLUTION

The sedimentation rate is not constant. It
varies according to the climate, bioproduction
etc as has been mentioned above. An example of
this is found in Figure 6, showing the sediment
growth from Lake Trummen (Lettevall 1973).

The sedimentation varies within a lake, with the
highest rates in the deepest part of the lake.
This is illustrated in Figure 7.

An example from Poland (Pawlikowski et al 1982)
shows lake development during the last 12000
years, see Figure 8. In this figure a cross
section of the sediments of an overgrown lake at
Woryty is shown. There are different layers with
peat and gyttja from different stages of the
lake development. Nine different stages have
been identified. The sedimentation velocity has
varied between 0.3 and 1.6 mm/y.

The sedimentation velocity in Swedish lakes
varies within the range 0.5-8 mm/y (Håkansson
1981). A value around 2-3 mm/y can be reasonable
to use for calculations. The compaction is
included in this value.

A theoretical calculation of the silting up for
different lake types has been done. Three
different types, very convex, slightly convex
and concave, have been used.

A constant sedimentation over the whole lake
area is assumed. The calculations are presented
in Figure 9. It can for example be seen that
when the depth has been reduced to half of the
initial value, the remaining lake volume is 35,
15, and 7 percent of the original for the

concave, slightly convex and the very convex
lake types respectively.

It is also interesting to study how the lake
areas change with decreasing depth for the
different lake types. These variations are shown
in Figure 10. The differences between the lake
types are great. For example, for a depth
reduction of 25 percent the remaining lake area
is 95, 60 and 15 percent for concave, slightly
convex and very convex type lakes respectively.
These differences are of importance for biomass
development in the litoral zone.

To be able to calculate the concentration of for
example, an element entering a lake, the re-
sidence time in the lake must be known. To illu-
strate how the silting up affects the residence
time for different lake types, a lake of the
size of Sibbofjärden has been chosen. The dis-
charge is that of the catchment area for Sibbo-
fjärden $(5.5 \ 1/s \cdot km^2)$. The sedimentation is
2 mm/y, and the maximum depth is 10 m. This
means that it will take 5000 years to fill up
the lake, if conditions are constant over the
whole period. The results of the calculations
are shown in Figure 11.

The residence time varies according to the
runoff, catchment area and lake volume. An
attempt has been made to find the limits for the
residence time.

The average annual runoff varies between 4 and
50 $1/s \cdot km^2$. The low values are found in the
south-eastern part of Sweden and the high values
are obtained in the high mountain areas (Tryselius
1971). Sweden can be divided into nine hydro-
logical regions (Melin 1980), see Figure 12. The

monthly river runoff from these different
regions has been studied (Gottschalk 1975). The
average annual runoff within these regions
varies between 8 and 20 $1/s \cdot km^2$. However, the
monthly runoff can vary as much as from 2 to 90
$1/s \cdot km^2$ within the same region.

In order to calculate the discharge from runoff
values, the catchment area must be determined.
As the lake areas are known from the geomorpho-
logical register, one way is to calculate the
catchment area from the lake percentage and lake
area. The lake percentage for the different
regions has been calculated from areas exceeding
200 sq km (SMHI 1970). In Table 6 the values
used are presented.

For the calculation of the residence time the
mean lake percentage has been used for each
region. In addition, a main lake type has been
chosen for each region. The residence time is
calculated for the three area classes for each
lake type. The results are given in Table 7.

Table 6   Runoff and lake percentage area

Region	Runoff ($1/s \cdot km^2$)			Lake percentage (%)		
	Annual Mean	Monthly Maximum	Minimum	Mean	Maximum	Minimum
A	17	60	3	4	13	1
B	20	90	2	8	16	2
C	17	73	3	8	16	2
D	17	72	2	6	12	1
E	11	36	2	5	8	3
F_a	8	15	2	9	15	1
F_b	14	23	5	6	14	0

Table 7 Residence time for different hydrological regions
         and lake types

Region	Lake type	Residence time (years)		
		Area<1 sq km	1 sq km<Area<10 sq km	Area >10 sq km
A	1	0.2	0.3	1.9
B	4	0.6	1.4	3.0
C	3	0.7	1.2	3.2
D	3	0.6	1.0	2.9
E	2	0.4	1.0	0.6*
F$_a$	3	1.8	3.1	8.4
F$_b$	3	0.7	1.2	3.3

* only 4 lakes are included in this class

Another method of calculation, using data from the Swedish geomorphological lake register, will be presented below. Three different classes, A0-3, A1-3 and A2-3, representing about 70 percent of the lakes within the register, have been chosen. The mean depths for the different classes are used to calculate the theoretical lifetime of the lakes. To simulate an increased sedimentation because of increasing organic production at the end of the life of the lake, different sedimentation rates are used for different depth intervals. The sedimentation rates are 3 mm/y for 0-2 meters, 2 mm/y for 2-5 meters and 1 mm/y for depths over 5 meters.

The results from the calculations of the time before the lakes are filled up with sediments are presented in Table 8. The 95 % confidence levels given are only calculated from lake data. No attempt has been made at this stage to evaluate the uncertainty of the sedimentation rates.

Table 8   Theoretical lifetimes for different
          lake classes

Class	Mean depth m	Standard deviation, m	Lifetime years
A0-3	4.2	2.5	1800+100
A1-3	7.7	4.2	6000+500
A2-3	14.6	10.2	13000+3000

These calculations will give the magnitude of
the remaining lifetime for the present lakes.
However, the calculations can be modified to
allow for different sedimentation rates at
different depths, the vegetation in the litoral
zone, bioproduction within the water mass etc.
These different parameters and their variations
must be included in future studies.

6   CONCLUSIONS

Most of the recent lakes in Sweden were originally
formed by the latest inland ice. However, the
lakes are occasional phenomena. Many processes
affect the evolution of a lake. Some of these
are discussed in this report.

Lake evolution goes through several stages,
where the relative importance of the different
processes will change.

The water is the carrier system for suspended
material, disolved substances, biomass etc.
Because of this it is important to know the
residence time during different stages of lake
evolution.

The sedimentation is also dependent on the water
transport to and within the lake. The little
known relation between sedimentation rate and
residence time must be determined, so as to be
able to find the transfer coefficients between
water and sediment, and their variations. The
water transport also affects the horizontal and
vertical distribution of the sedimentation.

Another important parameter is the stratification
of the lake, mainly the thermal stratification.
A reduced vertical water exchange will for example
affect the oxygen condition at the bottom.

In the coastal areas the transition from brackish
water to fresh water produces differences from
the pure lake systems as regards stratification,
sedimentation etc. Because of this the transition
period must be treated separately when dealing
with lake evolution. The Sibbofjärden-Trobbofjärden
areas are suitable for studying these differences.

In order to realize a program for lake, bog or
bay evolution for the Sibbo-Trobbofjärden area,
a field survey must be carried otu over some
years. This is required for two principal
reasons, firstly to acquire a knowledge of the
systems and secondly to collect soil, water,
vegetation and sediment for laboratory studies.

Such a field survey will include water and
material balances, sampling of soil and sediment,
measuring of sedimentation rate and the estima-
tion of pH and Eh in the field.

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Runoff map of Sweden. Average annual runoff for
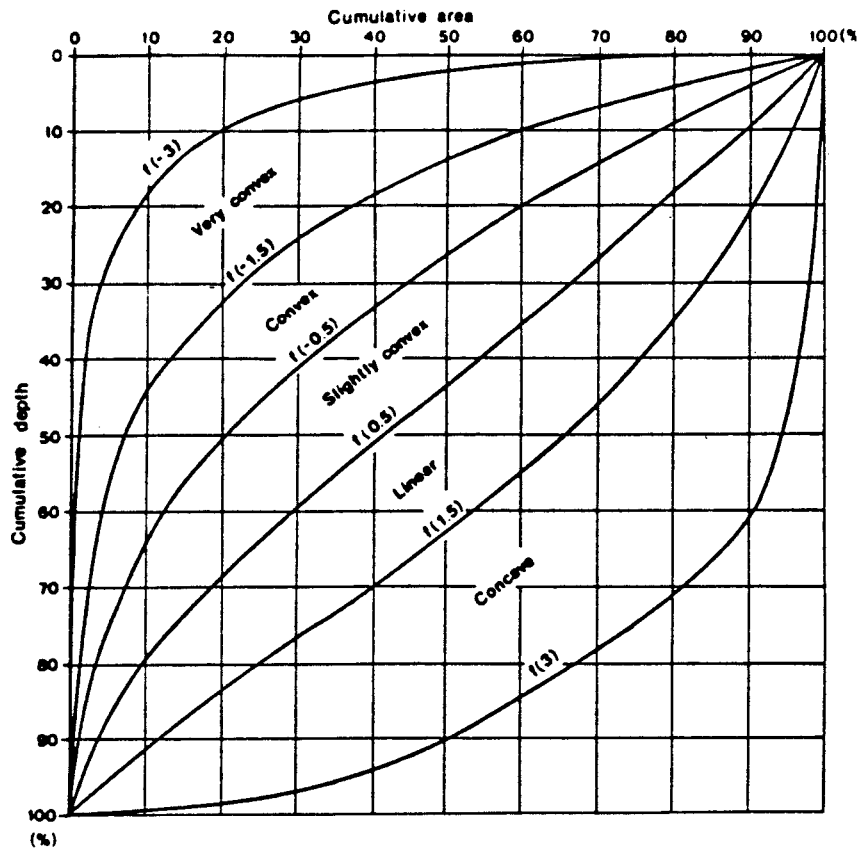the period 1931-60. SMHI, Med ser C Nr 7.

Figure 1. Relative hypsographic curves for
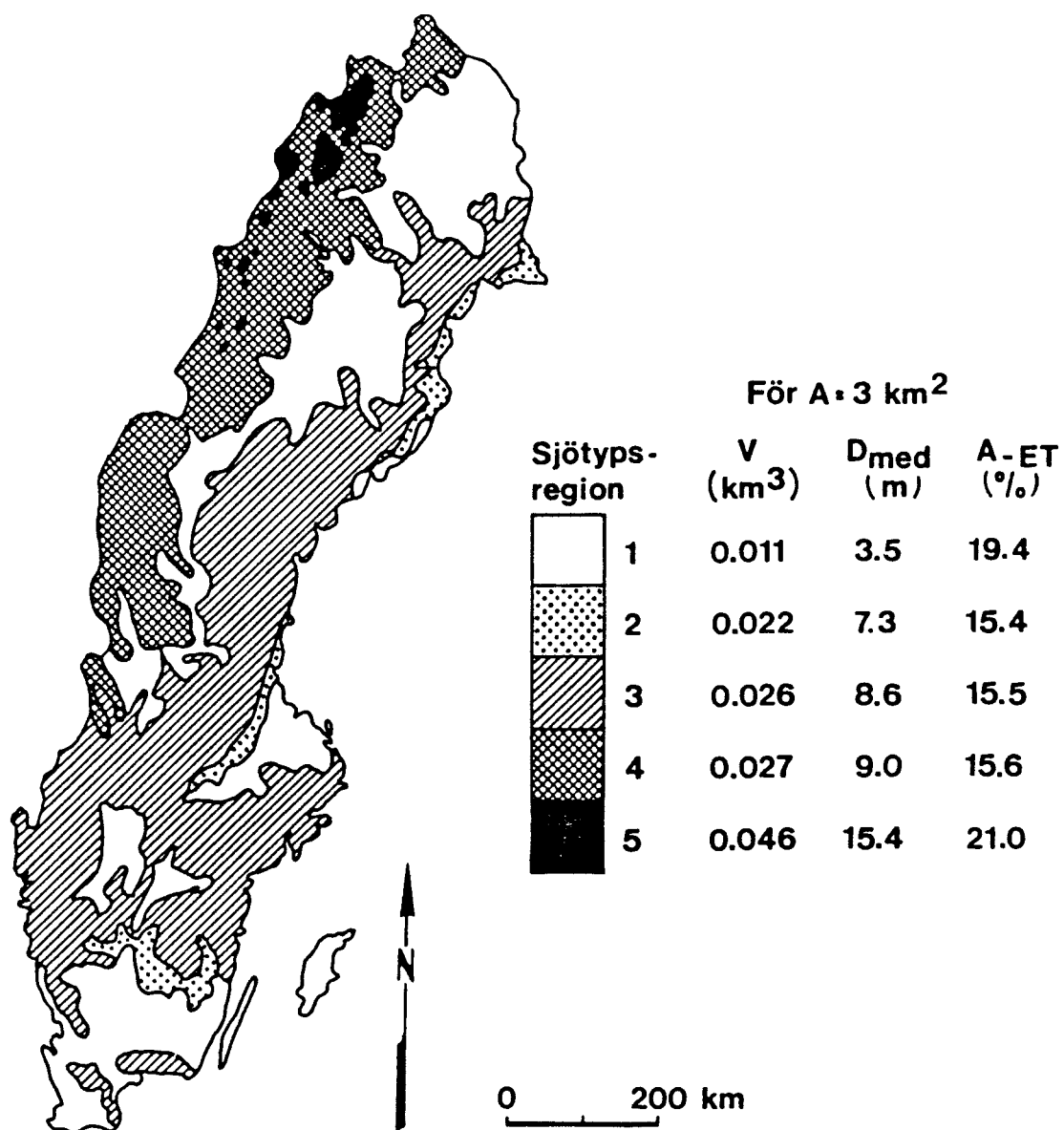the statistical lake forms.
From Håkansson (1983).

För A = 3 km^2

Sjötyps-region	V (km^3)	D$_{med}$ (m)	A-ET (%$_0$)
1	0.011	3.5	19.4
2	0.022	7.3	15.4
3	0.026	8.6	15.5
4	0.027	9.0	15.6
5	0.046	15.4	21.0

0 _____ 200 km
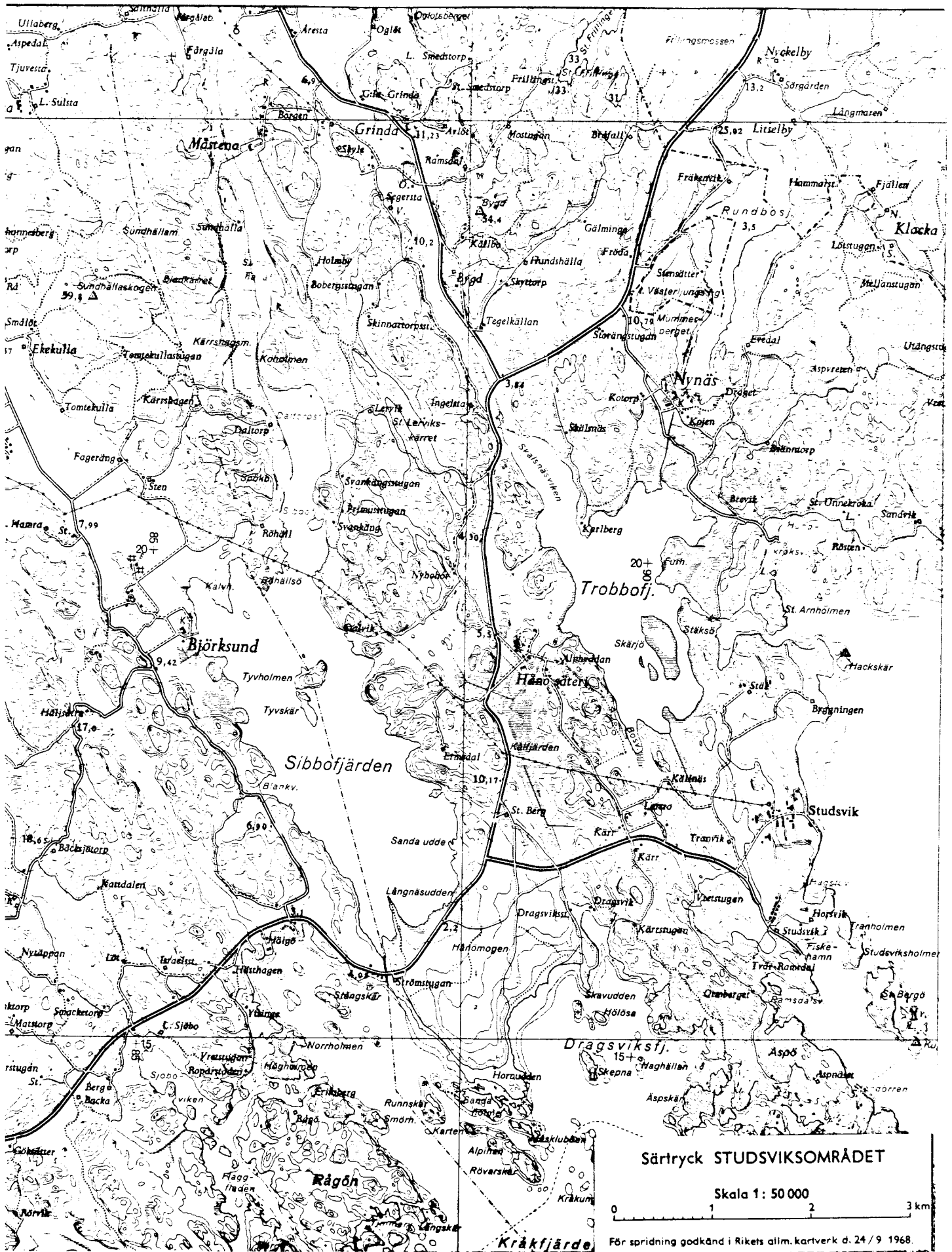
Figure 2. Geomorphological lake regions.
From Håkansson (1983)

Särtryck STUDSVIKSOMRÅDET

Skala 1 : 50 000

0   1   2   3 km

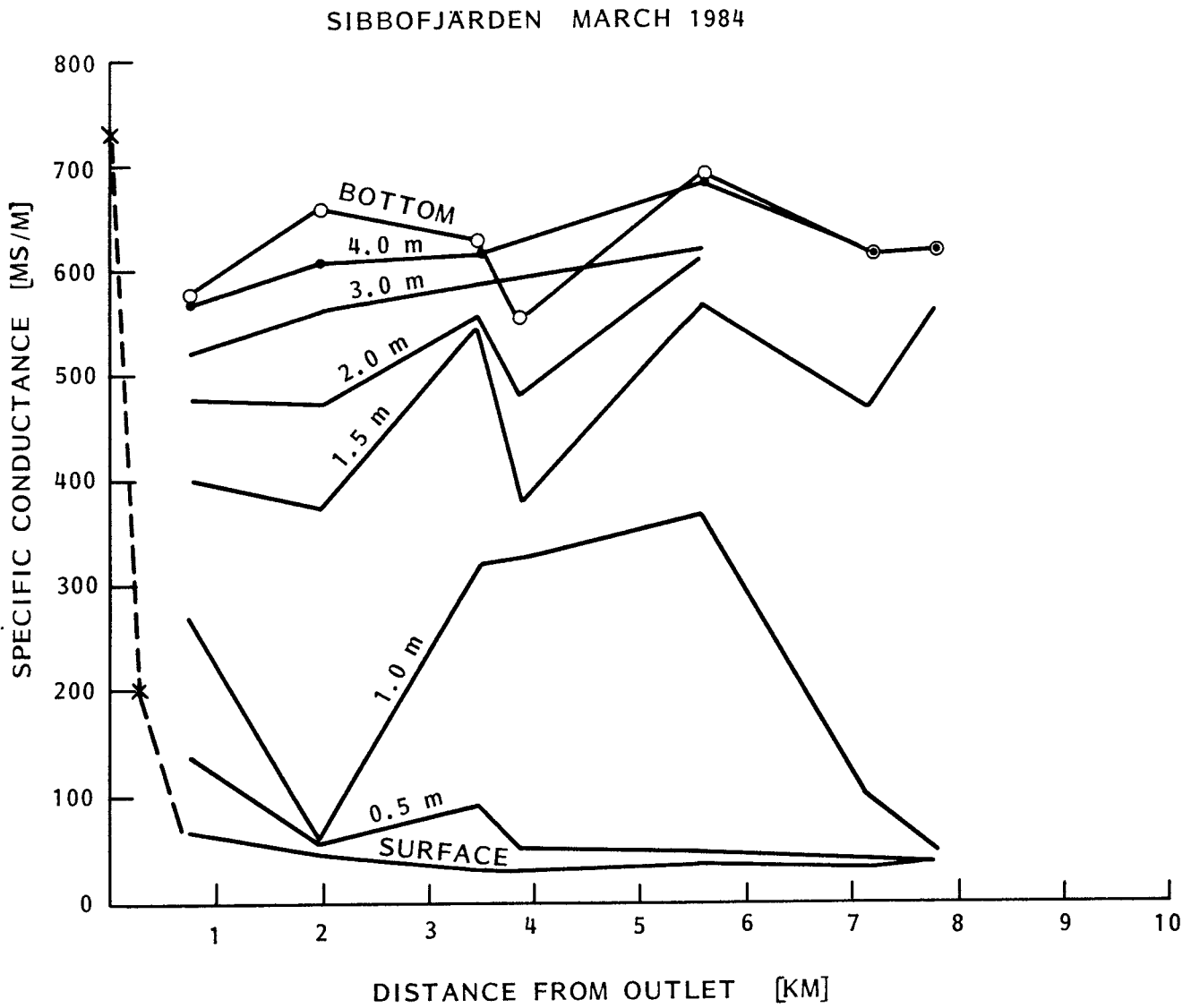För spridning godkänd i Rikets allm. kartverk d. 24 / 9 1968.

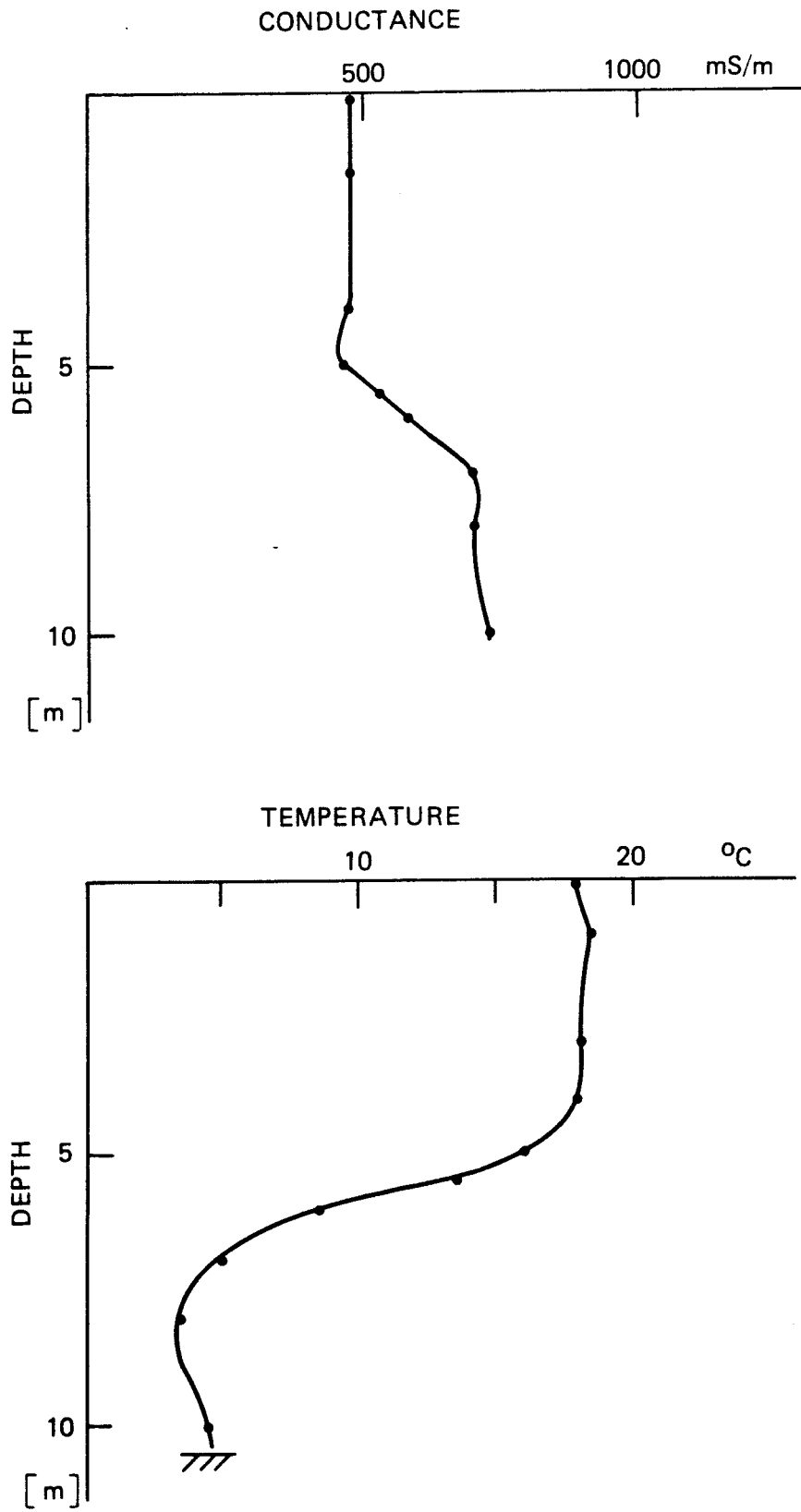Figure 4. Conductivity in Sibbofjärden
in March 1984.

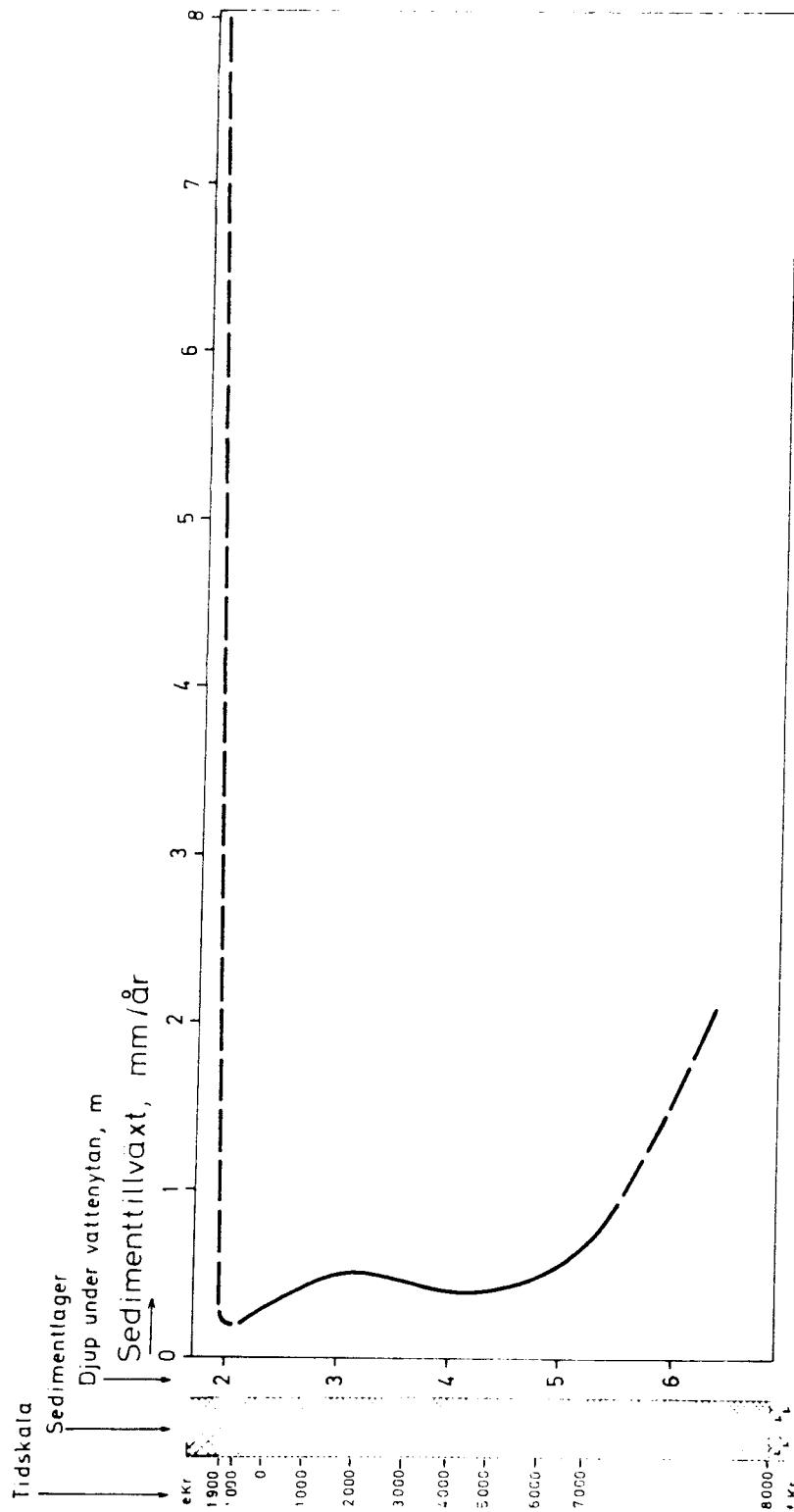Figure 5. Conductivity and temperature
in Sibbofjärden in June 1984.

Figure 6. Sedimentation rate in lake Trummen.
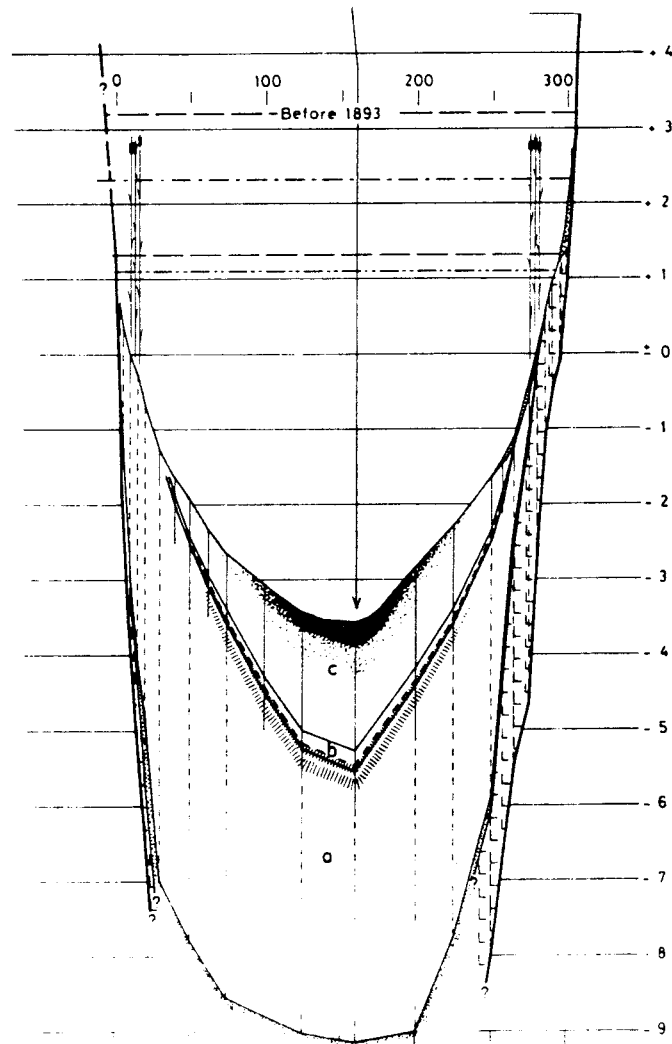From Lettevall (1973).

Figure 7. Sediments in the Utål chain basins.
Depths in meters. a = normal "marine"
clay-gyttja, b = clay-gyttja or gyttja
organic content 20-35 %, c = as "b"
with less saturated colours.
From Ingmar, Willén (1980).

Figure 8. Sediment structure. From Pawlikowski (1982).
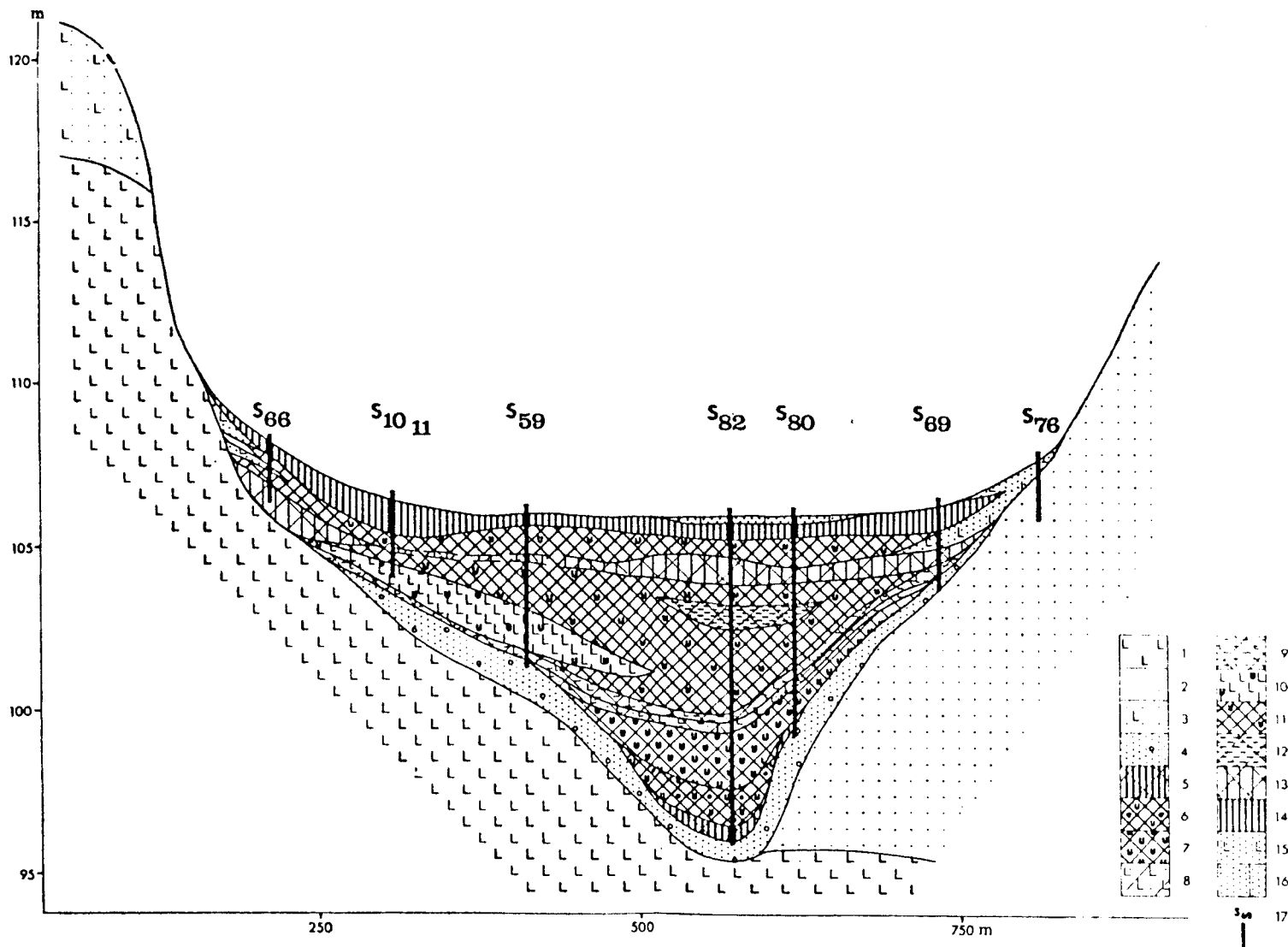
Section across the Woryty site along the A—B transect shown in Fig. 3 (symbols follow Troels-Smith, 1955): 1 — glacial till, 2 — fluvio-glacial sands, 3 — kame silts and fine sands, 4 — coarse sands with gravel, 5 — basal peat, 6 — ferruginous gyttja, 7 — calcareous gyttja, 8 — sandy silt with gyttja, 9 — dy-like sediment, 10 — calcareous silt, 11 — fine detritus gyttja with variable carbonate content, 12 — sapropel-like sediment, 13 — swamp peat with gyttja, 14 — surface peat, 15 — slope land-slide material deposited in the marginal zone of lake, 16 — surface sands, 17 — position of corings; S-82 and S-80 sampled for paleoecological investigations
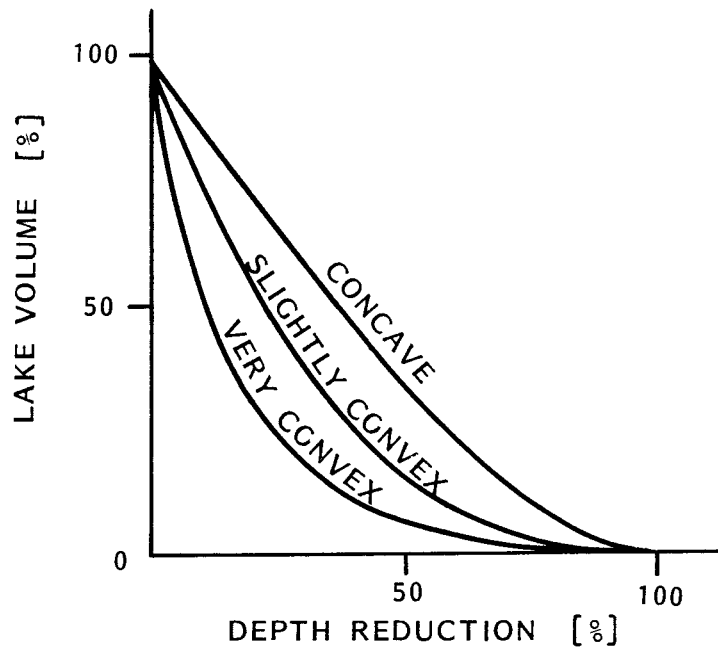
Figure 9. Depth-volume relations for
           different lake types.
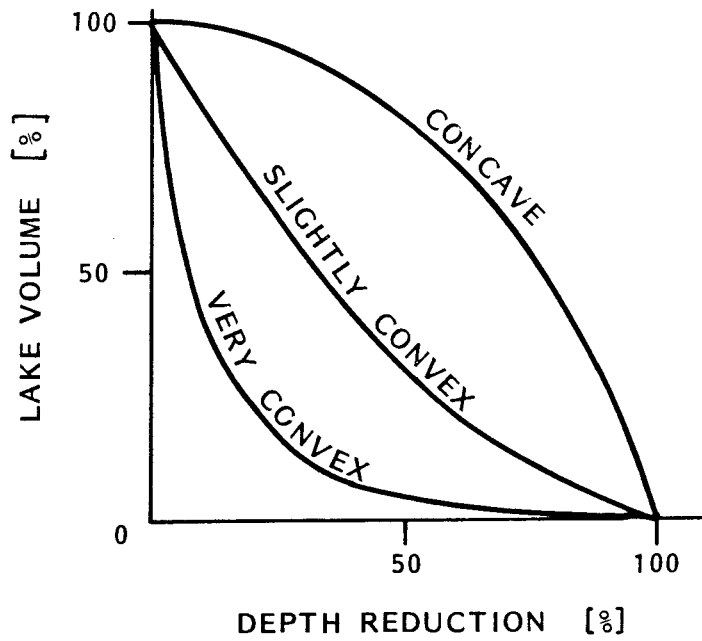
Figure 10. Depth-area relations for
            different lake types.
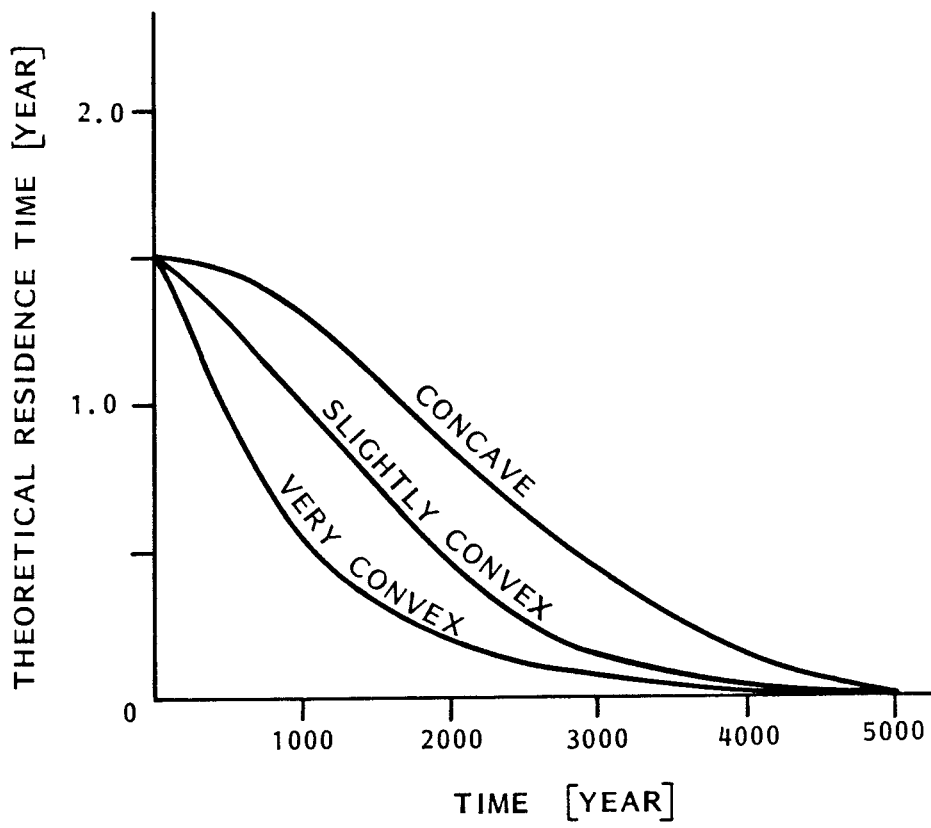
Figure 11. Theoretical residence time for different lake types.

A. Polar Rivers
B. Northern Mountain Rivers
C. Central Mountain Rivers
D. Mountain and Forest Rivers in the south of
   Norrland and north of Svealand
E. Forest and Coastal Rivers in Norrland
F. The Rivers of Southern Sweden
   a. eastern rivers
   b. western rivers
G. Rivers from the large Swedish lakes
H. Rivers with mixed regime
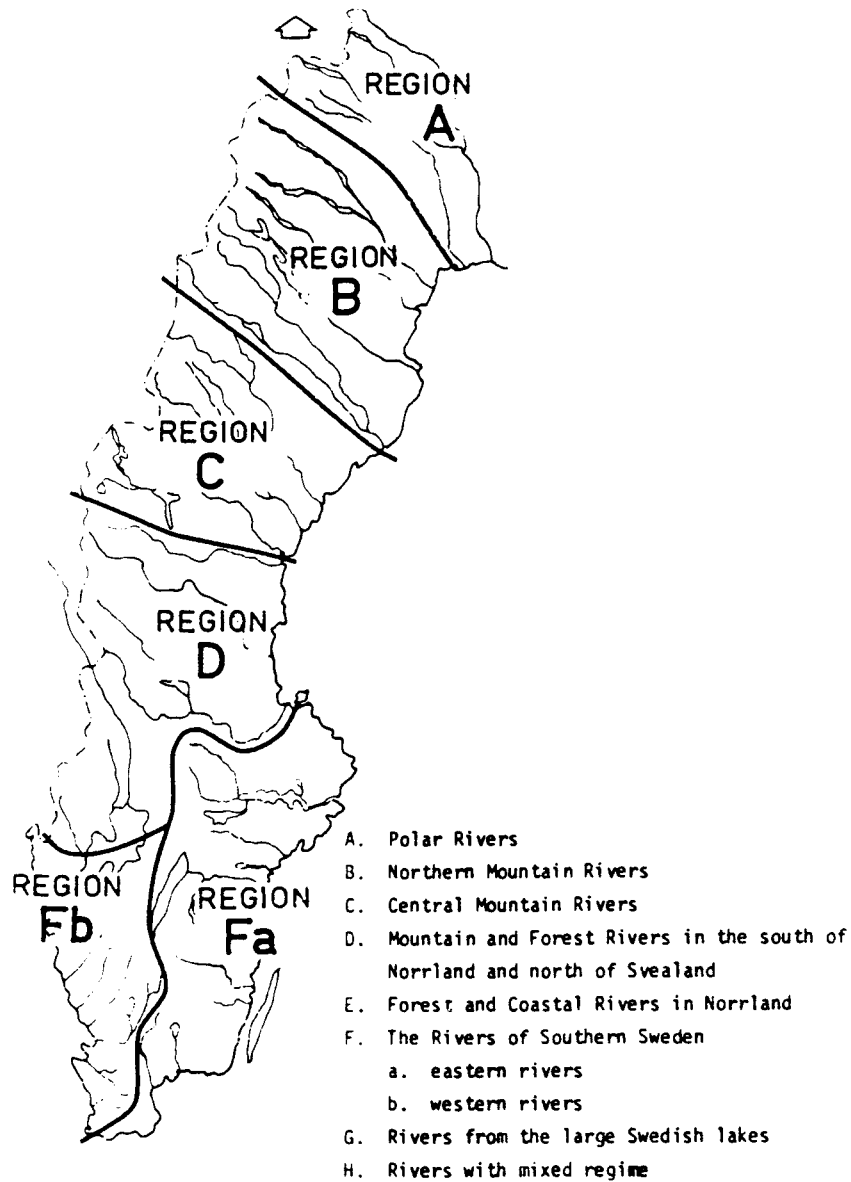
Figure 12. Hydrological regions according
to Melin (1970).

P O Agnedal

THE AGEING OF A LAKE -   CHANGES IN PLANT AND
ANIMAL LIFE

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CONTENTS

1   INTRODUCTION

A lake is not a stable formation but changes
during its life. Depending on the conditions of
its surroundings, it develops either into arable
land or into a peat bog.

Each step in this development has a character of
its own and the duration of each step can be
very different, but mostly the changes accelerate
towards the end.

The geo-morphological development of a lake is
described in another report /1/. Irrespective of
how a lake is formed it is initially poor in
nutrients. An exception, however, is when a
sea-bay, already rich in nutrients, is cut off
from the open sea.

A lake in an area that is rich in nutrients
normally ends up as arable land, but if it is
very rich in nutrients the silting up can
proceed very rapidly and the result can be a
rich fen which later develops into a peat bog
consisting of sedge and moss peat. If on the
other hand the lake is situated in an
environment of primary rocks and with no or
little supply of nutrients, it is influenced by
the surrounding woods and develops into a
dystrophic lake and later to a peat bog.

2   CHARACTERISTIC FEATURES OF LAKE TYPES

All the relevant types of lake can be found in
Sweden, and it is therefore possible to describe
the different stages in the development by using
existing lakes as examples. The oligotrophic
lakes are found in the northern part of Sweden
in the mountainous area, and also in the primary
rock areas in southern Sweden.

The eutrophic lakes are situated in rural areas
which mostly coincide with a background rich in
natural nutrients, but they also occur where a
poor environment is influenced by man.

The dystrophic lakes belong to areas surrounded
by moors or pine forests and situated in primary
rocks.

2.1   Water chemistry

2.1.1   Oligotrophic lakes

These lakes are most often clean and deep and
the shores are steep or narrow which prevents
rooted vegetation. The epilimnion is small
compared with the hypolimnion. The water has a
low conductivity due to the low concentration of
major constituents such as Na^+, K^+, Mg^{2+}, Ca^{2+},
CO_3^{2-}, SO_2^{4-} and Cl^-. The turbidity is also low
because of the small amount of plankton. It can
however increase owing to clay particles or
other allochtonous material.

The concentration of nitrogen and phosphorous
is low in oligotrophic lakes, which gives a
low production of phytoplankton. The oxygen
content in the water is high from the
surface to the bottom over the whole year,
as the amount of organic material on

the bottom is small and there is a low demand
for oxygen for decomposition.

2.1.2   Eutrophic lakes

In eutrophic lakes the concentration of major
constituents is high and normally calcium is the
dominant cation. The conductivity is therefore
higher. There is also a transport of nitrogen
and phosphorous to these lakes from the surroun-
dings, but as they are situated in populated
areas they also receive waste water and drainage
water from agricultural areas.

The oxygen concentration in the water of a
eutrophic lake varies during the year. A high
phytoplanktion production in the summer will
create oversaturated conditions in the epilimnion
and a deficit - sometimes no oxygen at all - in
the hypolimnion due to the high decomposition
rate of sedimenting organic material.

With the turn over in spring and autumn oxygen
is transported to the bottom water, but during
the stagnant period in the winter the oxygen is
consumed by decomposition.

During the periods with oxygen deficit in the
bottom water phosphorous is fixed in the sediment,
but it will be this is released when the oxygen
concentration increases. The transport of
phosphorous to the upper photic zone makes a new
production of phytoplankton possible.

The acidic precipitation during the last decades
has changed the pH values in many lakes in
Sweden, especially on the west coast and in
areas with primary rocks i.e. lakes with natural
low buffering capacity. It has been shown that the

decreasing pH-values change the conditions in
the lakes in such a way that elements e.g. Al
and U can be released from the bottom sediment.

Eutrophic lakes are not, however, as vulnerable
as oligotrophic ones because of the high
alkalinity values and high buffering capacity
due to the high concentration of calcium.

2.2   Phytoplankton

Lakes can be classified not only from a chemical
point of view but also from the species and
composition of phytoplankton.

In an oligotrophic lake the amount of plankton
is small, but it can be found not only in the
upper layer but also in deeper water. In a
eutrophic lake the presence of phytoplankton is
limited to the upper layer, at least during the
summer.

Green algae dominate over blue-green algae in an
oligotrophic lake while in a eutrophic one the
condition is reversed.

The seasonal variation of phytoplankton depends
on factors such as i.a. temperature, light and
nutrients. In the spring when the light increases,
already below the ice a very rapid increase in
the biomass - most by diatoms - occurs resulting
in a water-bloom. This is an effect of the
vertical mixing which has transported i.a.
phosphorous to the surface layer; other elements
have also reached their maxima at this time. The
dead phytoplankton settle and some are decomposed
before reaching the metalimnion, while others
settle on the bottom. In the first case the
nutrient will remain in the photic zone and can
be used again, whereas in the second case the

nutrients are bound to the sediment until the
next vertical mixing.

On account of the low concentration of nutrients
during the summer the standing crop of phyto-
plankton is rather low. After autumnal vertical
mixing a new maximum of phytoplankton can occur.
This bloom is normally not as large as that in
the spring because of the smaller amount of
light and a transport of cells from the photic
zone due to circulation. One notable observation
is that, according to many investigations, the
biomass of nannoplankton is larger in
oligotrophic lakes than in eutrophic ones. It
has also been found that with increasing
eutrophication, the size of the phytoplanton
increases /2/. Increasing eutrophication also
lead to a change in the composition of
phytoplankton species. (Fig 1).

2.3   Rooted vegetation

The presence of rooted vegetation in a lake
depends both on the formation of the shore line
and the bottom material in the trophogenic zone.
The vegetation is also different on wind exposed
shores compared with sheltered areas.

As already mentioned, oligotrophic lakes very
often have shores that slope more abruptly and
the area of rooted vegetation is therefore
rather small. In the shallower eutropic lakes
the bottom configuration is different and the
area available for rooted vegetation is much
broader (fig 2). On a wind exposed shore there
will be an erosion of the bottom material which
is transported to deeper areas, where sedimenta-
tion of fine material takes place. Between these
two zones there is a neutral zone where sediment
is collecting at the same rate as erosion occurs.

These circumstances are important for the possibilities of plants becoming rooted in the different areas.

Emergent plants like Phragmites and Typha can also exist on exposed shores and Ph. can grow in soil above the water surface and down to about 2 m depth. Thypa is more easily injured by wind and waves and is therefore more often found in sheltered areas. Outside the zone of emergent plants one will find floating-leaved plants such as Nymphea and Nuphar and also Potamogeton natans. These plants are rooted in sediment and can thus only live in areas where the bottom is stable enough and contains organic substances.

In deeper water the rooted vegetation consists of submerged plants such as Myriophyllym and Ceratophyllum. In the erosion zone in an oligotrophic lake the rooted vegetation is characterized by species of Isoetes, Lobelia and Litorella (fig 3).

The depth to which the vegetation extends depends i.a. on the light conditions, but most often mosses are the plants living at the greatest depth. (Fontinalis antipyretica, Drepanocladus).

The zonation of the vegetation in a fairly typical eutrophic lake is shown in figure 4.

2.4   Bottom fauna

As a result of changes in the chemical conditions and the spread of vegetation, the bottom fauna also changes in the evolution from oligotrophic to eutrophic conditions.

Of primary importance is the oxygen condition.
If the oxygen concentration is high and there is
no deficiency in the bottom water the bottom
fauna is characterized by species demanding high
oxygen concentrations like Tanytarsus /3/. The
fauna is also rich in species and the biomass is
relatively large.

In a eutrophic lake the bottom water can have an
oxygen deficit during stagnation periods which
prevents species with high oxygen demand from
existing in this biotope. The fauna is therefore
poor in species but the biomass is large.

According to Thienemann the above fauna is
denoted a chironomid-fauna, being named after
the most prevalent species.

One animal always present in such lakes is the
Corethra larvae which is the only planktonic
living insect-larvae in our lakes. With
increasing vegetation in the littoral zone a
good shelter and a good food supply is
developed which increases the biomass of the
bottom fauna.

2.5   Fish fauna

Not only the invertebrate fauna is but also the
fish fauna changes with the eutrophication.
Oligotrophic lakes are often very deep and have
cold water, at least in hypolimnion. The fish
fauna is thus composed of cold stenotherm
species such as Coregonus and Salmo spp. These
species are plankton feeders or carninvores.

In eutrophic lakes with warmer water and more
vegetation, species of the genera Cyprinus and
Leuciscus as well as pike and pike-perch are

dominant. These species are more adapted to warm
water and the first two genera mentioned feed on
vegetation and small animals living in or on the
vegetation belt. They are also more tolerant to
a low oxygen content in the water, but even
these species can not survive in too low concentra-
tions and can be killed during the winter
stagnation period.

2.6   Dysotrophic lakes

These lakes can be characterized in the follwing
way.

They are poor in nutrients and the oxygen
concentration in the deep water is in summer and
winter zero or near zero, due to the decomposi-
tion of floating and settling allocthonous humic
material.

The vegetation along the shores is not rich and
there is often a Sphagnum belt around the lake.
The amount of phytoplankton is mostly small but
the zooplankton is fairly rich. The bottom fauna
is very scarce because of the unsuitable bottom
type and low oxygen concentration.

With the increase of settling allocthonous
material, which causes a decreasing depth and an
increasing Sphagnum belt the lake will develop
into peat bogs, figure 5.

3   FACTORS AFFECTING EVOLUTION

The evolution from the oligtrophic to the eutrophic stage of a lake is influenced by many factors. The main factor is that the lake will be more shallow, and this condition can be reached in different ways. It is very difficult to separate the different factors affecting the development as they are very interconnected. The following list will try to give an idea of how the development can occur.

1) The lake will receive allochtonous material which settles on the bottom and diminishes the depth. This can also occur by sedimentation of dead plankton and parts of rooted vegetation (=autoch- tonous material).

2) By erosion the shore line can be broadened and more suitable for rooted vegetation which in its turn reduces the action of the waves on the exposed shore, so that more material can settle in this area.

3) The inflow of nutrients from surrounding agricultural areas or with waste water increases the production.

4) With diminishing depth the rooted vegetation of emergent plants and also floating-leaved plants can spread further in pace with light reaching the bottom.

5) With increasing vegetation the water movement will be reduced and the organic material produced will accumu- late on the bottom and the silting up will be enhanced.

6) The decreasing water depth will make it possible for grass and herb species and later bush vegetation to invade the fringe area.

The process described above is slow from the beginning but accelerates in the end. Once the lake has reached a high eutrophic level, which can take thousands of years, the development to

arable land can pass in a few decades or hundreds
of years.

If the amount of organic material settling on
the bottom is very large the lake might develop
to a swamp which later will be transformed to a
peat bog.

4   PRODUCTION

4.1   Phytoplankton

In connection with the description of
phytoplankton it was mentioned that the primary
production was low in oligotrophic lakes because
of the low concentrations of nutrients. Gessner
/4/ has given values from some Danish lakes
where the primary production by phytoplankton
has been beasured by the C-14 method.

	Tiefe m	Elektrolyt-Gehalt ($\varkappa_{20} \cdot 10^6$)	Primärprod. Jahresmittel mg C/m²/Tag	Compound-Index
Grane Langsö	11	49— 50	260	0,3— 0,6
Kattehale Mose	1,5	76— 83	80	0,7— 0,85
Store Gribsö	11	128—131	230	4
Fureso	36	290—310	770	2,3— 4
Frederiksborg Slotssö	5	360—410	1320	9 —11,7

As can be seen lakes with higher conductivity
also have a higher production per year.

Ross and Kalff /5/ have measured the production
of phytoplankton in a Canadian lake and obtained
an annual production of 175-219 g C/m^2 or an
annual average of 480-600 mg C/m^2 · d. The daily
average summer production varied between 794 and
1078 mg C/m^2 d. The great difference was due to
the conditions in different places in the lake.

The daily production in several European lakes
has been compiled by Rodhe /6/ who has also
shown the correlation between production per m^2
and per m^3. All values (figure 6) are achieved
during optimal light conditions and they vary
from 20 to >2000 mg C/m^2 · d.

The production is of course also dependent on
the biomass, but not on the number of cells as
in the sea where there is a fairly constant
relation between net and nannoplankton. In lakes
the production is correlated to the weight of
the total phytoplankton biomass, figure 7. As an
example of a very high production the figures
from Lake Trummen in Sweden can be mentioned
/7/. Before the lake was restored the production
by phytoplankton was 10 g C/m^3 · d and was
concentrated in the upper 20 cm. The euphotic
zone was at that time less than 0.5 m. Only
20-40 % of the production was performed by algae
able to pass through a 45 μm plankton net. The
average annual productivity was 375 g C/m^2
before restoration and dropped to 225 g C/m^2.
More than 60 % was however produced by algae
less than 10 μm.

4.2　　Rooted vegetation

The production of rooted vegetation has not been
measured to such an extent as that of
phytoplankton and the information is therefore
much less. A few examples will however be
reported to enable a comparison to be made with
the values for phytoplankton.

Schierup /8/ has investigated the total annual
net production of Phragmites in a Danish bog and
obtained a value of 1760 g dry weight/m^2 · a
including both aerial parts and rhizomes.

Penfound /9/, quoted in ref 4 reports an
American investigation regarding the production
of Typha latifolia which gave values of 3.84 to
10.94 g C/m^2 · d. The large variation is due
to the values being from different times
of the year. The production of floating-
leaved and submerged vegetation has

been investigated by Nygard /10/ who has found
that a biomass of Nitella of 20-40 g dw/m^2
produced an organic material corresponding to
150-300 mg C/m^2 · d. In Green Lake, Wisconsin,
the production of rooted vegetation is
50 kg d w/har and the phytoplankton has an
annual production of 2650 kg/har /11/.

In Mirror Lake, New Hampshire, USA, which is a
lake that can be compared with northern European
ones, the biomass has been investigated and
calculated /12/. The investigator has divided
the littoral into three zones. The first 0-2 m
has a biomass of about 15 g d w/m^2. The second
one 2-5 m has a much lower biomass, 1.5 g d w/m^2.
The last zone 5-7.2 m shows a higher biomass value
which depends on Nitella stands that locally
exceed 40 g d w/m^2. For the lake as a whole the
hydrophyte biomass is 4.2 g d w/m^2 or 7.1 g d w/m^2
if only the colonized zone is used as area. The
net production can be estimated to 1.6 g C/m^2 · y.

4.3   Zooplankton

The production of zooplankton is not so easy to
determine and the information is again fairly
sparse, but as an example the figures reported
by Culver and De Mott /13/ from Lake Eire can be
mentioned. They calculated the annual production
to be 99.4 g d w/m^2. In the above mentioned
Green Lake the production of zooplankton during
a year is 5.3 g d w/m^2 /11/.

4.4   Bottom fauna

The variation in the biomass of bottom fauna can be
very great; it is very dependent on which species
are present. In an oligotrophic lake the weight
is estimated to be about 1.24 g/m^2 whereas in a
eutrophic lake the weight can be 20-100 g/m^2.

It is therefore of little interest in this
report to give details of the composition and
related weights of the bottom fauna in different
lakes. However, Table 1 gives an idea of the
variations that can exist.

4.5   Fish

Finally the production of fish or the catch for
human consumption from different types of lakes
could be of interest for possible future work.
In figure 8 the fish catch in kg/har $\cdot$ y in
relation to the average depth of the lake is
given.

As can be seen, the catch decreases rapidly with
increasing depth. However, the increased catch
is not necessarily so valuable as could be
desired since the composition of fish species
can change to less valuable ones, as is illustrated
in figure 9.

5   PEAT

5.1   The formation of peat

When a lake has reached the state of a bog the
bottom is often covered with unfouled plant
material. The formation of peat will occur when
aerobic microorganisms, bacteria and fungi can
decompose the organic material. This process is
fairly quick when the material is in the air
above the soil. In water the process is slower
or impossible due to the low oxygen content.
With a high settling rate the organic material
will not be decomposed before new material has
covered the bottom. With a lower groundwater
level or a land uplift the gyttja will be
oxygenized and the decomposition by bacteria can
start again, which will result in the formation
of highly humified peat even in the deeper
layers.

The dead plant material contains cellulose,
hemicellulose, lignin, wax and resin as well as
nitrogenous proteins and amino acids.

Wax and resin are very resistant material and
will be unchanged during the peat formation,
contrary to the case for cellulose and lignin.
The relative concentration of the first two will
therefore increase during the process as will
also the nitrogenous substances.

The quality of the peat depends therefore on the
chemical composition of the peat-forming plants
and how far the humification has advanced.

5.2   Types of bogs

Depending on the varying adaptation of the peat-
forming plants to different environments, the
peats are divided into ombrogene, soligene

and topogene types. The way in which the water
is supplied is crucial for the formation.

Ombrogene areas   will receive their nutrients
                    and water from precipitation
                    and therefore form a hydrographic
                    unit that is separated from
                    the surroundings.

Soligene areas   receive their water supply from
                    the environment and percolation
                    over the surface.

Topogene areas   are formed and developed
                    as a result of their location
                    in the environment.

The peat formation in these three main types can
occur either by intrusion of higher plants into
a lake or by embogging of previously firm soil.

In the south of Sweden the common development
has been that a topogene peat area changes to an
ombrogene peat bog.

This evolution can be described in the following
way.

During Atlantic time a lake was silting up due
to evaporation and a high sedimentation rate. At
the end of this period a large part of the lake
surface was covered by Phragmites and Scirpus.
During the following period, the subboreal period,
the climate became dryer and a leafy marsh was
formed above the overgrown parts of the lake.
Gradually towards the end of this period a wood
of pine and birch with moss had occupied the
leafy marsh and formed a dense forest over the

whole bog. Nowadays a stump horizon in the peat
may in many places reveal the existence of this
wood.

With the change in climate in subatlantic time
the white moss (Sphagnum) invades the leafy
marsh. The trees will die and a peat-moss is
formed, and Carex-bogs appear in the lagg open
water at the edge of a bog.

Most of the bogs in Sweden are a combination of
the three main types described above.

6. CONCLUSIONS

The uptake of different elements and also of radioactive nuclides varies with the chemical composition of the water in a recipient. The uptake varies with different species of both plants and animals.

It is therefore important to have a thorough knowledge of conditions such as pH, redox-potential, nutrient contents, composition of flora and fauna in order to be able to calculate how much of an element will be accumulated by plants and animals and how much will be transported to the sediment after decomposition.

The uptake is also dependent on the total production of organic material in the different trophic layers.

As has been reported, the production can vary very much both during the year and in each trophic layer.

In an American lake the production of rooted vegetation was 50 kg dw/har per year and that of phytoplankton 2560 kg dw/har. The fish catch varies from less than 5 kg/har up to 20 kg/har depending on the type of lake.

In order to be able to calculate the amount of an element bound in each trophic layer a knowledge of the production seems necessary.

It can therefore be deemed necessary to study the transport of some elements in the different lake types and also in the peat, and from these studies try to draw conclusions about the behaviour of those radionuclides which might be of interest.
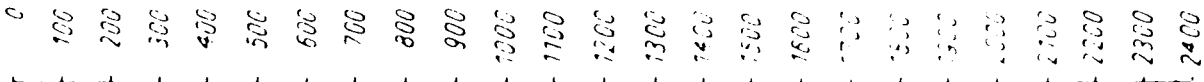
Comparable laboratory studies of the transport
mechanisms should also be carried out step by
step in experiments where the lake types are
simulated.

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Table 1

	100	200	300	400	500	600	700	800	900	1000	1100	1200	1300	1400	1500	1600					2700	2800	2300	2400

■ 34.7 – Mittel von 2.8 – 240.6 für 16 schwedische Seen (Schlamm)

■■ 63.5 – Mittel von 0.7 – 324.3 für 18 finnische Seen

■■ 59.2 – Mittel von 21.2 – 100.6 für 10 norwegische Seen

█████ 798.5 – Mittel von 13.1 – 4061.3 für 57 norddeutsche Seen

██ 140 – Green Lake in Nordamerika.

█████ 496.8 – Mendota-See in Nordamerika (Schlamm).

████ 287 – die Seen am Jllinois. Mittel von 64 – 952

█████ 744 – Fluß Jllinois. Mittel von 12 – 3030

████ 477 – Jlmensee Mittel von 285 – 668 ⎫

■■ 94.1 – Tschana-See

■■ 50.6 – Weißer-See ⎫ Moskauer Gebiet ⎬ UdSSR
(s. weiter unten)

■ 10.2 – Schwarzer-See ⎭

████ 180 – Buseroboje-See ⎭

█████ 933 – Mittel von 241.3 – 1626.2 für das Delta der Wolga
bei schwacher Strömung.

■ 54.9 – Mittel von 4.4 – 105.2 für das Delta der Wolga

██████ 2097.6 –
(max. 34090.2) schnell fließender Abschnitt der Wolga.

████ 139.7 – Mittel von 6.2 – 273.8 für Abschnitte der Wolga mit mäßiger Strömung.

■■ 75.4 – Mittel von 23.0 – 127.8 für die Überschwemmungsflachen der Wolga

█████ 1376 – Kaspisches Meer (süßwasser-
haltiges Küstengebiet).

████ 650 – Brackwassergebiet an der Mündung des Kaspischen Meeres.
Mittel von 52.5 – 416.8.

■ 28.1 – Kaspisches Meer (Mündungsgebiet mit bedeutendem Salzgehalt).

████ 198.8 – Kaspisches Meer. Mittel von 114.1 – 285.5 der Meereszone an der
Mündung der Wolga.

██████ 1600 – Mittel von 200 – 3000 und
bis zu 33941.25 der Barents-See. in verschiedenen Biozönosen in der Maschiginer Grube bei Nowaja
████ 500 – Karisches Meer. Semlja.

████ 270 – Mittel von 50 – 610 des Weißen Meeres in verschiedenen Biozönosen

████ 375.8 – Mittel der Ostsee von 0.0 – 1110, in der westl. Ostsee 950 – 1110 in den
übrigen Gebieten im Mittel 22.5 – 140 – 112.2, bei Gotland Maximum 612.

████ 2670 – Mittel des Asowschen Meeres, in versch. Biozönosen 750 – 910 – 1750 – 3530 ⌐7450

████ 8000 kg Muschelfleisch bei Bewirtschaftung – 37% des Gewichtes
mit den Schalen.

█████ 2238.8
Mittel von 574.6 – 3838.1 für die Küsten Dänemarks bei Limfjord.

█████ 10425 – Mittel von 32 – 187840 für den Großen Belt im Kattegat.

█ 2437 – Mittel von 108.8 – 21254.3 in der Nordsee

████ 753.4 – Mittel des südwestlichen Teils der Nordsee in verschiedenen
Biozönosen 69 – 251 – 281 – 307 – 981 – 1054 – 2328

Wolga und Kaspisches Meer

Biomasse der wirbellosen Tiere in kg/ha für verschiedene Binnenseen und Meere.
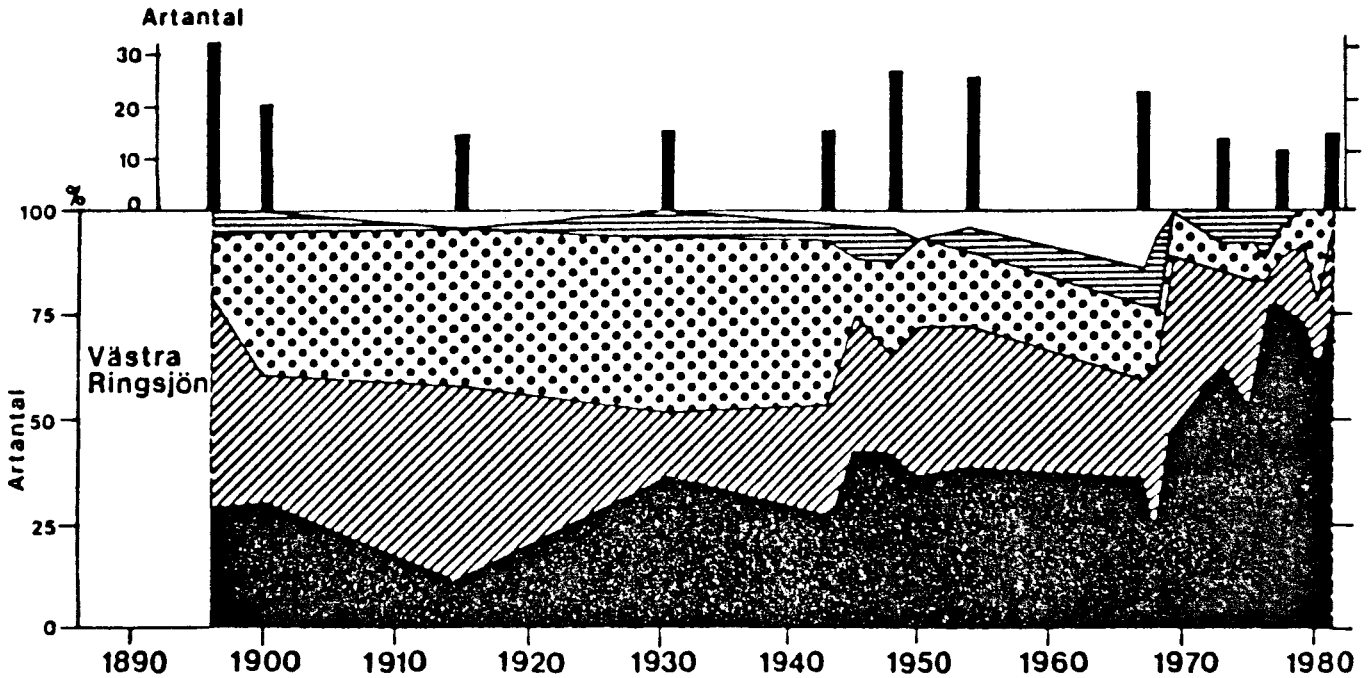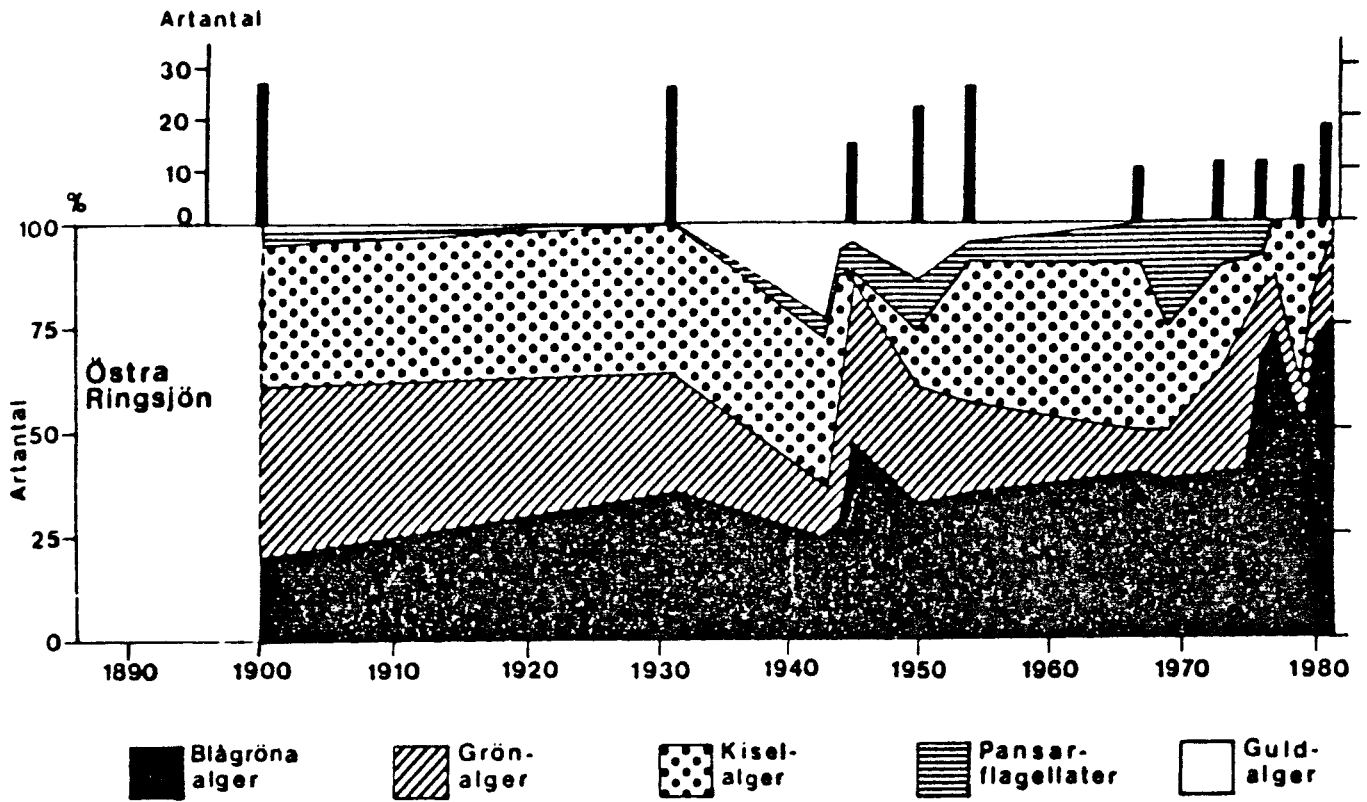
From ref 11

Figure 1. Phytoplankton succession in East and West Ringsjön 1896-198
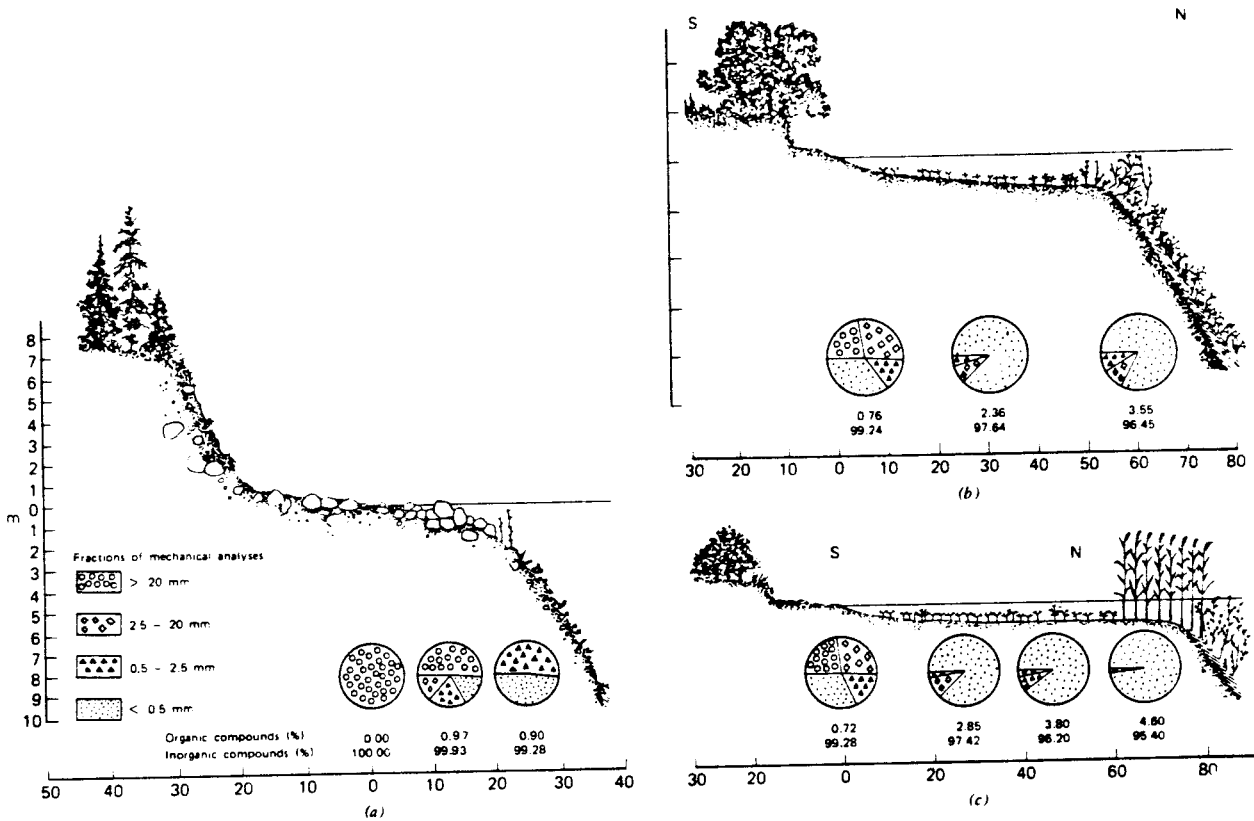(IV)

From ref 2

Fig 2. Profile across littoral region of (*A*) eastern shore of Lake Hańcza, a typical litholittoral; a little *Potamogeton pectinatus* rooted at about 2 m, and below if *Elodea canadensis* to about 9 m. (*B*) eastern shore of Lake Dargin, a typical psammolittoral; *Parvocharacetum* on sand in up to 1 m of water, *P. pectinatus* and *P. perfoliatus* at top of mont rooted in about 1 m of water, *E. canadensis* and Charophyceae at greater depths. (*C*) protected southern shore of same lake with psammolittoral bearing a narrow belt of *Phragmites australis* at the lakeward edge of the shelf, with *P. compressus, P. perfoliatus, P. lucens,* and *Ranunculus circinatus* rooted in 2-4 m water.

Sectored circles refer to the composition of the sediments at the points immediately above their centers (Bernatowicz and Zachwieja).
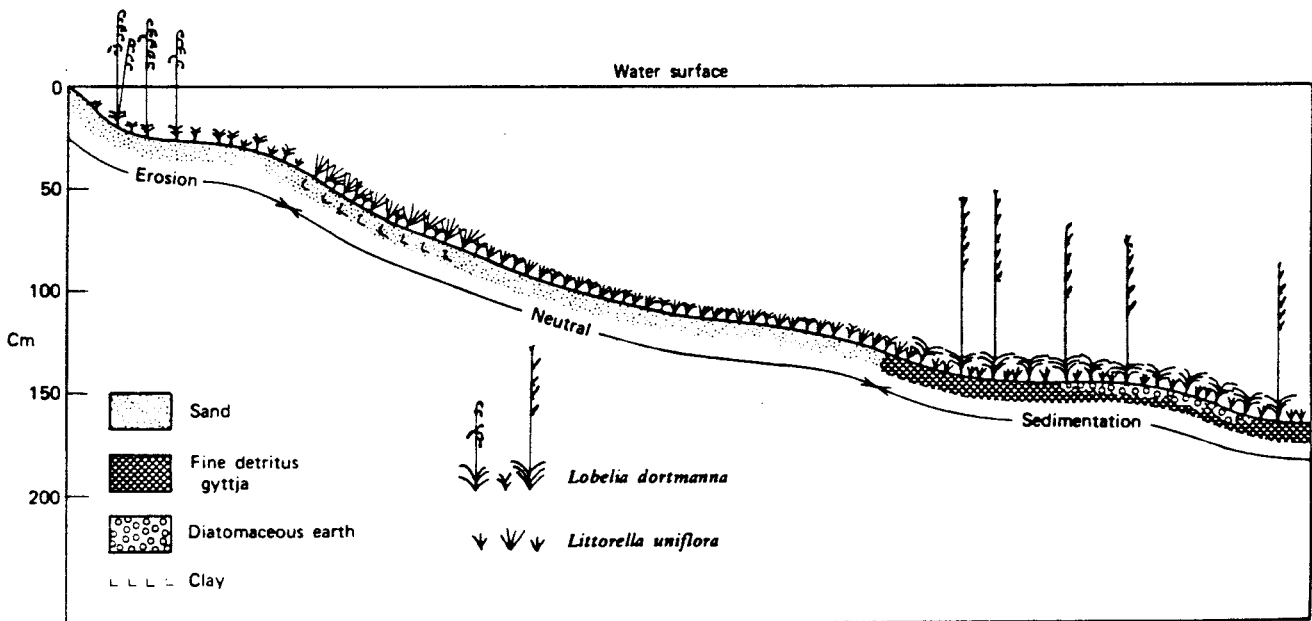
From ref 2

Fig 3. Distribution of *Littorella uniflora* and of *Lobelia dortmanna* on the protected western side of Lake Fiolen. Note the two zones of flowering of *L. dortmanna*.
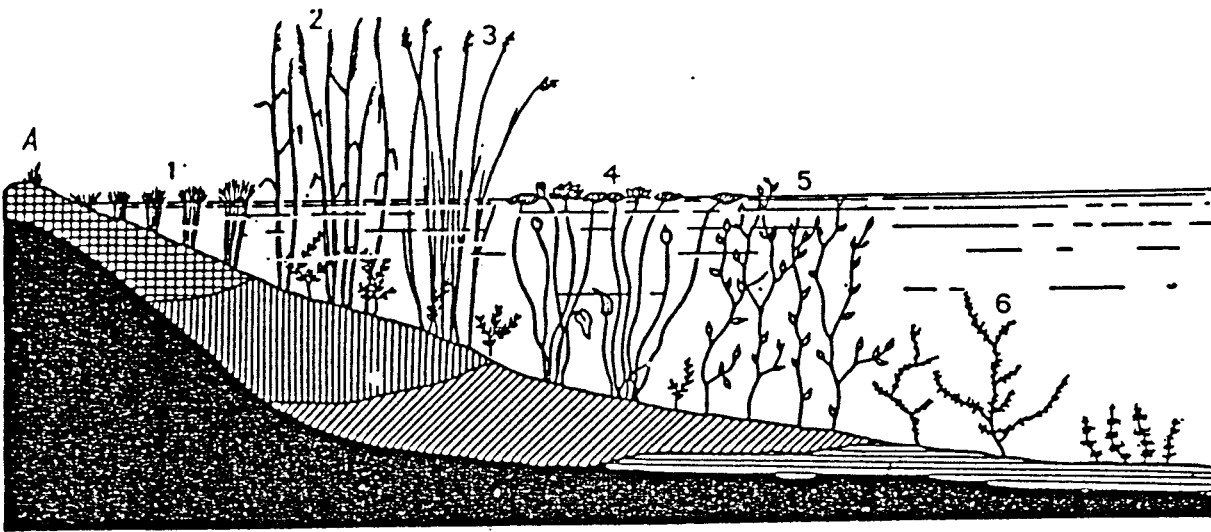
From ref 2

Fig 4. Schema der Verlandung eines Sees.

I Riedgraszone; *2* Zone des Schilfes, bis zu 2 m Tiefe; *3* Zone des Rohrs, bis zu 3 m Tiefe; *4* Zone der Seerosen, bis zu 4 m; *5* Zone der Laichkrautgewächse, bis zu einer Tiefe von 6 m; *6* Zone der Characeen, bis zu 8 m Tiefe; unter den Riedgräsern (gekreuzt gestrichelt) lagert sich Riedgrastorf ab; unter dem Schilf und dem Rohr findet sich Schilf- und Rohrtorf (vertikal gestrichelt); unter den Zonen der Seerosen und der Laichkrautgewächse liegt Sapropeltorf (schräg gestrichelt); unter der Zone der Characeen Sapropelit (horizontal gestrichelt); schwarz ausgezogen der Boden des Sees.

From ref 11
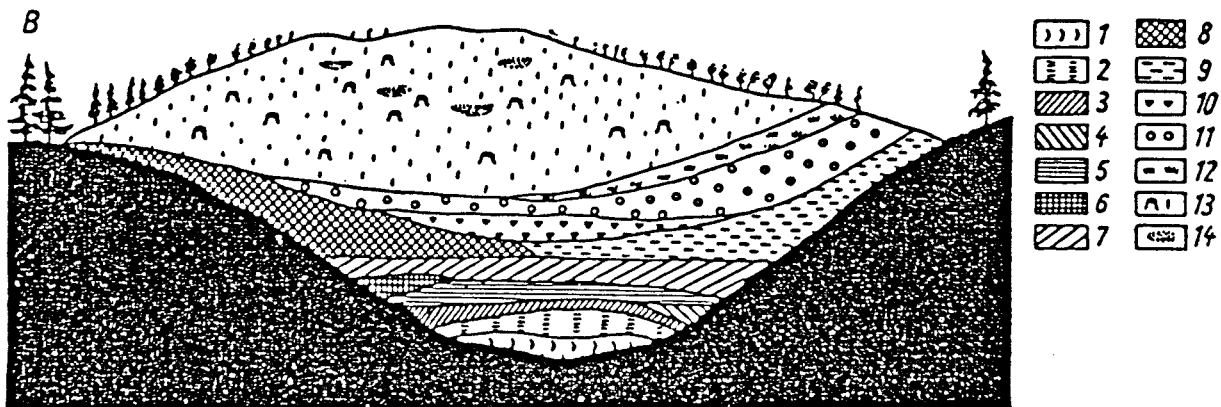
Fig 5. Schema eines Hochmoores, das durch die Verlandung eines Sees entstand.

I Süßwassermergel; *2* Sapropelit; *3* Sapropeltorf; *4* Schilftorf; *5* Rohrtorf; *6* Schachtelhalmtorf; *7* Riedgrastorf; *8* Riedgrasweidentorf; *9* Waldtorf; *10* Hypnotorf; *11* Scheuchzeriaccentorf; *12* Sphagnumtorf; *13* Sphagnumtorf mit Fichtenstubben; *14* Wasserlachen auf dem Sphagnumteppich; schwarz ausgezogen der Boden des ursprünglichen Sees.

From ref 11

mg C/m³ Tag
(max.)
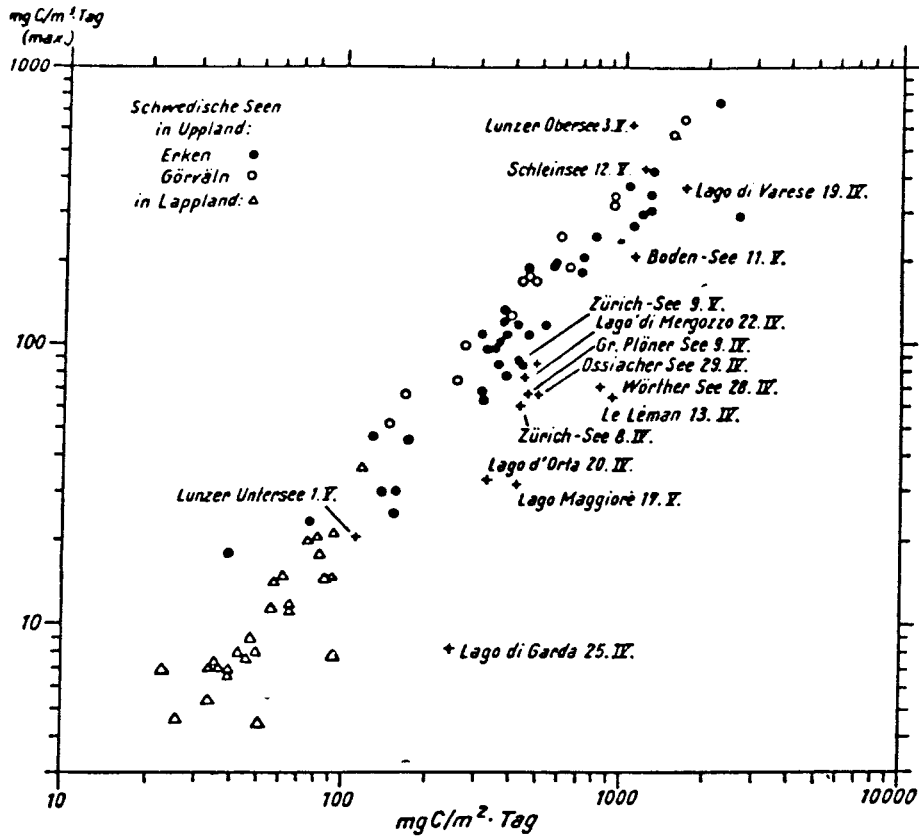
Fig 6. Tagesraten der Primärproduktion in mg C/m² in Beziehung zum Volumen maximaler Leistung (mg C/m³) unter optimalen Lichtbedingungen in verschiedenen Seen Europas, gemessen mit der ¹⁴C-Methode in 24 Stunden in situ. (Nach RODHE 1958.)

From ref 6

mg C/m³
(max.)
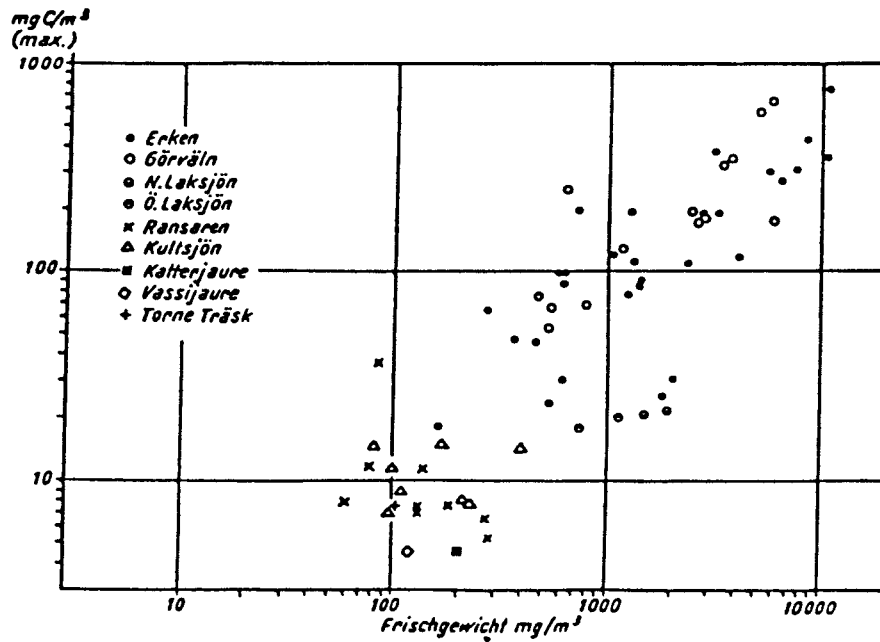
Fig 7. Beziehung zwischen der täglichen Primärproduktion im lichtoptimalen Kubik-meter und der durchschnittlichen Biomasse des Phytoplanktons in der trophogenen Schicht schwedischer Seen. (Nach RODHE 1958.)

From ref 4
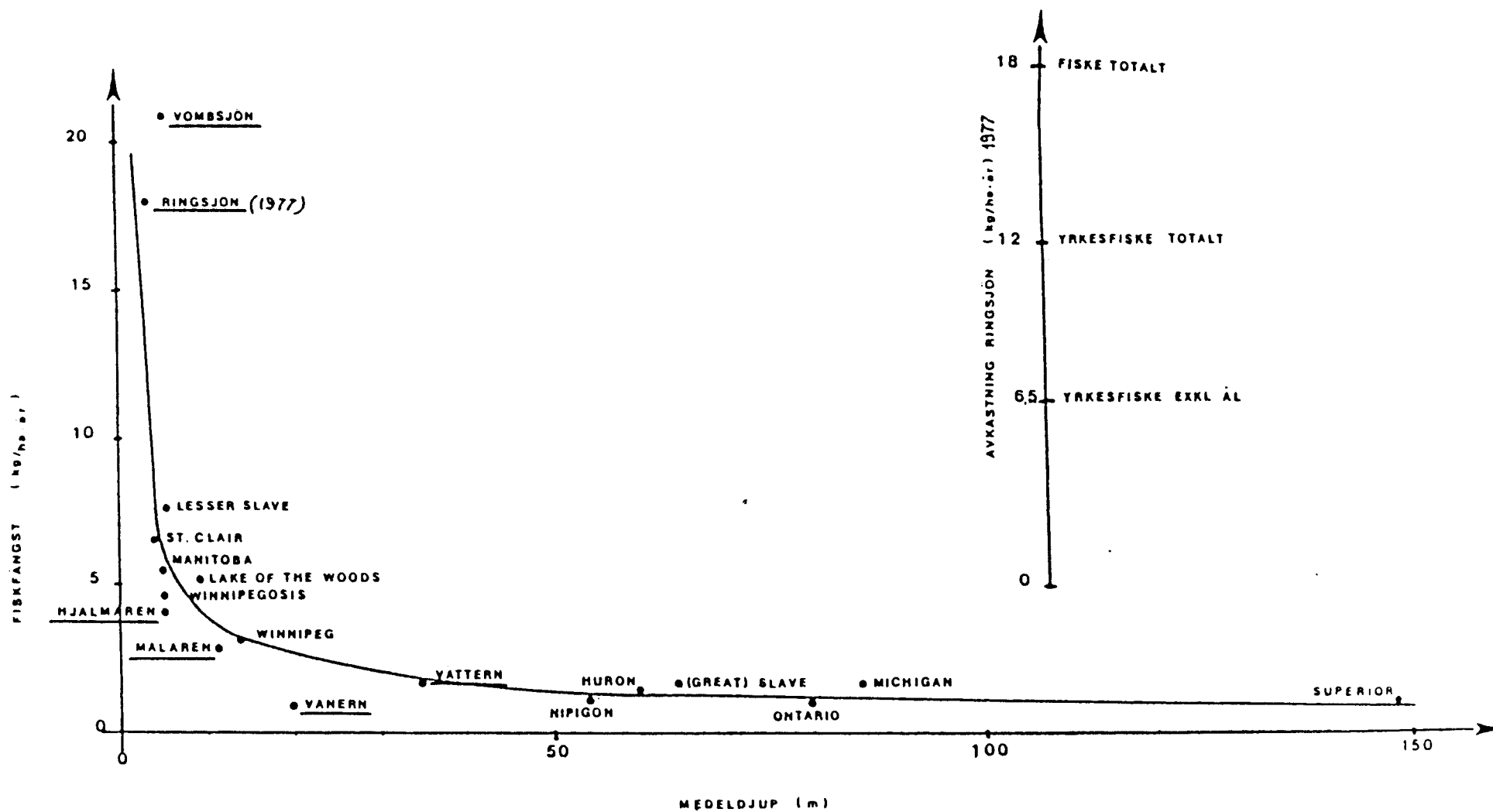
Fig 8. Relation between mean depth and fish production for some lakes (VIII).

From ref 2
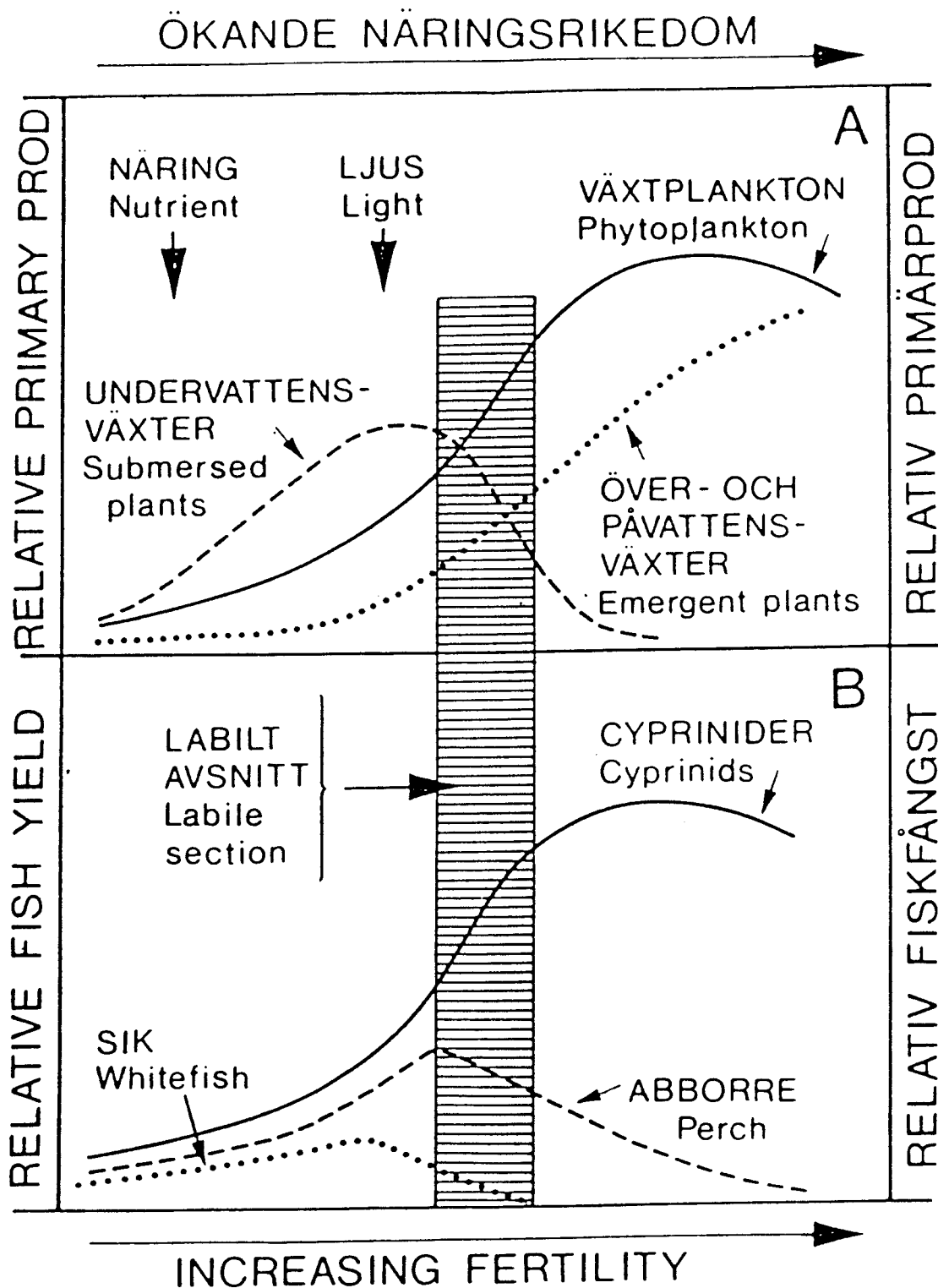
Fig 9. The general relationship between changes in primary production
between phytoplankton, submerged plants and emergent plants
and the catches of whitefish, perch and cyprinids with
increasing nutrient supply in shallow lakes.

(Ref 2)

Sverker Evans

ASPECTS OF SEDIMENTATION IN LAKES AND COASTAL
WATERS

ABSTRACT

Because of strong association in general with
suspended particles, sediments are normally the
major transport vector for pollutants in an
aquatic environment. The sediments also act as a
sink, effectively removing pollutants from the
water column. They may also become a source of
pollution long after the releases of pollutants
to the recipient have ceased. The structure and
composition of a lake or bay sediment is a
function of the characteristics of the catchment
area and the water column. However, the sedi-
ments laid down are also subjected to various
short-term pertubations such as resuspension,
bioturbation and gas convection. Thus, the
uppermost layer of most sediments reflects
events occuring near at hand, while the sedi-
ments from a defined layer yield information
integrated over time. An exceptional case is the
formation of varved lake sediments in some
Scandinavian lakes which allows calculation of
accurate annual deposition values. The major
factors governing the distribution of a dissolv-
ed constituent in the interstitial water of
unconsolidated sediments are diffusion, vertical
advection due to sedimentation and compaction,
and ion exchange. Furthermore, bioturbation and
near bottom currents will induce changes in the
pore water composition. Efforts to make long-term
forecasts on sediment growth and the subsequent
accumulation of elements in the deposits require
parameters based on field measurements, and data
extrapolated from laboratory experiments.

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1.   INTRODUCTION

The whole landscape of the subarctic region is
formed by the glacial period and by the tendency
of the land mass to rise after being depressed by
the ice shield. All lakes in northern Europe have
been influenced by glacial activity and the basins
are constantly under alteration by exogenous forces
such as weathering, erosion, transport and depo-
sition. The sedimentation rate is a function of
the surrounding land area from which material is
eroded, and the size of the water body in which
this material is deposited. Consequently, the
sedimentation rate in enclosed or semienclosed
water bodies is much higher than the average sedi-
mentation rate in the oceans. The rapid sedimen-
tation and the heavy load of eroded material ren-
der these basins, in terms of geological time,
shortlived phenomena. Human impingements on lake
development such as agricultural activity will
furthermore shorten the life-time of lakes and
watercourses. It is the ultimate fate of all
lakes to be filled up by sediments and in due
course be transformed into land.

The role of the sediment is important for several
reasons. Because of strong association in general
with particles, sediments are normally the major
transport vector of pollutants in the aquatic en-
vironment. The sediments also act as a sink, ef-
fectively removing pollutants from the water
column. They may also become a source of pollution
long after the releases of pollutants to the re-
cipient have ceased. Thus, the study of sedimen-
tation processes in estuarine and lacustrine en-
vironments is of major importance for assessments
of the accumulation of elements in various reser-
voirs, and for long-term risk assessments.

2.   LAKE CLASSIFICATION

A scheme where lakes are classified according to
their trophic level in relation to the rate of
supply of organic matter from allochthonous and
autochthonous sources is given in Figure 1. Oligo-
trophic lakes are characterised by a low primary
production and low concentrations of nutrients.
Eutrophic lakes in general have a high pro-
ductivity and high concentrations of phosphorous
and nitrogen. Dystrophic or polyhumuos lakes have
low productivity and receive a major part of their
organic input from surrounding drainage areas. The
origin and composition of the sediments in the dif-
ferent lake types may vary accordingly. However,
it should be emphasized that in nature there exist
only continuous transitions between various lake
types.

3.   PHYSICAL AND CHEMICAL SEDIMENT PAR-
          AMETERS

Sediments, like lakes and estuaries, can be
classified according to several principles and
by means of numerous parameters. A limited num-
ber of physical sediment parameters will be de-
fined. These are: water content, organic content,
bulk density and grain size. Most other physical
sediment parameters like porosity, permeability
and compaction may be defined in terms of these
four basic parameters (Håkanson and Jansson, 1983).

3.1   Water content

The water content can be defined as the weight of
water related to the total wet weight, i.e.

$$W = \frac{W_w}{W_t} \cdot 100 \qquad\qquad (Eq\ 1)$$

where

W = the water content (%)

W_t = the total weight (g)

W_w = the weight of water (g).

The water content of surface sediments varies from
about 30 - 50 % in minerogenic deposits to approxi-
mately 95 - 99 % in highly organic sediments. The
water content is distributed in a typical way in
lake sediments. Low values are found in shallow
water where coarse material often dominates, while
high water contents predominate in deeper parts
of the basins. The water content decreases down-
wards in sediment cores due to compaction.

3.2   Organic content

The loss on ignition is frequently used as a measure of the organic content of lake sediments.

$$IG = \frac{W_s - W_r}{W_s} \cdot 100 \qquad \text{(Eq 2)}$$

where

IG = loss on ignition in % of the weight of the solid particles (W_s)

W_r = weight of the inorganic residue (g).

3.4   Bulk density

The bulk (= wet) density (ρ) can be determined by the theoretical formula (Axelsson and Håkanson, 1971):

$$\rho = \frac{100 \cdot \rho_m}{100 + (W + IG^o)(\rho_m - 1)} \qquad \text{(Eq 3)}$$

where

ρ = the bulk density of the sediment (g cm^{-3})

ρ_m = the density of the solid particles (g cm^{-3})

IG^o = the loss on ignition expressed in % of the total wet weight.

This formula is based on the assumption that the density of the water as well as the organic material in the saturated sediment can be set at 1.00 g cm^{-3}. Then the content of inorganic materials (W_s^o) in % of the total wet weight (W_t) can be expressed:

$$W_s^o = 100 - W - IG^o \qquad \text{(Eq 4)}$$

$$W_s^o = (\frac{100}{\rho} - \frac{W}{1} - \frac{IG}{1})\rho_m \qquad \text{(Eq 5)}$$

The relationship between bulk density (ρ), water content (W), organic content (IG°), and the density of the inorganic particles (ρ_m) is thus theoretically described by Eq 3. Once attached to the sediment bed the mud particles undergo a process of initial consolidation, i.e. a squeezing out of the water between the particles and a gradual increase in density.

3.5   Grain size

In lake sedimentology, the following three grain size classes are the most important (Müller and Förstner, 1968a, b):

sand > 63 µm

silt 2-63 µm

clay < 2 µm

The main grain size is an important tool for the interpretation of sediment data in relation to bottom dynamics. Finer particles dominate in deep waters, where continuous accumulation prevails. In shallow waters, where erosion and transportation dominate, the material is generally coarser. The grain size of the material influences the sedimentation processes and the capacity for entrainment, as well as the capacity of the material to bind pollutants. The potentially active binding area increases significantly with decreasing particle size. (Figure 2).

4.   SEDIMENT COMPOSITION

Sediments are formed mainly from material supplied from the terrestrial surroundings of the lake or estuary and from material synthetized in the water column. The sediments contain the following three major components:

- organic matter

- inorganic matter

- inorganic biogenic matter (diatome frustules and biogenic precipitated calcium carbonate)

4.1   Major and minor constituents

The main groups of elements in lake sediments are summarized in Table 1. Major elements (Si, Al, K, Na, Mg) make up the largest group (80 %) of the sediment matrix in most Nordic environments. Carbonate elements (Ca, Mg and CO_3-C), nutrient elements (org C, N, P) and mobile elements (Mn, Fe, S) make up approximately 20 % of the sediments. The smallest group, the trace elements (Hg, Cd, Pb, Zn, Cu, Cr, Ni etc) constitute less than 0.1 % of the sediments (Kemp et al, 1976).

4.2   Organic compounds

Organic matter represents all types of biogenic particulate matter in various stages of decomposition. The particulate matter form a substratum for bacteria, protozoans, diatoms and fungi. This living microcosmos forms the diet of the detritus feeders, while the non-living component is discharged as fecal pellets, only to undergo further decomposition through repopulation with bacteria. Organic matter which survives degradation in the water column is incorporated in the sediments

where it can be more or less preserved or sub-
jected to further biological degradation. High
IG values in sediments are favoured by the fol-
lowing three factors (Håkanson and Jansson, 1983):

-        anthropogenic discharges of organic ma-
         terial

-        high productivity

-        reduced conditions in the sediments which
         hamper degradation of organic material

The low molecular end products of the decomposi-
tion processes in anaerobic sediments such as
H_2S, CH_4 and NH_3 migrate upwards and, reaching
the aerobic top layer of the sediment, they are
oxidized by aerobic bacteria, mostly chemoauto-
trophes. The organic content is highly correlated
to the water content and hence is distributed in
sediments in much the same way (Figure 3). The re-
lationship between the bulk density (ρ), water
content (W), and organic content (IG^o) is viewed
in Figure 4.

4.3   Humic compounds

Humic matter is formed mainly as a result of the
degradation of plants and can be of allochthonous
as well as autochthonous origin. Humic compounds
constitute a dominating part of the total organic
carbon in many sediments and are most important
for the metabolism and turnover of many inorganic
elements, particularly metals and phosphorous. The
most important compounds used for the synthesis of
humus are lignin, carbohydrates, proteins and
phenolic compounds. Further degradation gives humic
acids, fulvic acids and a number of gaseous res-
piration products. Humic compounds are broken down

very slowly. Of the functional groups in humus,
carboxyl and phenol groups are of particular
interest. They react with positive ions, giving
humic compounds ion-exchange and metal-complexing
capacity. The ability to bind metals is essential,
since humic-rich sediments can thereby neutralize
the toxic effects of heavy metals. Humic materials
affect the pH of lake sediments and may serve as
a buffer.

5.   EXCHANGE BETWEEN WATER AND SEDIMENT

Part of the water within the sediment layer is
bound in chemical structures or is tightly adsorbed
to the sediment particles. The rest constitutes
the mobile medium which takes part in exchange
processes between the solid and dissolved phases.
The interstitial water serves as a carrier of el-
ements between the sediments and the overlaying
water and its existence assures continuity of the
aqueous phase over long sections of the sediment
layer. This will enable chemical species to migrate
within the sediment as well as across the sediment-
water interface.

The major factors governing the distribution of a
dissolved constituent in the interstitial water of
unconsolidated sediments are diffusion, vertical
advection due to sedimentation and compaction, and
ion exchange. Furthermore, bioturbation and near
bottom currents will induce changes in the pore
water composition. Of the several mechanisms re-
sponsible for the movement of materials across
the sediment-water interface, ordinary diffusion
is probably the least important. Mixing of the
sediment by the burrowing activity of deposit-
feeders is far more important and may result in
homogenization of the chemical composition of the
upper sediment layer. An important control on pore
water chemistry also occurs when infaunal organisms
pump overlaying water through their burrows in
order to replenish their oxygen or food supplies.
A "biological pumping rate" of water across the
interface of $0.7 \pm 0.3 \ cm^3 \ cm^{-2} \ day^{-1}$ was calcu-
lated by Luedtke and Bender (1979) for the sedi-
ment of Narragansett Bay (USA).

6.   DYNAMICS AT THE BOTTOM BOUNDARY LAYER

The structure and composition of a given lake or shallow bay sediment is a function of the characteristics of the catchment area and the water column. However, the sediments laid down are also subjected to various short-term pertubations such as resuspension, bioturbation and gas convection (Figure 5). Thus, the uppermost layer of most sediments reflects events occurring near at hand, while the sediments from a defined layer yield information integrated over time. The resolution in time is generally governed by the sedimentation rate. However, the sedimentation within a basin may also vary considerably. This implies that a selected sediment layer may represent a different time-span even when sediment cores are collected in close proximity to each other. Consequently, most sediments only provide a long-term average of the trophic level of a basin.

6.1   Sediment transport

Physical disruption of the sediments may result in resuspension of sediment particles. Once in suspension, a load of sediment can be transported a considerable distance before settling to the bottom. A number of empirical experiments have been made to determine the critical flow-velocity needed to start a particle into motion. As one might expect, the critical fluid-flow increases with grain size. An exception to this rule is noted for cohesive clay bottoms. Because of their resistance to friction they need rather high velocities to erode them (Figure 6). On the other hand, biological activity can destabilize the sediment by the destruction of sediment cohesion through burrowing (Yingst and Rhoads, 1978). This will increase the roughness of the surface layer sufficiently to enable particles of sediment to be picked up by weak bottom currents not capable of

erosion on their own (e.g. Evans, 1982). MacIlvaine
and Ross (1979) have emphazised the importance of
biota in determining surface roughness and sediment
erodibility. Fecal pellets also seem to be more
easily eroded compared to the ambient sediment
(Rhoads and Young, 1970). However, benthic infaunal
organisms may also stabilize the sediment by pro-
ducing mucous coatings on the walls of their bur-
rows. Woodin (1976) found that dense tube mats
could cause an increased resistance to erosion.

6.2   Sedimentation rates

The sedimentation rates of three eutrophic lakes
in Sweden were calculated by Edberg (1980) using
^{137}Cs fallout data (Table 2). The deposition in
lake Ekoln was 2800-3400 g m^{-2} dry wt (Håkanson,
1976).

El-Daoushy and Johansson (1983) reported some
sedimentation data from Swedish woodland lakes.
These lakes were situated on bedrock which was
covered with thin layers of moraine, and had low
primary production because of lack of nutrients.
The sedimentation rates were accordingly low and
varied from 0.5 to 1.2 mm yr^{-1}. The accumulated
amounts m^{-2} of sediments, integrated over 25 year
intervals, and the totals which represent integrals
for 125 years, are presented in Table 3.

Generally, the deposition rate increases in a
linear fashion with increasing water depth. In
open water areas the maximal values are found in
the deepest parts of the basin. In small lakes
where the deposition pattern is fairly unaffected
by external factors such as wind stress and waves,
high correlation is found between the rate of de-
position and the water depth (Figure 7, top). In
large lakes, on the other hand, the deposition is
highly influenced by hydrographical factors and

the correlation is significantly less between de-
position rate and water depth (Figure 7, bottom).

6.3   Bioturbation

Bioturbation can be defined as the added effect on
the mobility of elements from bacterial activity,
microbiological gas convection, burrowing benthic
organisms and from fishes hunting prey items on or
within the sediment. Bioturbation plays a signifi-
cant role in modifying the sediment properties by

-        altering the distribution and the size of
         the sediment particles

-        changing the bulk characteristics of the
         sediment such as its permeability

-        varying the boundary properties of the
         near-bottom flow by altering the rough-
         ness of the sediment bed.

The depth range of bioturbation generally comprises
the upper 5 - 10 cm of the bottom substrate. Most
of the bioturbation that occurs near the sediment-
water interface is performed by deposit-feeding in-
vertebrates which rework the sediment they ingest.
Several experimental investigations concerning flux
rates of nutrients, heavy metals and radioactive
trace elements in sediments have given evidence of
the importance of bioactivity for the release of
elements to the water column. Holdren and Armstrong
(1980) investigated factors affecting the release
of phosphorous from intact sediment cores.
Bioturbation was found to have the greatest effect
on phosphorous release by changing the chemical
conditions of the pore water, while sediment sus-
pension removed phosphorous from the water column.
Boddington et al (1979) recorded a significant
density-dependent loss of mercury from sediments
holding tubificid worms. The loss occurred mainly
from the uppermost cm of the sediment. Renfro

(1973) performed experiments to determine the loss rate of ^{65}Zn from a marine silty sediment. 3 - 9 % of the absorbed zinc was desorbed within 30 days. The burrowing activity of polychaete worms caused losses 3 - 7 times higher through resuspension of fine particles and, possibly, due to loss of dissolved zinc. Rates of biogenic reworking by different benthic invertebrates are viewed in Table 4.

The effects of bioturbation on the mobility of elements decrease downwards in the sediment due to a reduced population density and an increased sediment compaction. However, to appraise the stratigraphic importance of bioturbation, we must also know the sedimentation rate. The impact of bioturbation is greater if the sedimentation rate is low, because deposits from more years are in-volved in the mixing process.

6.4   Formation of varved lake sediments

In Scandinavia and certain parts of North America, varved lake sediments seem to occur quite commonly. In northern Sweden, 14 lakes in which the entire sediment deposit is varved are at present known (Renberg, 1982). The longest sequence so far found comprises almost 9 000 varves. In consolidated sediments from northern Sweden, varve thicknesses of 0.1 - 1 mm have been recorded. In unconsolidated sediments the varve thicknesses is greater, usually between 1 - 4 mm, but occasionally up to 10 mm (Renberg, 1982). The basic conditions for the for-mation of varved lake sediments are the seasonal variations of the climate, which produce an annual rhythm in the sedimentation of both biotic and abiotic materials. However, varve formation is realised only under certain pre-conditions. The sediment should remain undisturbed once it has been deposited on the bottom and not become mixed

due to bioturbation, water movements, gas convection or other causes. Bioturbation is the most deleterious factor with regard to varve formation.

In Sweden, all lakes in which varved sediments are known are dimictric, i.e. they circulate twice a year, generally in spring and autumn. In thermally stratified lakes with a high productivity or a substantial influx of allochthonous organic matter anaerobic conditions arise in the hypolimnion during the summer and/or winter months. This leads to a seasonal depletion of the benthic fauna, and the probability for an undisturbed sediment growth will increase.

The quality of the varves is best in the deepest parts where the stagnant conditions will prevail longest. Mechanical disruption of the surface sediment due to water movement appears to be relatively slight. Conditions for varve formation are further improved when the lake bottom is flat which ensures minimal physical disturbance of the deposited sediment.

The seasonal sedimentary succession is well reflected in the annual layers. First, a layer of mineral grains is visible which has been deposited during the snow-melt period. Above follows a layer of predominantly biological material such as algal remains and pollen which is deposited during the summer period. Above this layer is a thin layer composed of organic material deposited beneath the ice-cover of the lake (Renberg, 1982).

6.4.1   Sampling of varved sediments

The surface sediment of a lake is often very loose and sampling is usually a difficult operation. However, the crust-freeze method yields undisturbed

samples and is the best method to use when samples
of the unconsolidated surface sediment are to be
obtained for varve-counting and analysis (e.g.
Saarnisto, 1979; Renberg, 1981).

6.4.2   Varve-counting and dating

The individual levels in a varved sediment can be
dated by making varve-counts, and accurate annual
deposition values can then be calculated. The error
of the dates given by the varve-counts normally
does not exceed ±2 % (Renberg and Segerström,
1981). An exact geochronological dating method for
events during the Holocene period is particularly
valuable, considering the problems which can be
encountered in connection with the radiometric
dating of sediment samples. Up to the present,
various radiometric methods have been used in
stratigraphic investigations. However, during the
past few years, the reliability of the dates ob-
tained primarily by the ^{14}C method has been more
often brought into question; dating using this
method sometimes makes the dates several hundred
years too high or too low (Renberg and Segerström,
1981).

Compared with normal homogeneous lake sediments,
annually laminated lake sediments have many advan-
tages. Samples of known age can be analysed and
e.g. pollutant influxes can be determined with a
high resolution in time. The absolute dates ob-
tained by varve-counting also help to estimate
other time-related events such as the land uplift.
A new shore-line displacement curve for southern
Västerbotten province has been obtained by dating
varve counts of sediment cores from lakes situated
at different altitudes above the present sea-level
(Renberg and Segerström, 1980).

7. ESTUARIES AND BAYS

An estuary is by definition a region containing a
volume of water of mixed origin derived partly
from a discharging river system and partly from
the adjacent sea, the region being partially en-
closed by a land mass (Barnes, 1974). A coastal
bay is a body of water semi-isolated from the
sea by a ridge etc. Fundamentally, estuaries and
bays differ in the width of their connections with
the sea and, thus, in the rate of fresh-water in-
flow. Bays tend towards comparatively small fresh-
water inputs. Their salinities are generally more
stable than those of estuaries.

The geomorphological character of estuaries has
a pronounced influence in determining the nature
of the processes operating within any given estu-
ary. Because one end of an estuary grades into
fresh water and the other is open to the sea, it
follows that a salinity gradient of some form will
exist along the estuary. The dominant salinity
regimes vary from a state of little mixing between
the fresh and salt water masses (salt-wedge estu-
aries) to more or less complete mixing due to
current-induced turbulence in areas with strong
tidal influence.

7.1 Deposition in estuaries

The mixing of fresh and salt waters has consider-
able effects on the deposition of particulate
matter in the estuary. By the time most rivers
reach the sea, they have already deposited the
coarser sediments in their upper reaches. Silt
particles, however, are transported in suspension
in the lower reaches of most rivers and will be
discharged into the adjacent estuaries. On contact
with a medium containing high concentrations of

cations, these silt particles tend to flocculate,
so that at the zone where fresh water meets water
of high ionic strength, the silt particles clump
together and sink more rapidly. Observations on
different flocculation of suspended particles of
colloidal or semicolloidal dimensions are restric-
ted to clay minerals (Whitehouse et al, 1960). The
most common clay mineral types are kaolinite,
illite, and montmorillonite. Less common clay types
are vermiculite and chlorite; the latter mineral is
frequently found in marine sediments. It has been
found that clay mineral particles usually have a
negative charge, which may be explained by prefer-
ential adsorption of anions, especially hydroxyl
ions, and cationic substitutions within the crystal
lattice (Postma, 1967). Because of structural dif-
ferences, different types of clay minerals floc-
culate in different manners. Experiments have shown
that flocculation of kaolinites and illites occurs
in salinities of less than 4 o/oo, whereas the
flocculation of montmorillonite increases gradually
with increasing salinity and thus continues over
the whole estuarine salinity range (Barnes, 1974).
In rapidly moving water the various clay minerals
may be transported without differentiating, but at
low current velocities illites and kaolinites may
be deposited, whereas montmorillonites remain in
suspension. The difference may be greater in fresh
and brackish water than in ocean water.

In most semi-enclosed water bodies, net deposition
exceeds erosion so that there is an overall ac-
cumulation of mud. In those estuaries in which sea
water is the dominant water mass, the bulk of the
deposited material is of marine origin. Under the
comparatively sheltered conditions prevailing close
inshore, the mud may settle out. This process is

greatly accelerated by the presence of salt-marsh vegetaton. Thus, through the dual action of shelter and flocculation, estuaries and lagoons tend to be regions in which fine sediments are trapped.

A discussion of coastal features must inevitably contain a consideration of the recent history of coastal development. According to prevailing opinion, a rapid rise in sea level from about 100 m below the present level started around 17 000 years A D and reached a level 3 - 5 m below the present about 6 000 years A D. Thereafter the sea rose slowly to its present level which was reached about 2 000 years ago (Gorsline, 1967). Gorsline also suggested that the present coastal configurations have been moulded predominantly by processes operating only over the past 6 000 years and in many cases less than 3 000 years.

There are several estimates of sedimentation rates for various marginal marine environments. In humid regions with major river sources, the rate of accumulation can be very high, such as in the delta of the Mississippi, where a maximum value of 300 m/1 000 yr was estimated by Shepard (1960). In bays and estuaries an accumulation of 2 - 4 m/1 000 yr may occur (Shepard, 1953). In arid regions, the rate of accumulation may be very low because sediment can be supplied only by winds and the sea. Rusnak (1960) reported an accumulation rate of only 1 m/1 000 yr. However, neglecting the extremely high sedimentation rates represented by the deltas, the rate of accumulation for the marginal marine environment is about 2 m/1 000 yr for humid bays and estuaries, and 1 m/1 000 yr for arid and semi-arid lagoons. These rates apply only to the accumulation which has occurred since the post-glacial rise of sea-level. An upper limit of sediment accumulation

based on the overall time-average rise in sea level
would have to be not greater than 6 m/1 000 yr over
the span of the last 15 000 years (Rusnak, 1967).

In terms of the earth's geological history, indi-
vidual estuaries are relatively transient phenom-
ena. Assuming a constant sea-level in the future,
many existing estuaries will be filled by deposits
and be extinct. As sediment accumulates, it is in-
vaded by salt-marsh vegetation and will become
stabilized. The deposition rate is accelerated and
a bar may be formed across the mouth of the estu-
ary, which can exclude the inflow of sea water. The
resulting enclosed water body will rapidly develop
into a swamp or into marshland and finally into the
climax vegetation typical of its geographical re-
gion. Conversely, erosion may widen the mouth of an
estuary, so that it will gradually become a shallow
coastal bay. If the barrier separating a water body
from the influx of sea water becomes complete, con-
tact with the sea is usually only via seepage
through the barrier and such bays quickly become
fresh. All semi-enclosed water bodies are ephemeral,
either reverting to the sea on destruction of the
barrier, or evolving through marsh stages into land.
The silt and detritus frequent in estuaries and
shallow bays make such areas extremely fertile and
the land so formed may be used for agricultural
purposes.

7.2   Sedimentation rates in the Baltic Sea

With the retreating continental ice sheet, the sea
floor of the Baltic became covered by erosional
material which formed a more or less uniform layer
of stratified sediments. Further, a substantial
part of the sea bottom consists of postglacial
clays and muds covering the older sediments. The

term postglacial is used to denote the sediments
occurring above the Litorina/Ancylus boundary
(Winterhalter, 1972). The mean thickness of the
postglacial sediments in the Bothnian Sea is
about 1.9 m, which gives an annual sedimentation
rate of 0.3 mm (Winterhalter, 1972). In the cen-
tral part of the Gotland Deep, the maximum total
thickness of the Litorina and post-Litorina sedi-
ments amounts to 7 m. Considering a duration of
7 000 years from the Litorina era to present-day
conditions, this would imply a yearly sedimen-
tation rate of 1 mm (Ignatius et al, 1971). By
means of ^{210}Pb measurements, an annual sedimen-
tation rate of 1.0 - 1.3 mm was calculated. This
suggests that the deposition rate has been very
uniform for several thousand years. The results
from the Baltic Proper show that the deposition
rate hardly exceeds 1 - 2 mm yr^{-1}. This is in
good agreement with the values 0.1 - 2.0 mm yr^{-1}
given by Ignatius (1958) for the average sediment-
tation rate in the Baltic. The sedimentation rate
in the West Bornholm Basin varies in its different
parts; an average value of 0.5 - 1.5 mm yr^{-1} is
given by Kögler and Larsen (1979).

The amount of suspended matter in the southern
Baltic is estimated to be 432 ± 72 mg m^{-3} dry wt
during winter and 993 ± 382 mg m^{-3} during summer
(Lenz, 1974). 80 % of the particulate matter
ranges between 1 - 150 µm. Organic detritus con-
tributes to the main fraction, followed by phyto-
and zooplankton. In the southern part of the
Baltic about 500 g m^{-2} dry wt is deposited annu-
ally, corresponding to a sedimentation rate of
2.5 mm yr^{-1} (Zeitschel, 1965).

8.   MODELLING OF SEDIMENTARY PROCESSES

Efforts have been made earlier to model the physical parameters of recent sedimentary deposits (e.g. Håkanson, 1977), the effects of bioturbation (e.g. Håkanson and Källström, 1978), and the migration of trace elements in lake sediments (e.g. Lerman and Lietzke, 1975; Booth, 1975; Vanderploeg et al, 1975).

Before developing a model, the following points must be carefully considered:

- A precise definition of the objectives of the model.

- Identification of critical processes (e.g. hydraulic characteristics, sediment characteristics) that must be included in the model.

- Suitable approximations that must be made according to the relative importance of the different processes.

- Field monitoring and experimental work in the laboratory designed to give relevant measurements of the model parameters.

- Sensitivity analysis of critical steps in the model.

The objectives of the model must be to gain a better understanding of the different processes occurring in the natural systems and to provide a tool for decision making. The field and laboratory work must be planned and executed bearing in mind that all relevant physical, chemical and biological processes must be identified. The sensitivity of the model must be examined. The model result will be more sensitive to the value of some parameters than others. These parameters must be more accurately defined, presumably by a more intensive sampling effort or laboratory work.

Generally, the aim of a sedimentary model should
be to describe how physical and biological par-
ameters interact in recent deposits in areas where
material is being deposited. The model should also
enable the calculation of upward and downward
transportation of material and determination of
the vertical distribution of elements deposited
during defined periods of time. Critical data in
the model are recipient data (morphometrical shape,
residence time of the water), sediment data (sedi-
mentation rate, chemical and physical sediment
parameters such as porosity, K_d etc) and biologi-
cal data (uptake in benthic invertebrates, biotur-
bation rate etc).

9.   CONCLUSIONS AND RESEARCH PROPOSALS

Theoretical models of ecological processes in estu-
arine and lacustrine environments call for data
suited to the calculation of sediment budgets in
relation to space and time. Accurate long-term
forecasts about sediment growth and the subsequent
accumulation of elements in the deposits laid down
require a knowledge of production rates and sources
of particulate matter, sedimentation rates, affin-
ity of elements to solids, mobility of elements
when deposited on the sediment bed etc. Efforts to
model sediment dynamics must be based on parameters
derived from field measurements together with data
extrapolated from laboratory experiments.

The following parameters should be given priority:

-       Determination of the sediment growth.

-       Determination of basic sediment par-
        ameters.

-       Determination of the forcing functions
        of radionuclide migration within the
        sediment bed.

The parameter of main interest is the sediment
growth. However, the sedimentation rate is far
from uniform, either in space or in time. Conse-
quently, the sediments from a defined layer only
provide information that is integrated over longer
periods. The resolution in time is highly governed
by the sedimentation rate which varies from about
50 g $m^{-2}yr^{-1}$ in oligotrophic waters to about
3 000 g $m^{-2}yr^{-1}$ in eutrophic waters, Thus, the
measurement of the seasonal sedimentation rate
is of crucial importance for the interpretation
of sediment growth within a water body. The sedi-
ment growth in Sibbofjärden and Trobbofjärden

should be determined using sediment traps and the variations in the sedimentation rate both within and between years should be established during a two-year period.

The interstitial water within the sediment layer serves as a carrier of elements between the sediments and the overlaying water. This will enable chemical species to migrate within the sediment as well as across the sediment-water interface. In order to classify the physical environment in which the radionuclides are resident, it will generally be enough to determine a limited number of sediment parameters: water content, organic content, bulk density and grain size. Most other physical parameters such as porosity, permeability and compaction may be defined in terms of these basic parameters.

The sediments are subjected to various short-term pertubations such as seasonal variations in the sedimentation rate, resuspension by near bottom currents and bioturbation due to the micro- and macrobiological activity occurring within the deposits. These physical and biological factors will to a large extent govern the chemical status of the sediment bed. The mobility of radionuclides within the sediment will vary accordingly. In order to identify critical processes, experimental work should be performed in the laboratory and in the field. Radionuclides should be implanted in sediment cores and the propagation of elements within the sediment then correlated with environmental and sedimentary characteristics. In order to obtain "real world" conditions, a set of cores with implanted activity will also be restored to the Sibbofjärden and Trobbofjärden areas. At regular intervals of time, a number of cores will be

analysed with respect to physical and chemical characteristics and radionuclide contents. As long-term measurements are desirable, the implantation tests should also be performed over a two-year period.

The field and laboratory work should be planned and executed with the objective of developing a sedimentary model. This model may predict the transportation of radionuclides deposited in the sediments. In the laboratory, the effects of critical parameters on radionuclide mobility will be studied. A mathematical model of a laboratory experiment has two aims: to reveal the interactions and causal relationships between its variables, and to make predictions about the behaviour of the system under study. Since it is not possible to model all the complex interactions, the first stage in building a model is to formulate a simplified expression of the system. The next steps are to estimate and select the parameters of this simplified expression. The final step is to compare the data generated by the mathematical model with the experimental data obtained.

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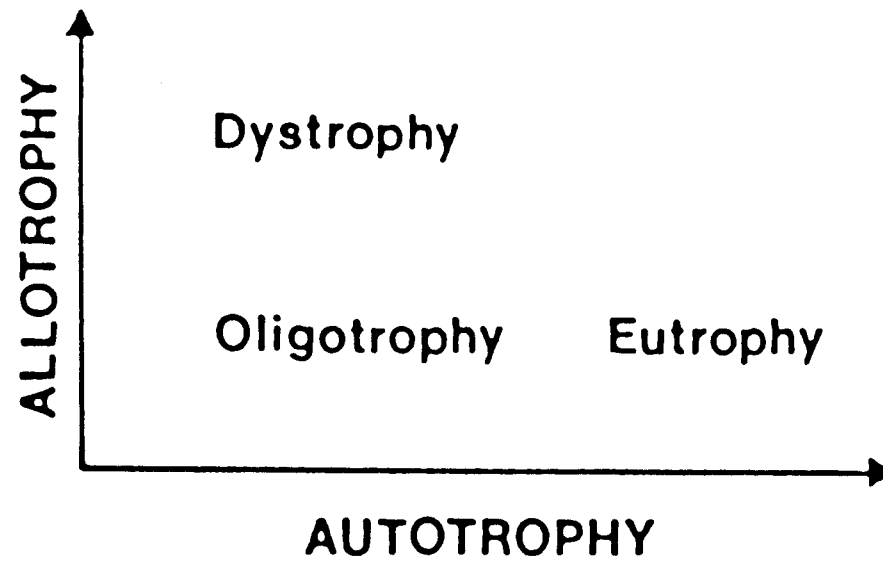und mittleren Ostsee. Kieler Meeresforsch
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Figure 1

Trophic types of lakes; from the relationship between the supply of organic matter from autotrophic (inside) and allotrophic (outside) sources (Wetzel 1975, Rodhe 1969)
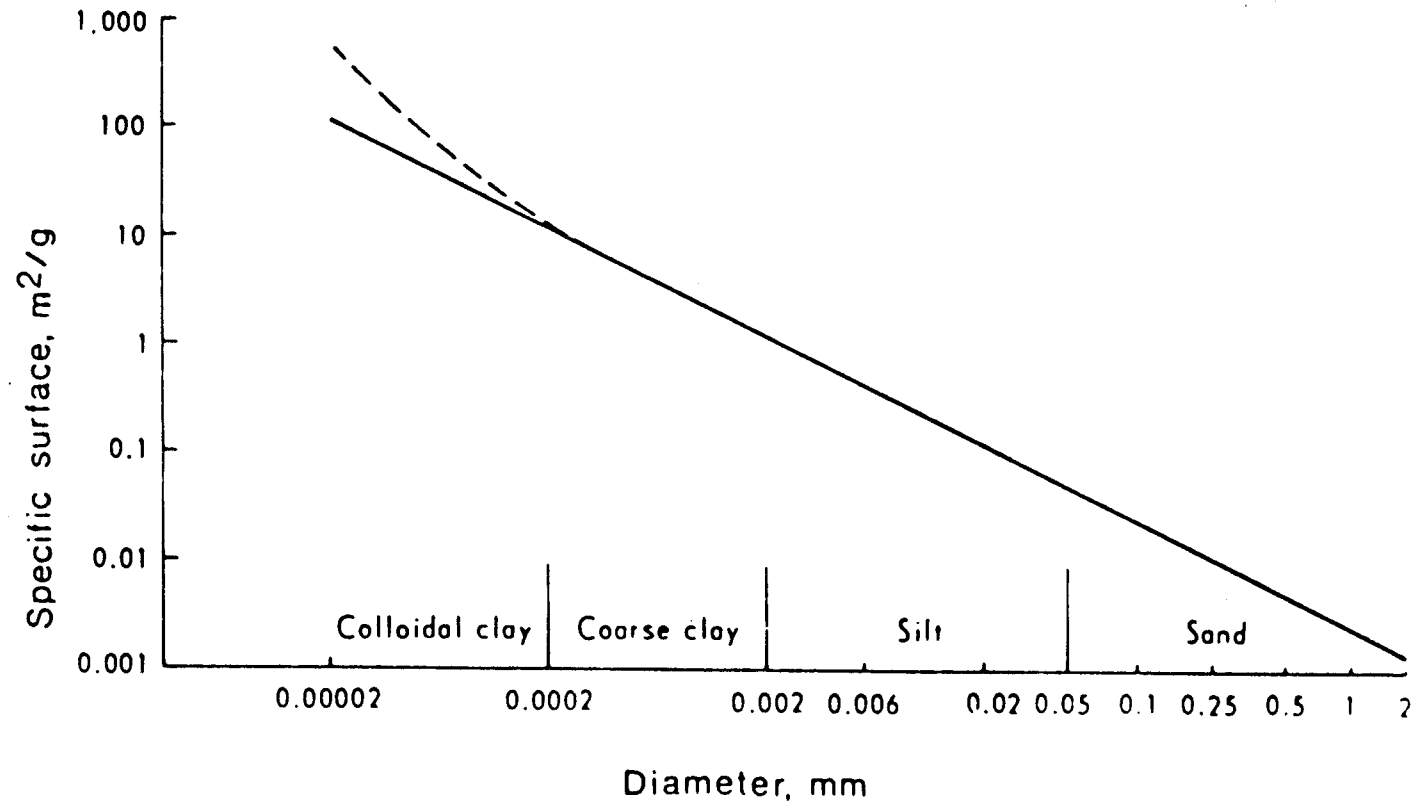
Figure 2

Relationship between particle size and approximate specific surface. Full line is calculated for spheres of density 2.65 g/cm^3. Broken line accounts for the fact that clay particles are more plate-shaped and that fine clays have a large internal surface (Kohnke 1968)
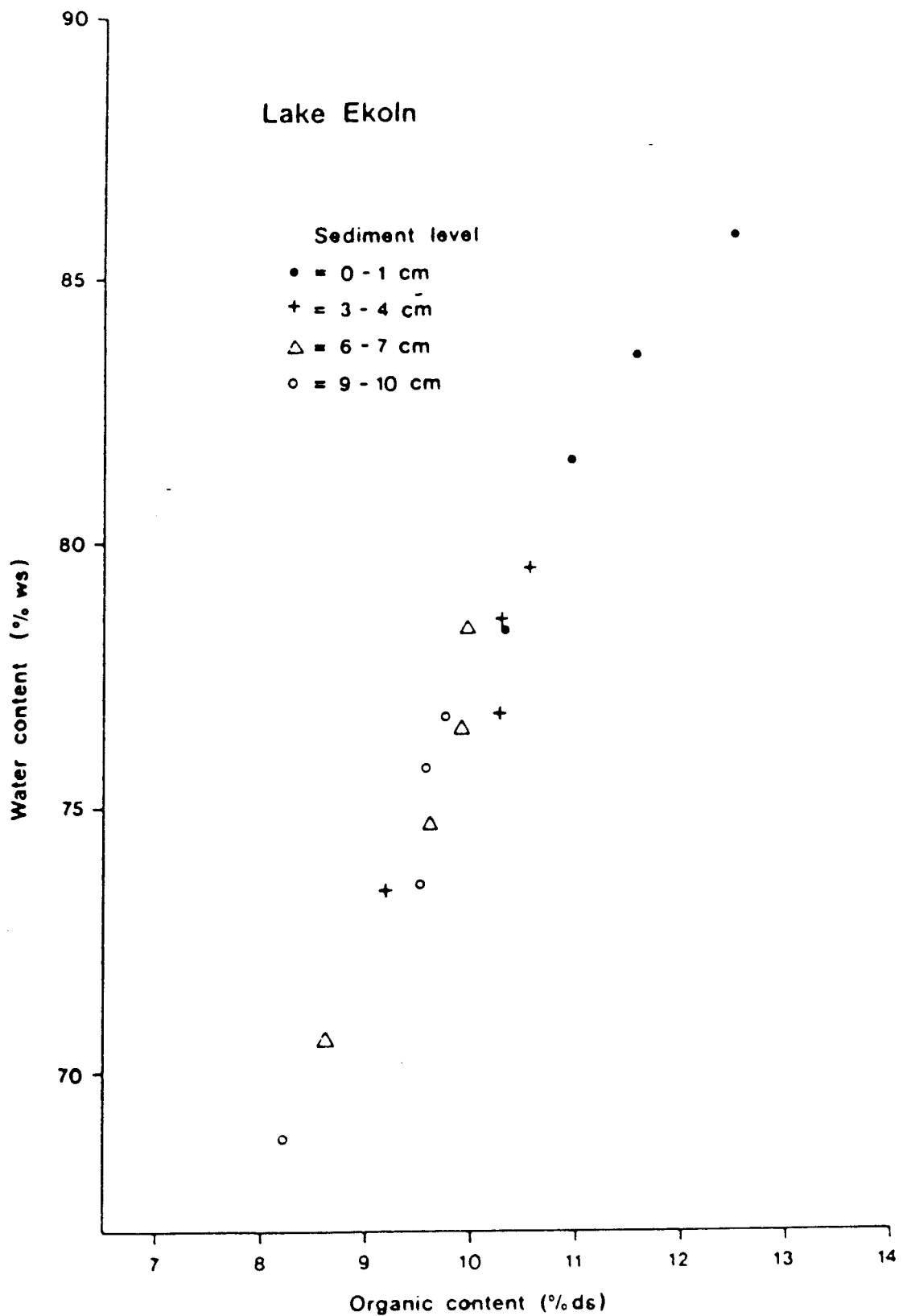
Figure 3

Relationship between water content and organic content (loss of ignition) at different sediment depths in Lake Ekoln (Håkanson 1977a)
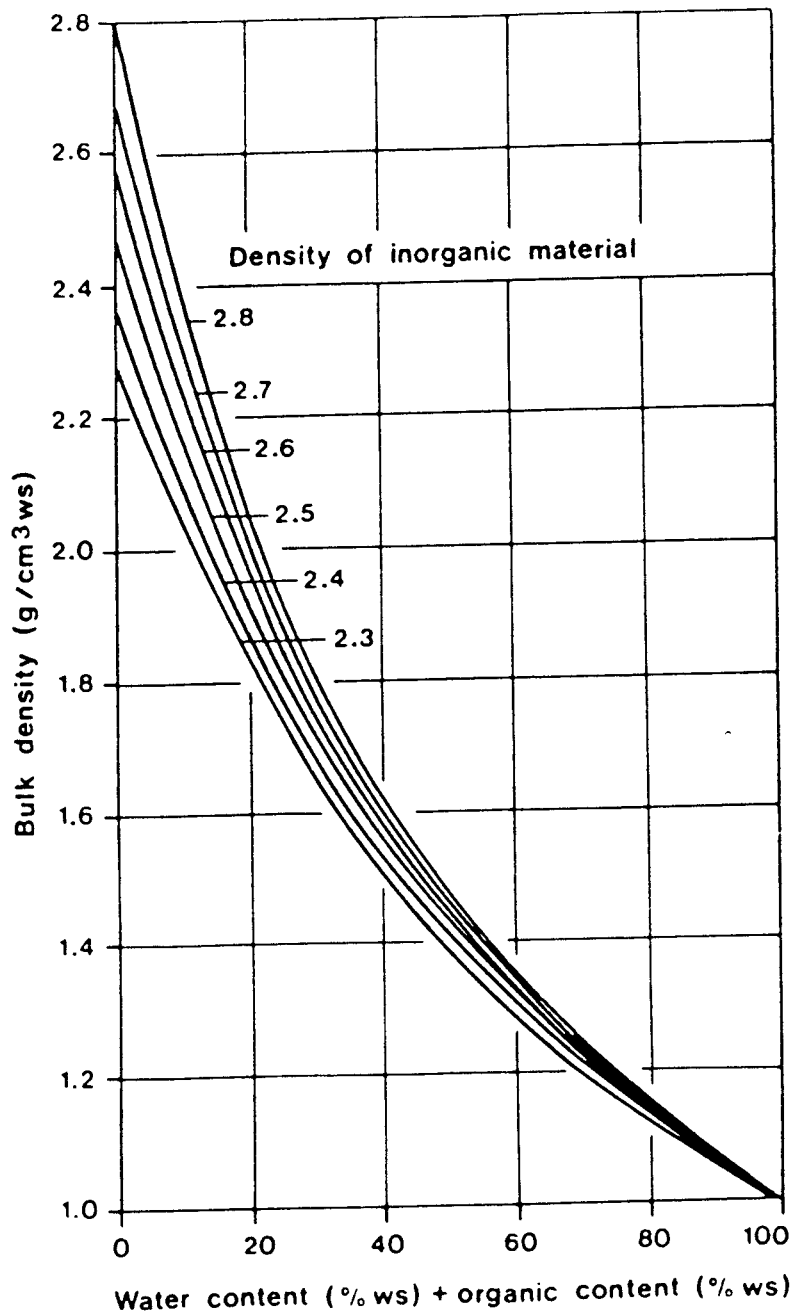
Figure 4

Nomogram illustrating the relationship between water content, organic content, bulk density and density of inorganic material (Axelsson and Håkanson 1971)
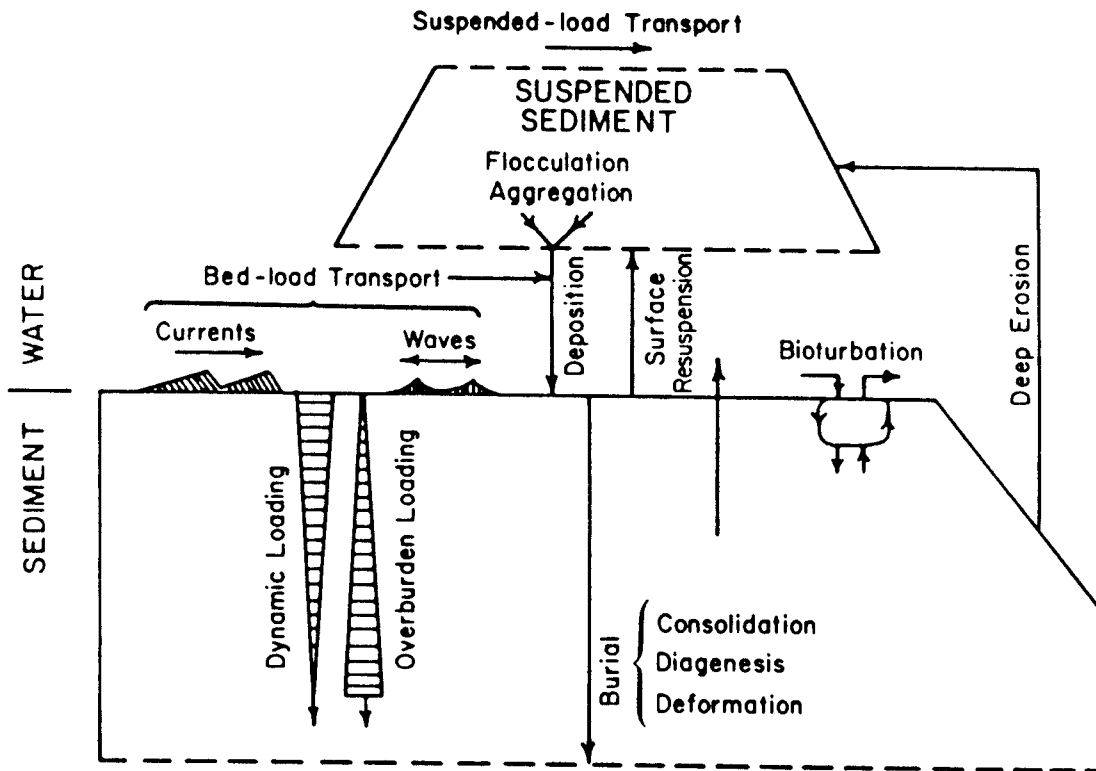
Figure 5

Benthic boundary sedimentary processes. From
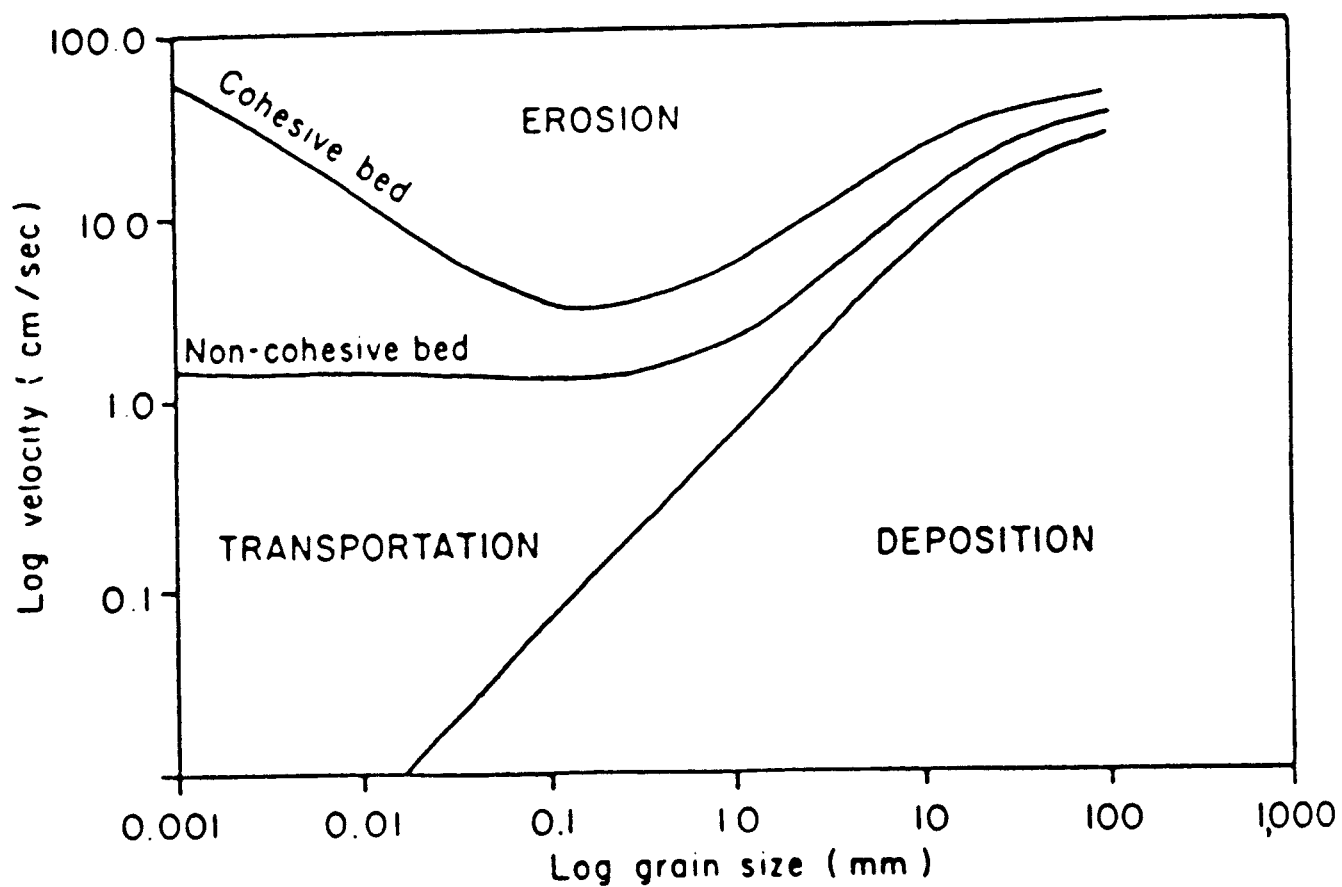Richards (1976)

Figure 6

Hjulstroms graph showing the critical velocities required to erode, transport and deposit sediments of varying grades. (After Sundborg, 1956.)
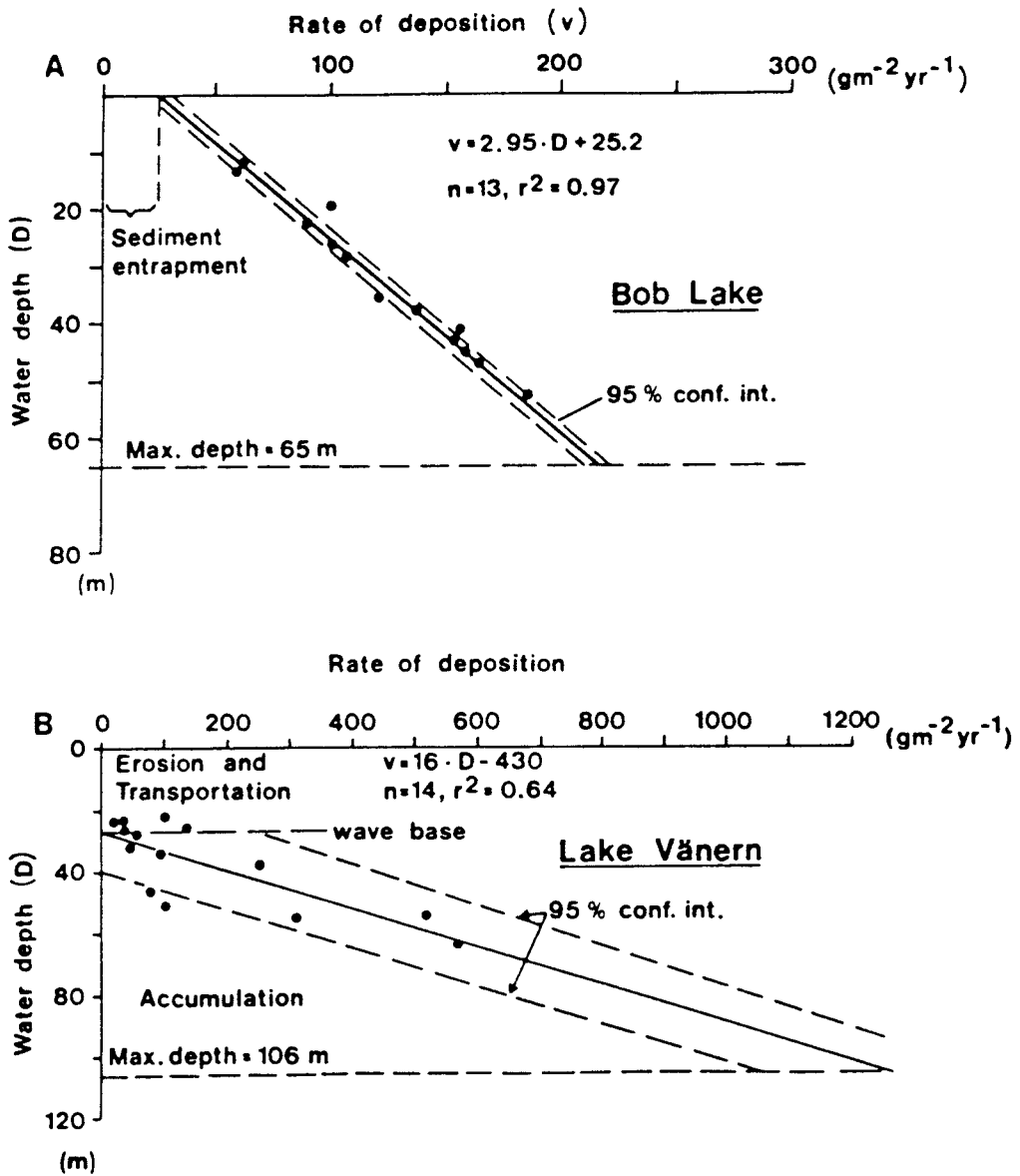
Rate of deposition (v)

A 0   100   200   300 $(gm^{-2}yr^{-1})$

$v = 2.95 \cdot D + 25.2$

$n = 13, \ r^2 = 0.97$

Sediment entrapment

Water depth (D)

20

40

60

Bob Lake

95 % conf. int.

Max. depth = 65 m

80

(m)

Rate of deposition

B 0   200   400   600   800   1000   1200 $(gm^{-2}yr^{-1})$

0

Erosion and Transportation

$v = 16 \cdot D - 430$

$n = 14, \ r^2 = 0.64$

wave base

Lake Vänern

Water depth (D)

40

95 % conf. int.

80

Accumulation

Max. depth = 106 m

120

(m)

Figure 7

The relationship between rate of deposition and water depth: A In a small lake, Bob Lake, Canada (redrawn from Evans and Rigler 1980). B In a large lake, Lake Vänern, Sweden (Håkanson 1975)

Table 1

Chemical classification of elements in lake sediments (Kemp et al 1976, Håkanson 1977d)

1.	Major elements (Si, Al, K, Na and Mg), make up the largest group of the sediment matrix.
2.	Carbonate elements (Ca, Mg and CO_3-C), constitute the second largest group in sediments, about 15 % of the materials by weight.
3.	Nutrient elements (org-C, N and P), account for approximately 10 % in recent lake deposits.
4.	Mobile elements (Mn, Fe and S), make up about 5 % of the total sediment weight.
5.	Trace elements (Hg, Cd, Pb, Cu, Zn, Ni, Cr, Ag, V, etc), the smallest group, accounting for less than 0.1 % of the sediments.

Table 2

Sediment growth in three eutrophic Swedish lakes
between 1963 and 1975 calculated from Edberg (1980)

| Lake | Depth (m) | Sediment growth (mm yr^{-1}) | |
		deep parts	shallow parts
Norrviken	~ 9	8 - 10	3 - 4
Ramsjön	1.5 - 2	8 - 9	3 - 5
Ekoln	~ 17	8 - 11	

Table 3

Accumulated amounts of sediment (kg m^{-2}), integrated over 25-year intervals, and the total amount integrated over 125 years. L O I % refers to the loss of ignition in per cent as averaged over 125 years. Asterisk = approximate value. From El-Daoushy and Johansson (1983).

Period, years	Lake Skärvsjön		Lake Björken	Lake Väster- -Täckelsjön	Lake Tussjön
	S1	S2			
Sediment (kg·m^{-2})					
0– 25	1.00	0.85	1.40	2.55	0.65
25– 50	1.10	0.85	1.70	2.80	n.d.
50– 75	1.15	1.00	1.70	3.05*	n.d.
75–100	1.05	0.85	n.d.	3.15*	n.d.
100–125	1.05	0.75*	n.d.	n.d.	n.d.
0–125	5.30	4.30	8.15*	15.15*	$\geqq$ 4.55*
L.O.I.% average (0–150)	30	30	25	30	40

Table 4

Sediment reworking capacities of marine and freshwater benthic invertebrates. From Evans (1982).

Species	Biotope	Locations	Reworking rate	Ref
Polychaete worms				
Heteromastus filiformis	marine intertidal	mud	$100 \text{ ml m}^{-2} \text{ day}^{-1}$ $= 4 \text{ cm sed yr}^{-1}$	Cadeé (1979)
Pectinaria californiensis	marine subtidal	mud	$1 \text{ kg m}^{-2} \text{ month}^{-1}$ (dry wt)	Nichols (1974)
Clymenella torquata	marine subtidal	mud	$274 \text{ ml ind}^{-1} \text{ yr}^{-1}$	Rhoads (1963)
Clymenella torquata	marine subtidal	mud	$96\text{-}246 \text{ ml ind}^{-1} \text{ yr}^{-1}$	Magnum (1964)
Pectinaria gouldii	marine subtidal	mud	$400 \text{ ml ind}^{-1} \text{ yr}^{-1}$	Gordon (1966)
Oligochaete worms				
Limnodrilus sp	freshwater profundal	mud	$2 \text{ cm sed ind}^{-1} \text{ yr}^{-1}$	Davies et al (1974)
Bivalves				
Yoldia limatula	marine subtidal	mud	$250\text{-}400 \text{ ml ind}^{-1} \text{ yr}^{-1}$	Rhoads (1963)
Nucula annulata	marine subtidal	mud	$365 \text{ ml ind}^{-1} \text{ yr}^{-1}$	Young (1968)
Crustaceans				
Callianassa	marine subtidal	mud	75 cm thick layer transported to the surface $\text{ind}^{-1} \text{ yr}^{-1}$	Warme (1965)

Karin Andersson

CHEMICAL PROPERTIES OF LAKE SEDIMENTS AND PEAT
OF IMPORTANCE FOR TRACE METAL BEHAVIOUR

ABSTRACT
A trace metal released into a lake, river or sea
recipient may be expected to sooner or later
reach a sink in the sediments. Evolution of the
recipient may however eventually turn the former
sediments into farming land or a peat bog, where
a release of the metal may occur.

The chemical environment may change considerably
during this time and a review of the present
knowledge of interstitial water and solid phase
chemistry in sediments and peat is given in this
report.

Some important parameters are:
pH - may vary from 3.5 for some peatlands to
over 8 in other peat types and in sediments.

Redox potential - both anoxic and oxic
environments may be found.

Ionic strength - depends both on water supply
(rain water to Baltic sea water) and reactions
in sediment or peat (usually higher ionic
strength than in surrounding water phase).

Complex formers - high contents especially of
organic complex formers (humic substances).

CONTENTS

1   INTRODUCTION

A trace metal which is released into a recipient (lake, river or sea) will be transported through different pathways until it reaches a more or less final sink in the sediments. Evolution of the recipient may however eventually turn the lake or bay into a agricultural land or a peat bog.

The chemical environment for the metal may be expected to undergo a series of changes during the evolution, and may thereby turn the sink into a source, i.e. release the fixed metal.

The time span for this kind of change will be of the order of thousands of years, and predictions of the long term fate of metals during the evolution have to be based on our present knowledge of systems existing in different stages of evolution.

Here, a review is given of the present knowledge of the chemistry of the interstitial water and of the solid phase in sediments and peat. Complexation and sorption processes for actinides and fission products as well as for some heavy metals have also been included.

One way in which analogous phenomena in nature may be studied is through the enrichment of metals, e.g. uranium, thorium and lead, in peat bogs. Some examples of sites where this has occured are given.

Since the aim of the present study is to provide information in order to be able to predict the transfer of radionuclides released into the biosphere from deep rock repositories of high

level nuclear waste, the main emphasis has been
focused on the behaviour of actinides and
fission products. Other trace metals, e.g. heavy
metals, have however been studied in sediments
and peat for a long time. These may show many
similarities, especially to fission products but
also to actinides, os regards their chemical
properties; for this reason heavy metal data
have been included to some extent in the present
review.

2   THE CHEMISTRY OF SEDIMENTS

The chemical conditions within a sediment are
determined by complicated interrelations of
reactions occurring within the different phases
in the sediment as well as between these. Not
only equilibrium conditions have to be
considered, since a decomposition of organic
matter is continuously going on in the
sediments. In additions the inorganic reactions
may be governed by slow reaction kinetics.

Some parameters of importance to trace metal
behaviour in the sediments have been studied by
various investigators - pH, Eh, ionic strength,
content of complex formers as well as properties
of the sorbing solids are among the most important.
The sediments consist of organic and inorganic
solids and an interstitial water phase which has
a composition different from that of the
overlaying lake water. The concentration
gradient between sediment and lake water may
cause a diffusion of solutes, usually from
sediment to lake water.

2.1   Organic compounds in sediments

The organic compounds constitute a very important
component both in sediments and in overlaying
water columns, influencing factors such as
oxygen content, carbon dioxide content and redox
potentials. This in turn controls solubilities
e.g. of phosphorus, iron and manganese. The
redox conditions are also of fundamental impor-
tance to the biological activities.

The preservation and interconversion of organic
matter is not only a biological activity but
also a function of morphology, trophic state,
stratification and oxygen content. Seasonal

density stratifications may lead to a depletion
of hypolimnetic oxygen, affecting the decomposers
in the water column as well as the bentic
animals (BAR 78).

The concentration of organic carbon is generally
low in the water column (10^{-3} to 10^{-2} g/l),
while in the sediments concentrations in the
order of 10 g/l are encountered. The highest
organic concentration is found at the sediment
surface, with a decrease deeper down due to
decomposition (AVN 84).

The organic compounds which are of major importance
to the behaviour of trace metals in soils, peat
bogs, lake and groundwater as well as in sediments
are the complex formers defined as humic substances.
These constitute a heterogeneous group of
substances with some common properties. The
complex forming capacity is due to the presence
of carboxylic, hydroxy and phenolic groups. Four
different kinds of humic substances are defined
by the solubility properties: humic acid is
soluble in alkali but not in acid, fulvic acid
is soluble in both alkali and acid, hymatomelanic
acid is the fraction of humic acid which is
soluble in alcohol and humin is insoluble in
alkali (OLO 83).

The properties of humic acid vary according to
its origin. In general the carbon content of
lake and marine humic acids is lower and their
H/C and N/C ratios are higher than those of soil
humic acid. The molecular weights may vary for
all types from 700 up to over 200 000. Components
weighing over 100 000 are however more common in
lacustrine and marine humic acids than in those
from soil (ISH 72).

The molecular weight distribution is different
for organics in interstitial water, and humic
and fulvic acid. In Table 1 the various fractions
found in Great Lakes sediments are given (KEM 74:2).
It is possible that the water soluble organic
matter in the interstitial water is the breakdown
product from the larger molecular weight compounds
which undergo transformations. The complexing
properties of humic compounds are treated in
more detail in chapter 4.

Table 1. Molecular weight distribution of organic
compounds in various fractions of Great
Lakes sediments acc. to Kemp (KEM 74:2)

Fraction	Molecular weight		
	<700	5000-10000	>200 000
Interstitial water	70-80%	-	-
Fulvic acid extract	20-23%	23-35%	18-29%
Humic acid extract	-	27-48%	25-51%

2.2   Nutrients

Among the nutrients phosphorus is an important
element, participating in many reactions within
the sediments, including complex formation with
trace metals.

The total amount of phosphorus in lake sediments
varies between 0.6 and 5 mg/g dry weight and
there does not seem to be any close relationship
between the trophic level of the lake and the
phosphorus content. The greater part of the
phosphorus in a lake ecosystem is bound to
sediment particles and sorption may occur on
iron(III)- and aluminum(III)-hydroxides, clays,
calcite and humic substances containing metal
ions.

In an oligotrophic lake most of the phosphorus
input is deposited in the sediments, but with
an increasing trophic level the interactions
become more complex. During certain periods the
release of phosphorus may exceed the deposition
and outflow will occur both to anaerobic and
well aerated water. A dissolution of iron and
phosphorus is observed when iron(III) is reduced
at potentials lower than 200 mV. There are
however indications that iron bound to humic
compounds is less sensitive to reduction.

A high pH may cause the release of phosphorus
bound to iron and aluminum, while in calcareous
sediments the formation of solid hydroxy-apatite
increases with pH (BOS 82). There also seem to
be interrelations between the phosphorus release
and nitrate (BOS 82).

Table 2 and Figure 1 show some mechanisms and
parameters affecting the phosphorus.

The many forms in which phosphorus may exist in
sediments thus create a very complex situation
as regards deposition and release reactions.

Table 2. Effects of important environmental factors
          on phosphorus mobilization acc. to
          Boström (BOS 82)

Environmental parameter	Effect on phosphorus mobilization
Redox potential	Iron-bound phosphorus is released at potentials below 200 mV when iron (III) is reduced to iron (II).
pH	An increase in pH decreases the phosphorus-binding capacity of iron and aluminum compounds, primarily due to ligand exchange reactions where hydroxide ions replace phosphate. Calcite and apatite formation at higher pH-values increase the phosphorus-binding capacity of calcium.
Temperature	An increase in temperature gives primarily indirect effects due to increased bacterial activity, which increase oxygen consumption and decrease the redox potential. The production of phosphate-mobilizing enzymes and chelating agents might increase accordingly.
Equilibrium criteria	Affects adsorption-desorption and dissociation of precipitates.
Chelating agents	Replace phosphate from salts with calcium, iron and aluminum. Chelating agents can be produced by bacteria and algae or occur as a pollutant.

Figure 1. Important processes of phosphorus release
          from lake sediments acc. to Boström (BOS 82)
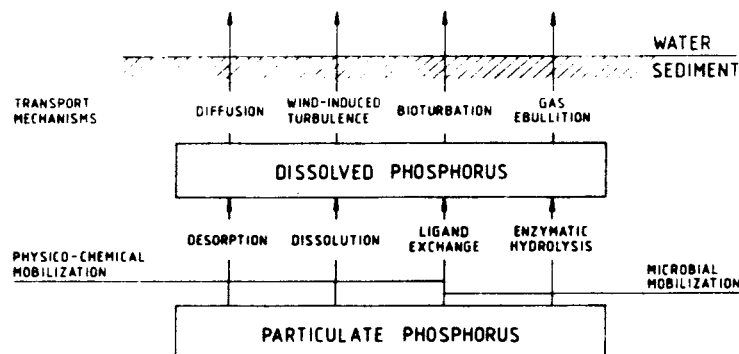
2.3 Trace metals

The total content of metal ions in sediments has
been studied by several investigators.

Sediment cores from Lake Como, Italy were sliced
and analysed for grain size, mineralogy and
chemical composition. The sampling points were
chosen so as to represent lithological differences
in the drainage areas: intrusive bodies, metamorphic
rocks, ophiolithes, limestones, dolomites etc.
Generally a surface enrichment of Cr, Cu, Pb and
Zn was encountered. These elements had higher
contents than natural (BUG 81). Elemental
analysis of sediments from Chautauqua lake, NY
by NAA (Neutron Activation Analysis) was performed
and the result compared with data for the
surrounding bedrock. Eu, Na, Mn, K, Br, As, Ga,
La, Hf, Cs, Tb, Sc, Fe, Ta and Sb were determined.
The lake sediments were found to be enriched in
Na, Mn, Br, As, Hf, Tb, Ta, and Sb. There was
also a relationship for many elements between
their concentrations and the clay fraction of
the sediment, possibly due to the higher cation
capacity of the clay minerals (HOP 76).

2.4 Mineralogy

The minerals in sediments may be of three
different origins:

- Allogenic, i.e. coming from outside the
  lake and supplied by rivers, shore
  erosion etc. Among these minerals,
  silicates (including clays) and carbonates
  are the most common.

- Endogenic, i.e. precipitates formed in
  the lake water. Among these are calcite
  and iron(III) compounds as well as diatoms

with sorbed species such as silica and
others.

- Authigenic, i.e. formed within the
    sediments due to changes in chemical
    and/or physical properties.

In Table 3 a list of minerals likely to be found
in lake sediments is given (HÅK 83). A minera-
logy that indicates allogenic sources of the
minerals has e.g. been observed in Lake Como,
Italy where the following minerals were identi-
fied in a 27 cm sediment core: quartz, muscovite,
illite, chlorite, plagioclase and dolomite
(BUG 81).

Important authigenic minerals are phosphates,
e.g. vivianite, reddingite and anapaite. These
are the stable and most probable phosphate
minerals in reducing environments, and are
considered to constitute the buffering mechanisms
for Great Lakes sediments, regulating both the
levels of phosphorus in the interstitial waters
and the release of phosphorus to the overlying
lake waters (NRI 74). In Greiffensee, Switzerland
it has however been shown that the phosphate
minerals formed are not the most stable thermo-
dynamically, and kinetics here seem to be of
great importance for mineral formation (EME 78).

Sulfides are also formed authigenically. This is
considered to be a process largely controlled by
biological processes (NRI 68). The sulfides are
responsible for the dark colour of the sediments
and treatment with dilute acid may remove the
dark colour and simultaneously release H_2S
(DOY 68).

Table 3. Minerals from lake sediments acc. to Håkansson (HÅK 83)

Mineral	Type of source		
	Allogenic	Endogenic	Authigenic
Non-clay silicates			
Quartz – SiO_2	X		
Potash feldspar – $KAlSi_3O_8$	X		
Plagioclase – (Na, Ca) (Al, Si) Si_2O_8	X		
Mica – K (Mg, Fe, Al)$_3$AlSi$_3$O$_{10}$(OH)$_2$	X		
Amphibole – (Ca, Mg, Fe, Al)$_{3.5}$ Si$_4$O$_{11}$(OH)	X		
Pyroxene – (Ca, Mg, Fe), Si$_2$O$_6$	X		
(Other heavy minerals, $\rho > 3.0$)	X		
Opaline silica (diatoms)		X	
Clays			
Illite – K$_{.8}$Mg$_{.35}$Al$_{2.26}$Si$_{3.43}$O$_{10}$ (OH)$_2$	X		
Smectite – X$_{.3}$Mg$_{.2}$Al$_{1.9}$Si$_{3.9}$O$_{10}$ (OH)$_2$	X		
Chlorite – Mg$_5$Al$_2$Si$_3$O$_{10}$ (OH)$_8$	X		
Kaolinite – Al$_2$Si$_2$O$_5$ (OH)$_4$	X		
Mixed clays, vermiculite – intermediate	X		?
Palygorskite – (Ca, Mg, Al)$_{2.5}$ Si$_4$O$_{10}$ (OH) · 4H$_2$O	X		?
Nontronite – X$_{.5}$Fe$_2$Al$_{.5}$Si$_{3.5}$O$_{10}$ (OH)$_2$	X		X
Carbonates			
Calcite – $CaCO_3$	X	X	x
Dolomite – $CaMg(CO_3)_2$	X		?
Aragonite – $CaCO_3$	x	X	
Mg-calcite – intermediate		x	X
Rhodochrosite – $MnCO_3$			X
Monohydrocalcite – $CaCO_3$ · H_2O		X	?
Siderite – $FeCO_3$	?		?
Fe-Mn oxides			
Goethite, Lepidocrocite – FeOOH	X	x	X
Magnetite – Fe_3O_4	X		
Hematite, maghemite – Fe$_2$O$_3$	X		?
Birnessite – (Na, Ca) Mn$_7$O$_{14}$ · 3H$_2$O	?		X
Todorokite – (Na, Ca, K, Ba, Mn)$_2$Mn$_5$O$_{12}$ · 3H$_2$O	?		X
Psilomelane – (Ba, K) (MnO$_2$)$_{2.5}$ · H$_2$O			X
Ilmenite – $FeTiO_3$	X		
Phosphates			
Apatite – Ca$_5$(PO$_4$)$_3$ (OH, F)	X		x
Vivianite – Fe$_3$(PO$_4$)$_2$ · 8H$_2$O			X
Ludlamite – (Fe, Mn, Mg)$_3$(PO$_4$)$_2$ · 4H$_2$O			X
(?) lipscombite – Fe$_3$(PO$_4$)$_2$ (OH)$_2$			X
(?) phosphoferrite – (Mn, Fe)$_3$(PO$_4$)$_2$ · 3H$_2$o			X
(?) anapaite – Ca$_3$Fe (PO$_4$)$_3$ · 4H$_2$O			X
Sulfides			
Mackinawite – FeS$_9$		x	X
Pyrite – FeS$_2$	X		x
Griegite – Fe$_3$S$_4$			X
Sphalerite – ZnS		x	
Fluoride			
Fluorite – CaF$_2$			X

Print size is varied to illustrate the frequency of occurence. X in formulas refers to monovalent cation exchange

2.5   Redox conditions

The redox potential is an important parameter,
determining the valence state and thus the
chemical properties of many trace metals. It is
closely related to the oxygen content, and
sediments formed under varying oxygen conditions
depending on the time of the year sometimes
show a laminated structure which can be seen
visually as alternating dark (sulfide) and light
layers and can also be measured with a platinum
electrode. The electrode potential decreases
with depth and a column of sediment may be gray
or black at depth and brown, yellow or orange at
the surface near the zone of oxygenated water
(DOY 68). A typical platinum electrode potential
gradient in a sediment core and the overlying
water column is shown in Figure 2 (MOR 71). The
pS^{2-} has also been determined, Figure 3. Both
the profiles show large gradients in the upper
20 cm and constant (or undetectable) values in
the lower portion of the core. The boundary
corresponds to the general appearance of the
sediment which changes from black to light gray
in this region (EME 76).

Sulfur is an important element in sediment
chemistry as the sulfur compounds are redox
sensitive. In lake sediments sulfide dissolved
in the interstitial water, acid volatile sulfide,
elemental sulfur, organic sulfur and sulfates
have been identified (NRI 68).

Figure 2. Platinum electrode potential across
sediment/water interface in "undisturbed"
cores from the deepest basin of Windermere
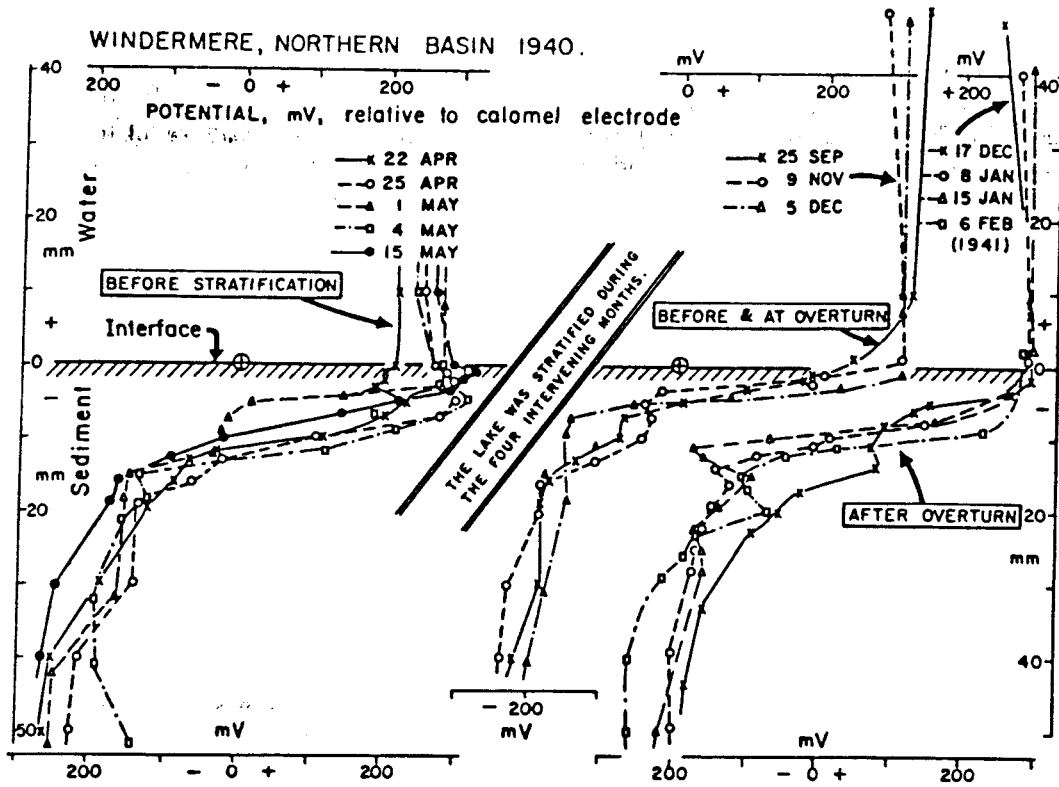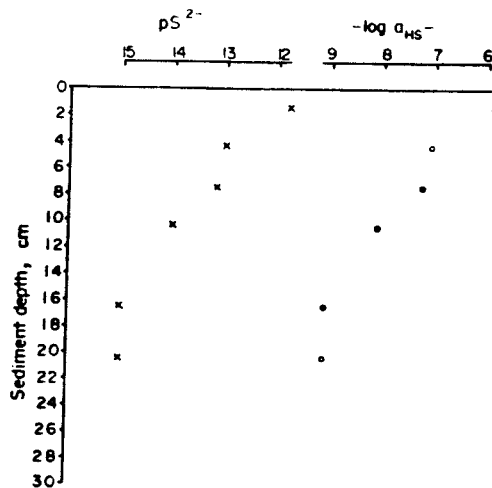during three seasons, acc. to Mortimer(MOR 71)

Figure 3. Sulfide activity as a function of depth in
a lake sediment core acc. to Emerson (EME 76)

2.6   Interstitial water composition

The composition of the interstitial water in
lake sediments has been investigated. Usually
the water has been extracted by pressing the
cores, but in some cases the water phase after
centrifuging has been used.

pH. Emerson et al studied the pH of sediment
cores from Greifensee by inserting glass
electrodes directly into the core. The pH was
found to be quite constant (= 7.05) with depth,
cf. Figure 4 (EME 76). The pH of sediments from
Seneca Lake, NY, was determined at different
depth. This is a lake of glacial origin. The pH
down to 20 cm depth varied very little (6.8-7.1)
and was equal to the value observed in the
overlying lake water. In some cases slight
decreases with depth were observed (HAR 67).

Na. The Na content in interstitial water in
sediment cores from Greifensee, Switzerland
showed a decrease down to 50 cm. This may partly
be explained by an increased salting of the
roads since the mid 60´s (EME 76).

In sediment cores from Lake Eire, the pore water
was analyzed, in one case down to 180 cm depth,
cf. Tables. An enrichment in Na compared to the
lake water was observed. In some cases a
gradient of Na (increase with depth) was
observed (NRI 74).

K. Likewise for K there was a slight increase
with depth in some Lake Eire sediment cores (NRI
74).

Ca. The content of calcium also increased with
depth in Greifensee sediments, but the large

increase in the first few centimeters was
followed by a moderate maximum at 6-10 cm. At
larger depths the profile was vertical (EME 76).

In Lake Eire sediments there was a general
increase with depth for Ca (NRI 74).

Figure 4. pH of the interstitial water of a lake
sediment core. X = results from inserting
the electrode directly into the sediment,
O = data from the supernate solution after
centrifuging, Δ = first reading for the
alkalinity Gran titration acc. to Emerson
(EME 76)
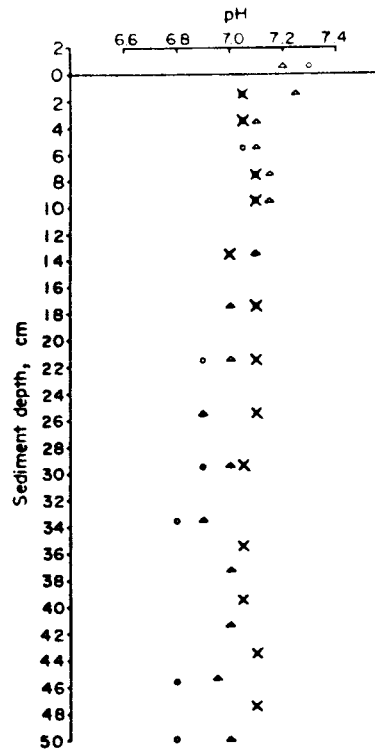
Mg. In Greifensee sediments no gradient of Mg
could be found (EME 76).

In Lake Eire, there was also a gradient of Mg.
This was not however the case for all the
sediment cores (NRI 74).

Mn. No gradient of Mn was found in Greifensee,
cf. Figure 5 (EME 76). The concentration of Mn
in Lake Eire was higly variable but tended to
increase with depth (NRI 74).

Fe. A very large gradient of Fe in the pore
water of Greifensee sediments was found, cf.
Figure 5 (EME 76).

In lake Eire, the Fe content was highly variable
but tended to increase with depth (NRI 74).

P. A large gradient of phosphorus was observed
in Greifensee sediemnts, c.f. Figure 5 (EME 76).

In lake Eire no apparent P-gradient was observed
(NRI 74).

N. No evident gradient in N was found in
Greifensee sediments (EME 76).

Si. In a study of the silica content in river,
lake and organic bog sediments, very high silica
concentrations were found in river and organic
bog sediments at 6 cm below the surface. This
was associated with a low pH. An attempt was
made to explain this in terms of decomposition
of organic matter (HAR 67).

Cl. Chloride was also enriched in river, bog and
lake sediments and the gradient was, as for
silicate, confined to the top layer of the
sediment. Below 8 cm there was a high and
constant concentration (HAR 67).

Figure 5. Dissolved P, Fe, and Mn in interstitial
water of lake sediment core acc. to
Emerson (EME 76)
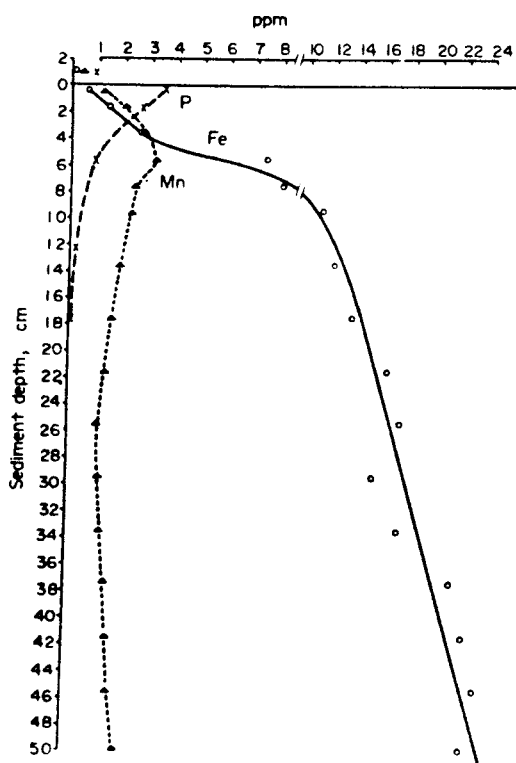
Table 4 gives a review of composition data for
some natural waters. The total amount of
dissolved species is of the same order of
magnitude in sediment interstitial water as in
shallow groundwater. The quoted sediment data
are however for freshwater lakes. The pore water
composition in Baltic sea or ex Baltic sea
sediments may be different, considering the much
higher ionic strength in Baltic sea water.

A concentration in the water phase as regards
Na, K, Mg, Ca, Mn, Fe and possibly N seems to
occur in lake sediments. A larger amount of
material and also data for interstitial water
from Swedish lakes would however be required to
draw further conclusions on the pore water
composition development.

Potential complex formers among the inorganic
ions are phosphates, nitrates, carbonates and
silicates. Silica, as well as Fe-, Al- and
Mg-hydroxides, may also act as a colloidal
sorbing agent. Unfortunately, the studies on
pore water composition encountered in the
literature do not include carbonates and
sulfates.

Table 4. Concentrations of major ions in some types of natural water. Data in mg/l.

	Rain water[d]	Shallow groundwater[d]	Intermediate groundwater[d]	Lake water[a]	Lake sediment[a,b] 0 – 6 cm	Lake sediment[a,b] > 6 cm	Sea water[d]	Baltic sea[d] water	Peatland[e] waters
Na	0.3-20	2-25	10-100	1.3-15	2.3-21	4.4-36	10560-10766	1620-4980	1.3 -40
K	0.1-4	0.5-5	1-5	0.5-1.35	1.2-4.6	1.5-4.8	380-399	54-180	0-5
Mg	0.1-0.5	0.5-10	5-20	2.7-10	3.7-16	4.9-18	1272-1292	186-600	0.1-43
Ca	0.5-5	5-50	25-50	13-61	26-109	30-109	380-399	70-190	1.2-110
Mn			0.1-0.5	0.002-0.7	0.4-7.9	0.9-8.9	400-413		0.01-0.6
Fe		0-0.5	1-20	0.005-0.1	0.3-7.4	1.2-22.8	<0.02		0-9
Cl^-	0.1-20	5-20	5-25	16-118	18-129	10-152	18980-19353	2860-8960	1-50
NO_3^- [c]	0.1-4	0.1-5	0.1-0.5	6.6	6.6-12.4	12.0-13.3	<0.7		0-18
HCO_3^-	<1	60-250	60-400				140	73-140	1-420
SO_4^{2-}	0.1-5	0.5-25	1-25				2650-2712	410-1260	2-180
PO_4^{3-} [c]			0.01-0.1	0.6-2.8	0.1-14.4	0.07-2.8	<0.1		
SiO_2	<1		5-30	0.46	1.5	2.4-2.6	0.1-7		
Σ	2-60	75-400	100-600	41-216	60-323	67-370	35000	5273-16310	6.6-875
pH	4-6	5-8.2	7.2-8.5	7.1-8.0	6.9-8.0	6.8-7.6	8.1	8.1	3.5-7.1

[a] Lake water and lake sediment data based on measurements in Seneca Lake, N.Y., Greifensee, Switzerland, and Lakes Eire, Ontario, Michigan, Huron and Superior, according to (HAR 67), (EME 76) and (NRI 74). [b] Sediment = interstitial water.

[c] Total N and P data recalculated to NO_3^- and PO_4^{3-}, respectively.

[d] Data for rain water, groundwater and sea water from (DYR 80), (LAU 83) and (GOF 77)

[e] Data for peatland waters from (CLY 83), (TOL 74), (LAR 76), (SCH 77), and (CHM 77)

2.7   Changes in chemistry with time

The changes in chemical properties of a lake sediment with time may be discussed using the distributions of elements observed in sediment cores.

In the interstitial water there is an increase in ionic strength with depth, a decrease in redox potential and very slight changes in pH.

Changes in total chemistry are mostly confined to the organic matter, which undergoes a continuous degradation due to biological activity as well as to changes in the chemical environment.

The degradation rate in sediments from 64 small and shallow lakes in the United States has been determined. For organic carbon the decay coefficient was of the same order of magnitude for all the lakes and around $4 \cdot 10^{-3} \, y^{-1}$. The same value was observed for organic nitrogen. For phosphorus, which in contrast to carbon and nitrogen does not form volatile decomposition products, the changes with depth were not as obvious, even though a large fraction of the phosphorous in the sediment is organic. A similarity of the decay rates of organic carbon, nitrogen and phosphorous in lakes located in different climatic and geographic regions was however observed (AVN 84).

The changes with time in the organic content of sediment cores from Lake Haruna, Japan, have been studied. Sediment cores down to a depth corresponding to 1000 years have been analysed. In this environment the decomposition seems to be low, although the amount of extractable humic substances, which include both humic and

fulvic acids, decreases slightly with depth. In
the humic acids there seems to be an increase in
carbon content, C/H ratio, C/N ratio and colour
intensity, and a decrease in proteinaceous
material and in carbohydrate like material.

Three possible pathways for long term transforma-
tion of sedimentary humic acids are suggested:
(1) sedimentary HA - soil HA - graphite, occuring
in oxidizing environments, (2) sedimentary HA -
coal - graphite, in pyrolytic conditions, (3)
sedimentary HA - kerogen (or petroleum) -
graphite, in reducing environments (ISH 72:2).

A comparison of sediment cores from Lake Washington
taken in deep water in different parts of the
lake in 1958/59 and 1968 and 70 was performed.
During this interval the sewage effluent that
had been causing eutrophication of the lake was
diverted and the lake began to regress to a
relatively unproductive state.

All the cores showed higher concentrations of
organic matter, phosphorous and nitrogen in the
top few centimeters than deeper down, but in
general the higher surface values of the earlier
material had decreased somewhat since they lay
below the surface in the more recent cores (SHA
71).

3 THE CHEMISTRY OF PEAT LANDS

The chemical conditions variy between fen and bog. The bog obtains its water mainly from precipitation and it is thus poor in minerals. Bog vegetation is typically dominated by Sphagnum moss and ericaceous shrubs. Soils in bogs are usually undecomposed fibric peat overlying partly decomposed humic peat. The water may be quite acidic. Fens or minerotrophic peatlands are nourished by regional groundwater and are therefore ion rich and only slightly acidic to slightly basic. Fen vegetation is dominated by sedges (Carex spp.) and tall shrubs, e.g. birch (Betula spp.) alder (Alnus spp.) and willow (Salix spp.).

Many factors affect the chemistry of peat. Amongst the most important are the nature of the original plant matter, the supply of inorganic solutes, the activities of plants and animals and of microorganisms, environmental conditions (particularly temperature and the extent of waterlogging) and finally the age and history of the peat. These factors are of course not independent and there are marked vertical gradients associated with these.

In general our knowledge of the organic composition of peat is fragmentary, although its origin indicates that it will contain the same components as the plants which form the peat and decomposition products of these (CLY 83).

One important property of the peat is the cation exchange capacity. The cation exchange capacity (CEC) of organic soils is commonly in the range of 1 - 2 meq/g when measured at pH 7. For lower pH, the CEC will be lower. Considerable indirect

information exists to show that covalent bonding
of divalent ions (Ca, Mg, Fe, Cu, Zn and Mn)
occurs in peats and in isolated humic, fulvic
and hymatomelanic acids. Monovalent ions are
sorbed less strongly than divalent, and Mn^{2+} is
more tightly held than Ca^{2+} (SIK 83)

For peats, the CEC may vary considerably and
over short distances both horizontally and
vertically. An example of a profile is given in
Figure 6. Values of CEC from 0.07 meq/g to
1.9 meq/g have been observed in peats (CLY 83).

Buffer curves of acid peats generally have
pK_a-values of about 6. This is weaker than
predicted from consideration of the carboxyl
group present, presumably a result of the
formation of covalent bonded hydroxy complexes
of iron and aluminum such that an increase of pH
involves hydrolysis of these complexes rather
than ionization of carboxyl groups (SIK 83).

Figure 6. Profile of cation exchange capacity of peat
from Stordalen, Abisko, northern Sweden.
Peat type Sphagnum fuscum, with some
Eriophorum at points marked e. Acc. to
Clymo (CLY 83)
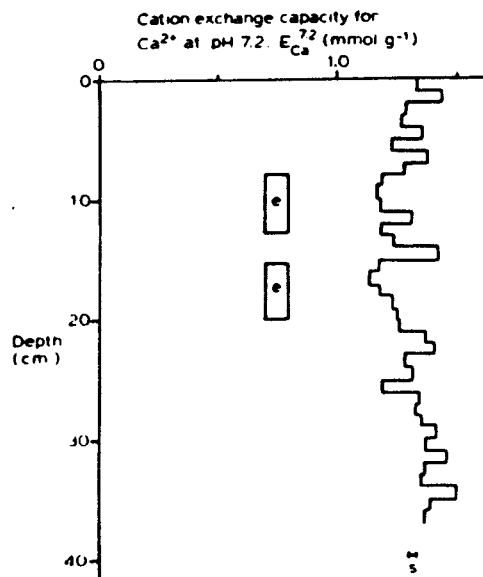
3.1   Redox conditions

The "redox potential", i.e. the platinum electrode potential, is one indicator of the state of peat There are difficulties in obtaining reproducible potentials and a "true" redox potential is probably never obtained. Within a certain set of peat cores the potential may however be a useful tool for determining the changes in redox conditions.

Three other features are commonly correlated with "redox potential" profiles: the position of the water table, the presence of sulfides and the oxygen concentration. There is usually a smell of hydrogen sulfide correlated to low redox potential and this is usually occurs just below the ground water table, where the redox potential may decrease rapidly, cf. Figure 7 (CLY 83).

There is a need for concerted studies of chemistry and of microbiology at the same site in the field. There seem to be no published measurements of valence states and few of sulfide or oxygen concentrations in peat (CLY 83).

The redox potential and the formation of sulfides have also been discussed in chapter 2.

3.2   Organic compounds

The water in peat contains large amounts of organic substances originating from the decomposition of the peat-forming material. Among the best known of these are the humic substances. The characterization of humic substances was described in detail in chapter 2.

Natural humic substances originate from lignin or cellulose, which in a certain state of

decompostion lose their original character
yielding compounds of the polyphenol type. The
types of humic acids differ in the degree of
polymerization, the nuclei and the type of
reactive groups and linkage arrangements.
However, due to their structure pattern the
humic acids have many common and similar
properties.

In Czechoslovak peat water the highest concentra-
tions of humic acids were found in stagnant
water bodies in high moor areas and in the pools
left after peat exploitation. Values of 200 mg/l
with a maximum of 600 mg/l were found in S and
SW Bohemia (CHA 66).

In a study of the humic acids of peat water from
Poland, molecular weights of 175 to 50 000 were
found in the fulvic acid fraction, i.e. the
fraction most readily soluble in water. The mean
molecular weights were usually 2000 to 3000.
There seemed to be the largest iron binding in
the compounds with the highest molecular weights
(CHM 77).

Figure 7. "Redox potential" profile on September 17 in Sphagnum fuscum peat near Abisko, northern Sweden. E_m=as measured; E_5= calculated at pH 5 using the temperature profile and pH value by each point. Acc. to Clymo (CLY 83)
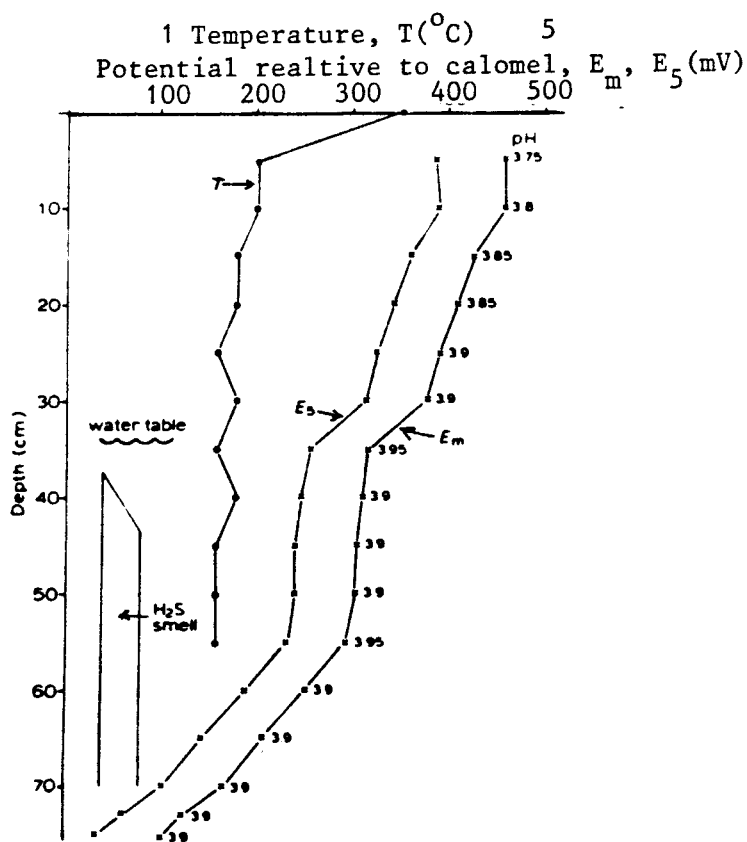
3.3   Inorganic compounds

The concentration of inorganic substances in peat water is dependent on a number of conditions. One important parameter is the quality of the water supply. For oligotrophic bogs the supply is mainly precipitation and the mineral content of the water is thus low. The concentration in the precipitation of elements such as sodium, magnesium and chlorine varies and increases markedly towards the coast. Other elements in precipitation and dry deposition such as aluminium, manganese and iron probably derive from soil dust. Industrial and domestic gases, smoke, industrial dust, road dust, volcanic ash and extra-terrestrial particles (meteorites) may also contribute. Nitrogen may be supplied as ammonia gas, and absorbed directly by acid peat or fixed by the roots in the form of atmospheric nitrogen (CLY 83).

In minerotrophic bogs the main supply of mineral particles in soluble or suspended state is via surface drainage water, floodwater or under-ground water. Suspended mineral particles not only deposit together with the peat formed, but also enter the solution under the action of biochemical and other processes. Redox conditions also influence these processes (LAR 76).

It does not seem to be easy to draw distinct borders between the composition of fen and bog water. This was concluded from a study of 229 samples from 32 different mire complexes in Finland (TOL 74).

The value of pH of the water in peat lands may vary over a wide range. Values between 2.8 and 7.4 have been reported for different organic

deposits, although acid peats are more common than neutral ones (SIK 83). pH is a function of the type of peatland. It is low in oligotrophic bogs (3.5 - 5.9) while in eutrophic bogs it is higher (5.3 - 7.1) (LAR 76). In general fen peats have pH values near neutral as a result of calcium and magnesium ions, which together with bicarbonate raise the pH. Soils and sediments tend toward neutrality when flooded, regardless of the initial pH, mainly as a result of the consumption of hydrogen ions during reduction of iron through equilibrium reactions. Precipitates of $FeCO_3$ and FeS may also be formed during long-term water logging (SIK 83).

The mineralization also varies over a wide range. In oligotrophic bogs the water at 0.5 m depth may have a mineralization of 8-45 mg/l. The main ions are Ca^{2+}, Na^+, K^+, Cl^- and HCO_3^-. The total silica content is higher than that of Ca. In eutrophic bogs at the same depth the mineralization of the water is 60 - 500 mg/l. There are changes in mineralization both within the same bog complex and between different seasons. There are maxima in mineralization in the autumn and in the beginning of spring. Climatologic factors may also influence the mineralization (LAR 76).

In fens with pH about 7.5 the predominating cations are Ca^{2+} (totally about 0.5 mM, of which 1/20 is extractable) and Mg^{2+} (about 0.3 mM, with 1/50 extractable). In bogs with pH about 4.0 the total concentration of extractable Ca^{2+} and Mg^{2+} is not reduced by quite as much (CLY 83).

The relations between total concentrations of the major cations and the amount extractable in different extractants varies depending on the

cation studied. The sodium is bound to the peat
to a lesser degree than Ca, Mg and K (CLY 83).

The relations between the contents of some
metals (Ca, Mg, Zn and Fe) in moor waters and
peat have been studied in Poland. For calcium
and magnesium there was a connection, while for
iron and zinc no such relation was observed. The
iron and zinc in the waters were almost entirely
fixed to the organic substances (CHM 77).

The statistical distribution of water characteris-
tics has been analysed in a study of the water
in 45 undisturbed Minnesota peatlands. Most of
the data were log normally distributed. Only
water temperature, pH, dissolved oxygen and
fulvic acid were normally distributed. Mean
values are given in Table 5 (CLA 83).

The ranges of water compositions in peatlands
are given in Table 4 (p 21).

Table 5. Mean values of water quality characteristics in natural peatland runoff for five sampling periods in 1979 and 1980. Acc. to Clausen and Brooks, (CLA 83)

Characteristic	Units	Peatland Type		
		Bog	Transition	Fen
No. of Samples		66	55	54
Discharge	cfs	2.51	9.31	9.31
Temperature**	°C	11.1	13.4	12.8
Specific Conductance**	μS/cm	45	72	171
Dissolved Oxygen	mg/l	6.4	5.4	5.6
O_2 Saturation	%	56	50	53
pH**		5.6	6.5	7.0
Acidity as $CaCO_3$*	mg/l	21.3	17.1	17.7
Alkalinity as $CaCO_3$**	mg/l	10.1	28.8	74.5
Color*	mg/l	311	260	242
Suspended Sediment	mg/l	5.1	5.3	5.4
Potassium	mg/l	1.24	0.99	1.15
Calcium**	mg/l	6.76	11.57	27.27
Magnesium**	mg/l	2.37	3.93	7.90
Aluminum**	mg/l	0.55	0.25	0.25
Iron	mg/l	2.64	2.07	2.37
Sodium**	mg/l	1.84	1.32	2.74
Manganese**	mg/l	0.14	0.27	0.29
Mercury	μg/l	6	3	5
Arsenic	μg/l	2	3	3
Selenium	μg/l	1	1	1
Total Phosphorus	mg/l	0.06	0.05	0.08
Total Kjeldahl Nitrogen	mg/l	1.5	1.6	1.8
Nitrate-N*	mg/l	0.06	0.05	0.08
Ammonia-N**	mg/l	0.1	0.2	0.2
Organic-N	mg/l	1.4	1.4	1.6
Humic Acid**	mg/l	11	9	8
Fulvic Acid	mg/l	100	89	86
Chemical Oxygen Demand**	mg/l	118	104	97

*Significant difference among peatland types at α = 0.05.
**Significant difference among peatland types at α = 0.01.

3.4   Changes in chemistry with time

The basal and surface peats have high
concentrations of silica, calcium and iron but
the middle peat has lower concentrations. The
same pattern, though less conspicuous, is shown
by aluminum and manganese while potassium has a
particularly pronounced surface peak. Other
elements such as magnesium are more variable,
but sodium and hydrogen ions tend to increase in
concentration from the bottom to the top of the
profile. Nitrogen and phosphorus have been more
seldom measured. They have high concentrations
at the surface but remain constant below it.

The high concentrations in the top 50 cm may
reasonably be connected with the relocating
activities of live plants and animals and with
the change from aerobic to anaerobic conditions.
The major changes occur in the top 50 cm or so
even if some changes occur deeper down (CLY 83).

Even if an organic soil is waterlogged throughout
the year, its profile will exhibit vertical
gradients with respect to oxygen (redox potential)
and temperature. In measurements of the dynamic
changes in peat profiles (pH 2.7 - 3.1) in
northern England over a one year period, a
decreasing pH with depth was found. Differences
have also been recorded before and after heavy
rain (45 mV lower after the rain) in an acid
peat (SIK 83).

There is probably little mass redistribution of
inorganic solutes in humified peat, due to the
low hydraulic conductivity. What may occur is
diffusion. For phosphate a diffusivity of
10^{-10} m^2/s has been recorded. For cations no
data are available, but the high cation exchange
capacity of peat indicates a low diffusivity.

Transport data for cations show that the transport
may not be interpreted as a pure diffusion
process in all cases. If diffusion is at all
important there must be other processes super-
imposed. The valence state also influences the
release.

The inorganic constituents of the upper 20 cm of
the peat may also be relocated as a result of
the absorption of solutes by plant roots. Most
species cycle in the top 20 cm of the peat, but
there are species with deeper roots such as the
bog species Eriophorum (60 cm) and the fen
plants Cladium mariscus and Calamagrostis spp.
and the largest species of Carex Spp. Rubus
chamaemortus has a type of vertical root which
thrusts straight down to near the water table -
10 to 60 cm deep. Reports of 180 cm deep Rubus
chamaemortus roots exist (CLY 83).

4   TRACE METAL REACTIONS IN SEDIMENTS AND PEAT

The behaviour of trace metals in solution is dependent on a number of parameters. The interactions between solid and liquid phase, usually named sorption, are of course important to the behaviour in sediments, soils and peat. The sorption reactions are governed by a number of parameters of the liquid phase such as the content of complex formers, ionic strength and pH as well as by the chemical properties of the trace metal itself.

Since the complex formation is of major importance to the sorption reactions it is treated separately below.

4.1   Complexation

The chemical behaviour of a metal ion in water may be drastically changed by complex formation. The successive complexation of a cation may yield cationic as well as anionic and neutral complexes, all with different properties e.g. as regards sorption onto particle surfaces.

In the following the term ligand refers to any anion or neutral molecule which can be coordinated to a metal ion.

For inorganic ligands, complex formation constants are quite well known for most metals and standard literature summarizing these is available, e.g. Sillén et al (SIL 64).

For actinides the inorganic complexes are less well known, but there is a recent review of data (ALL 82).

Similarly there are standard publications for
simple organic ligands and metals.

However, humic and fulvic acids have not been so
well investigated. These compounds constitute a
heterogeneous group, although they may have
common properties as regards complex formation.
Investigations of complexation have been performed
for both fulvic and humic acids with heavy
metals and actinides and some general conclusions
may be drawn from these.

In a study of the metal binding to organic
compounds in solution in natural water from a
small creek running through a forest, an oxbow
lake with a bog in its catchment, and water
from a raised peat bog, lead was found to be
bound essentially to macromolecules in the
water, while the remainder was found in free
form. Manganese was mainly chelated to oligomeric
organic substances, except in the acid water
from the raised peat bog, where it was also
found in the free state (STE 78). In a laboratory
study of the interactions of copper, lead, zinc
and cadmium with humic acid, the metal ions
seemed to be concentrated to the fractions with
large molecular sizes (>3.1 nm) (GUY 76).

The binding of metals to humic acid seems to
occur at binding sites of different strength.
Guy et al (GUY 76) have found two types of sites
with a difference in stability constant of about
10. They also conclude that divalent metal ions
may be complexed by salicylic type groups in
humic acid. In a study of the complexation of
zinc by humic acid, three different types of
binding sites have been observed (RAN 65). One
of these gave very stable complexes but the

capacity was very low - only 1 % of the total sorption capacity.

For fulvic acid Schnitzer et al (SCH 81) concludes that in the formation of soluble and insoluble metal-fulvic acid complexes, probably chelation, adsorption, and coprecipitation probably occur simultaneously. The type of interaction will depend on the specific metal ion as well as on the pH of the system.

Fulvic acid has been found to form mononuclear (1:1) complexes at an ionic strength of 0.1 between pH 3 and 5 with the following metal ions: Cu^{2+}, Ni^{2+}, Co^{2+}, Pb^{2+}, Mn^{2+}, Zn^{2+} Ca^{2+} and Mg^{2+}. At the same ionic strength but at pH 1.70 and 2.35 respectively, fulvic acid also formed 1:1 complexes with Fe^{3+} and Al^{3+} (SCH 70).

The stability constants of some heavy metals have been found to vary in the order Pb>Cu>Cd>Zn and for the ligands investigated EDTA>humic acid>tannic acid (GUY 76). In another investigation of humic acid the order for some metals was $Cu^2>Fe^{2+}>Zn^{2+}>Ca^{2+}$ (RAN 65). The order of stability of some fulvic acid complexes has been investigated. At low pH it was:
$Fe^{3+}>Al^{3+}>Cu^{2+}>Ni^{2+}>Co^{2+}>Pb^{2+} = Ca^{2+}>Mn^{2+}>Mg^{2+}$
(SCH 70).

In an investigation of water soluble fulvic acid, the order of formation of insoluble metal complexes at pH6 was Fe=Cr=Al>Pb=Cu>Hg>Zn=Ni=Co=Cd=Mn. A correlation with valence, 1st hydrolysis constant and effective hydrated ionic diameter of the metal ions was observed. At high fulvic acid/metal ratios (weight ratio >2) the formation of water soluble fulvic acid complexes was favoured, while insoluble complexes were formed

at lower ratios. High fulvic acid concentrations
seem to inhibit the formation of metal hydroxides
(SCH 81).

Studies of the complexation of Eu(III) and
Am(III) indicate that trivalent actinides form
strong complexes with humic acids. Strong
complexes have also been observed for Th(IV)
with both humic and fulvic acid. Studies of the
interaction of U(IV) and U(VI) with humic and
fulvic acid have indicated the existence of two
different types of binding sites. Strong binding
to humic substances in nature has been reported
for Th(IV), Am (III) and plutonium. The actinide
humic interactions have been reviewed by Olofsson
et al. (OLO 83) and will not be discussed in
more detail here. Reported stability constants
for some metals are listed in Tables 6 and 7.

The complexation with humic substances is
strongly dependent on pH. The stability constants
increase for many metals up to a certain pH e.g.
for zinc and humic acid the maximum is at pH 8.5
(RAN 65:2). The competition with hydrolysis and
carbonate complexation becomes significant at
high pH values for many metals, e.g. for the
actinides (OLO 83). At high concentrations of
fulvic acid, the solubility of metal fulvic acid
complexes was not pH dependent, as it was for
low concentrations (SCH 81).

The value of the stability constant decreases
with increasing ionic strength (SCH 70), cf.
Figure 8.

The interactions of humic compounds with metals
may be influenced not only by equilibrium
reactions but also by reaction kinetics, since

the breaking of metal humate bonds has been
observed to be very slow in some cases (OLO 83).

Figure 8. Effect of ionic strength (μ) on log K
values of metal-FA complexes at pH 3.0.
Acc.to Schnitzer and Hansen (SCH 70)
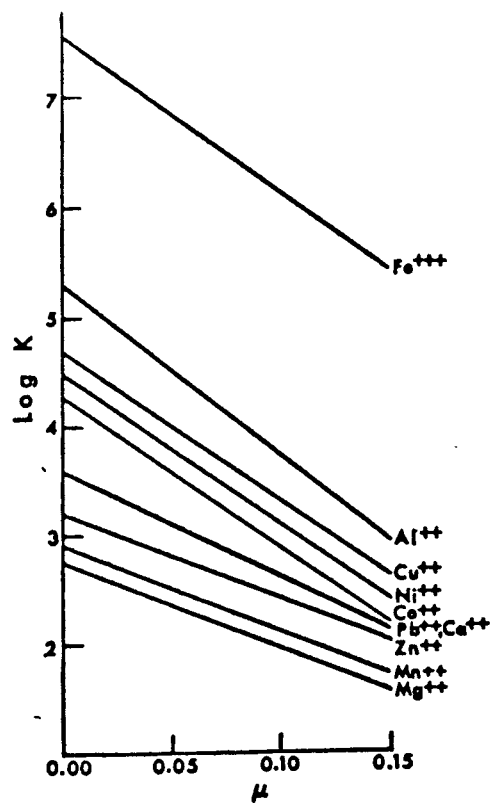
Table 6. log K for some metal fulvic acid complexes.

Me	pH	FA/Me	logK	μ	Ref
Cu^{2+}	3.0	1.0	3.3	0.1	SCH70
Cu^{2+}	5.0	1.0	4.0	0.1	"
Cu^{2+}	3.0		4.7	0	"
Ni^{2+}	3.0		4.5	0	"
Co^{2+}	3.0		4.3	0	"
Pb^{2+}	3.0		3.6	0	"
Ca^{2+}	3.0		3.6	0	"
Zn^{2+}	3.0		3.2	0	"
Mn^{2+}	3.0		2.9	0	"
Mg^{2+}	3.0		2.7	0	"
Fe^{3+}	1.7		7.6	0	"
Al^{3+}	2.9		5.3	0	"
Fe^{3+}	1.7		5.4	0.15	"
Al^{3+}	2.4		2.9	0.15	"

Table 7. Stability constants for some metal-
fulvic acid (FA), humic acid (HA) and
tannic acid (TA) complexes.

Me	K_1	Compl	Ref	
Eu	8×10^6	FA	BER22	
Am	18×10^6	HA	"	
Eu	60×10^6	HA	"	
Th	$10^{10.6} - 10^{13.1}$	HA (lake)	NAS80	
U(VI)	5.4×10^6	HA	LI80	strong site
"	5.3×10^4	HA	"	weak site
U(VI)	2.7×10^7	FA	"	
"	3.6×10^5	FA	"	
U(VI)	2.3×10^6	FA	"	
"	9.2×10^4	TA	"	
U(IV)	9.5×10^6	HA	"	
"	3.2×10^4	HA	"	
U(IV)	4.4×10^6	FA	"	
"	8.8×10^4	FA	"	
U(IV)	8.5×10^6	TA	"	
"	1.1×10^5	TA	"	
Cu	1.6×10^6	HA	GUY76	strong kg/m^3 HA
"	1.2×10^5	HA	"	weak
Cd	1.1×10^5	HA	"	
Zn	1.0×10^5	HA	"	
Pb	3.4×10^6	HA	"	
"	2.0×10^5	HA	"	
Cu	2.2×10^5	TA	"	
Pb	2.4×10^5	TA	"	
Zn	1.8×10^4	TA	"	
Cd	1.8×10^4	TA	"	
Eu(III)	$10^{7.28}$	HA	CHO80	
Am(III)	$10^{3.26}$	HA	"	
Th(IV)	$10^{11.1} - 10^{13.2}$	HA	"	
UO_2^{2+}	$10^{3.28}$	HA	"	
Cu(II)	$10^{5.6-6.3}$	FA	SAA	strong
	$10^{3.8-4.1}$	FA	"	weak
Cd(II)	$10^3 - 10^{4.4}$	FA	"	

4.2   Sorption reactions

The sorption of cations on solid materials may be of major importance in the transport of metal ions in geological media. Since the predominant surface charge of solids in water solutions is negative, the sorption of anions is generally much lower.

In sorption reactions a number of different mechanisms for the interaction of ions in solution with the solid surfaces are included, such as ion exchange, physical sorption, chemisorption etc. See (AND 83), where an overview of the interactions occuring in water/mineral systems is given. The interactions at minerals and rocks with some fission products has also been reviewed in a KBS report (AND 83:2), and some conclusions from this may be quoted:

For cobalt and nickel the sorption is related to hydrolysis. High sorption is observed at a pH where hydrolysis becomes important and the sorption is high as long as cationic hydrolysis products are formed. When the pH is increased, neutral or negatively charged hydrolysis products may be formed and the sorption decreases.

For these reactions nuclide concentration CEC (cation exchange capacity) and temperature are of minor importance.

For strontium hydrolysis is of no importance at the normal pH of groundwater, but in groundwater above pH 9 carbonate complexation may occur. For most minerals, the sorption is low, usually with a pronounced pH dependence. Other important parameters are ionic strength and CEC. A

nonselective sorption due to electrostatic
interactions between negatively charged mineral
surfaces and Sr^{2+} seems to occur.

For <u>cesium</u> no hydrolysis may be expected and pH
has less importance than for Sr. For most
minerals, however, the sorption of Cs is higher
than for Sr. Important parameters are nuclide
concentration and ionic strength. A selectivity
for Cs-sorption is found for some minerals,
mostly sheet-silicates.

For <u>technetium</u> sorption is due to a reduction of
TcO_4^- to $TcO_2(s)$ and as anions are poorly
sorbed, the sorption is dependent on the redox
potential.

<u>Iodine</u> is also anionic and poorly sorbed.
Minerals containing ions capable of forming
iodides with low solubility (Ag, Mg, Pb etc)
do, however, sorb I^-.

For actinides the sorption reactions have been
summarized by Allard (ALL 83).

The experimental observations on the sorption
behaviour of actinides on geological material,
for Am, Np, Pu, Th, Pa and U can be summarized
qualitatively as follows:

- For all systems the sorption is drastically
  increased when hydrolysis starts.
  Sorption maxima are generally obtained
  in the pH-range where neutral hydroxy
  complexes would dominate in solution.

- A decrease in sorption at high pH seems
  to coincide with the formation of
  anionic species (e.g. with OH^- or CO_3^{2-}).

- In the pH-range where hydrolyzed
  products dominate in solution the
  sorption is less affected by the cation

exchange capacity of the solid sorbent than in non-hydrolyzed systems.

- The sorption is fairly independent of the ionic strength for actinides in their lower oxidation states in the pH-range where hydrolysis dominates. For unhydrolyzed systems the sorption decreases with increasing ionic strength.

- The sorption of hydrolyzed species increases slowly with time. The time to reach equilibrium decreases with increasing ionic strength and/or increasing temperature.

- The formation of colloidal fractions e.g. of clayish materials, that would lead to lower apparent distribution coefficients, decreases with increasing ionic strength.

- The sorption actinides in the lower oxidation states (III, IV) are fairly independent of the nuclide concentration (for total concentrations well below the solubility product of any sparingly soluble compound). For the higher oxidation states, in cases when polynuclear anionic species can be formed (e.g. An $(VI)-CO_3^{2-}$), a drastic reduction of the sorption with increasing nuclide concentration (or concentration of e.g. CO_3^{2-}) can be obtained.

- The presence of organic complexes can lead to a reduced sorption.

Note should also be taken of the changes in sorption behaviour which would occur if the oxidation state were varied, e.g. from III or IV, which subsequently could be highly hydrolyzed in the environmental pH-range, to V with a low degree of hydrolysis, or VI to form strong anionic carbonate complexes.

It is however very importanat to note that in all these investigations the influence of organic complex formers has not been considered. This may lead to complexes of different charges and properties.

Sorption may also occur on solid organic substances.

Dissolved organic matter may lead to a lower adsorption of metal ions on mineral surfaces due to an adsorption of organics on the solid. This has been shown for the sorption of copper and zinc on $\gamma-Al_2O_3$ and lake sediments, cf. Figure 9 (BAC 82).

Figure 9. Cu, Zn and phosphate adsorption as a function of particle concentration and natural organic matter. O---O without organic matter, ●——● with 2 mg DOC/l Acc. to Baccini et al (BAC 82)
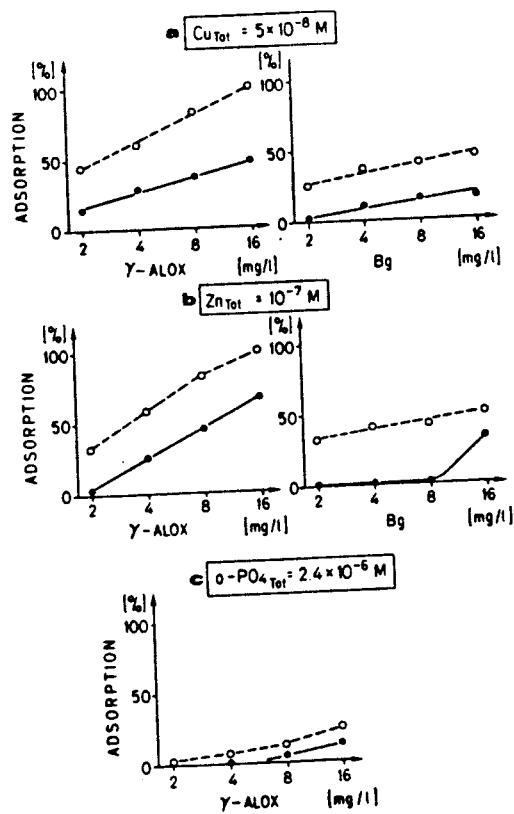
In an investigation of the sorption of metal
ions on humic acids isolated from marine
sediments, an effective sorption by chelation,
cation exchange and surface adsorption has been
observed. The quantities of metal ions complexed
varied from 40 to 205 mg/g of organic matter. In
the presence of Co, Cu, Mn, Ni and Zn humic acid
and peat moss showed preferential absorption of
Cu. The bonding strength was very firm because
it could not be displaced by ferric iron or
cation exchange reagents. Peat moss was found to
absorb significant quantities of various metal
ions (RAS 74). Unfortunately all studies were
performed with constant concentration (200 ppm)
of the metal ion and the result was only given
as mg sorbed/kg solid, not in terms of K_d or
sorption isotherms.

By developing a simple equilibrium model for the
adsorption of trace metals based on the linear
portion of adsorption isotherms, it has been
shown that trace metal partitioning can poten-
tially be predicted from laboratory studies with
the use of conditional equilibrium adsorption
constants. The constants determined for Cu and
Cd adsorption on bentonite clay, $Fe(OH)_3$, MnO_2
and humic acid in seawater show that this model
is applicable for trace metal concentrations
existing in the natural environment. The constants
determined for Cu were much higher than those
for Cd for all solids. The affinity of humic
acid relative to the other phases was high for
Cu and low for Cd. On the basis of the laboratory
results, the model predicts that the clay
fraction will be a major sink for Cu and Cd in
the artificial phases used (OAK 81).

The sorption isotherms for Cu and Cd in seawater on humic acid are linear over a larger concentration range than observed for bentonite, $Fe(OH)_3$ or MnO_2, cf. Figures 10 and 11. The distribution of the metals between the different solid phases is given for various concentrations of clay. For Cu the humic acid is responsible for a large part of the sorption capacity, while for Cd the clay is more important. As might be expected for a hydrolyzing metal, the sorption increases with pH for Cu in the range where hydrolysis becomes important. An almost total sorption occurs below pH 8. The copper sorption on bentonite is not especially dependent on the salinity of the water phase - a fact that might be expected from the hydrolytic behaviour of Cu.

In Table 8 the distribution coefficients for Cu and Cd on the various solids used are given. The sorption is high - the unit l/g equals m^3/kg, and Cu is thus almost completely sorbed (OAK 81).

Table 8. Distribution coefficients for Cu and Cd
in seawater. Acc. to Oakley et al
(OAK 81).

solid phase	Cu		Cd	
	K_{AD}, L/g	R^2	K_{AD}, L/g	R^2
bentonite clay	43	0.97	1.1	0.90
Fe(OH)₃	205	0.86	4.4	0.98
MnO₂	7339	0.95	15.0	0.96
humic acid	366	0.98	0.2	0.99

ª pH 8.0; S = 32‰; T = 20 °C.

Figure 10. Adsorption isotherms for Cu in
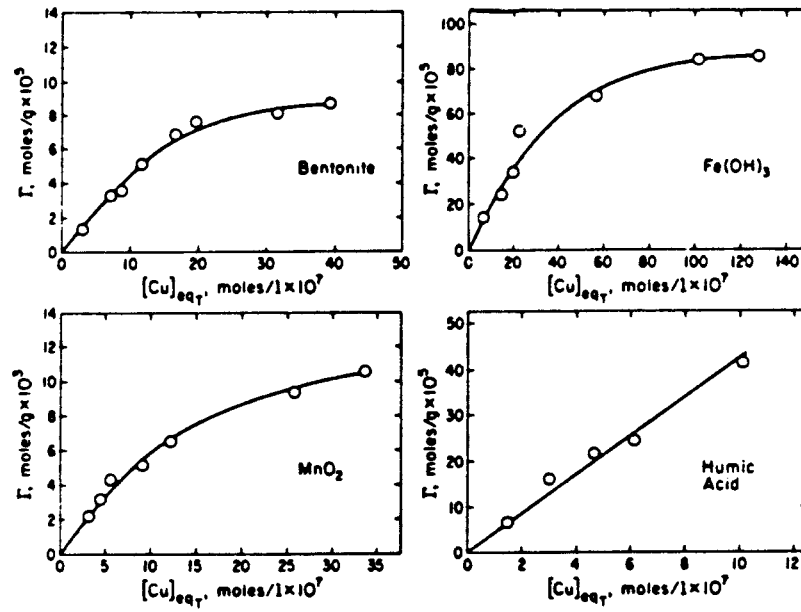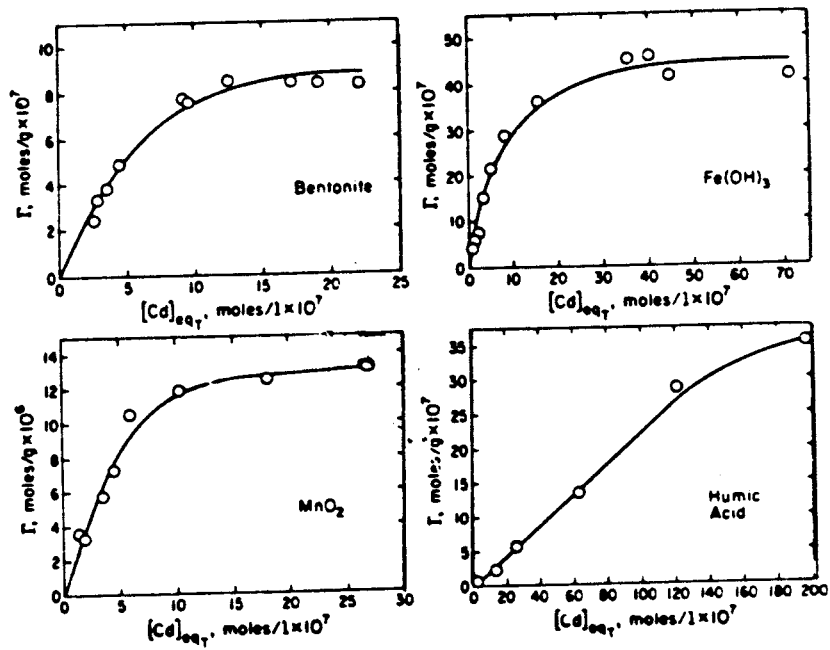   seawater. Acc. to Oakley et al (OAK 81)

Figure 11. Adsorption isotherms for Cd in seawater.
           Acc to Oakley et al (OAK 81)

5   TRACE METALS IN PEAT - ANALOGIES

The content of mobile forms of microelements in peat depends on the pH of the peat as well as its degree of decomposition. A high migration ability of zinc and manganese in peat deposits has been established. In general lowmoor peat deposits (pH 7-8) contain the greatest concentrations of microelements as compared with highmoor deposits (pH 3.5-4), with the exception of some elements such as zinc and others. Some kinds of peat possess the selective ability to concentrate certain microelements several times more than background contents. For example highmoor Schezeria sphagnum peat accumulates copper and zinc much more than other kinds of peat; manganese is accumulated in Sphagnum peats kinds of the highmoor type; lowmoor woody peat contains several times as much cobalt and molybdenum as other kinds of peat. Plants of eutrophic type are much more enriched in microelements. Concentration increases are observed in boundary zones adjacent to the point where feed water flows in. Two maxima of the contents are generally encountered: in the root layer and in the ground horizon. (Studied elements: V, Cr, Co, Ni, Cu, Be, Ga, Mo, Pb, Zr, Y, La) (LAR 72).

The contents of trace elements in Northern Ontario peat have been analyzed at 0-20 cm and 20-40 cm for Pb, Cu, Cr, Cd and Zn. Only in the case of Pb were differences observed between different peat types and depths (GLO 82).

In a study of the mobility of ^{226}Ra in soils, a strong fixation of Ra released 30 years ago has been observed in a strongly podzolic gley soil. Simple organic acids seem to increase the mobility. There is a direct correlation between

the ^{226}Ra content and humus, exchangeable bases and pH in the root zone (TAS 78).

Uranium escapes from the lithosphere in the form of mobile carbonate complexes. Humic acids may complex the uranium and solid humic compounds give a fixation of uranium. In a study of 14 arbitrarily chosen peat bogs in different parts of Finland, the uranium concentration was found to have a sharp peak at a level about one fourth from the bottom of the bog. Lead accumulates both at the surface and at the same level as uranium, cf. Figure 12. The accumulation of lead at the surface is attributed to the influence of living plants, while a complexation with humus may explain the accumulation of both uranium and lead at greater depths. Nickel is not enriched by living plants and the complexation with humus is also less than for the other metals (ERÄ 69).

Uranium is often found in coal strata, and the enrichment of U in coal has been demonstrated in laboratory experiments (MO054). Lignite, peat and other similar decayed fossil plant debris adsorb uranium quickly from a very dilute aqueous solution in the pH range of 3-7. A very steep sorption isotherm has been observed (Langmuir isotherm). At trace concentrations the enrichment factor (distribution coefficient) is 10 000 ml/g (10^3m^3/kg). The initial concentrations used in these experiments correspond to those commonly found in nature. Saturation of peat occurs when the content is about 10 %. The investigators have showed that the sorption is reversible. For thorium similar sorption isotherms have been reported, with enrichment factors of 30 000 ml/g (30 m^3/kg). At pHs below 3 the U can be easily desorbed, while fixation occurs between pH 3 and 7. It has also been shown that the enrichment

constant increases very strongly with the valency and with the atomic weight of the cation. UO_2^{2+} and Th^{4+} cations and the ions of rare earth metals are adsorbed on to peat to an extremely large extent, because their valencies and atomic weights are high. The various cations compete with each other and with those of higher valency and atomic weight and expel others of lower atomic weights and valency. If the humic acid is extracted from peat by NaOH the peat looses the major part of its U fixing capacity while the resulting humic acid, liberated by acid, still has the U sorbing capacity.

The author claims that the fixation by means of cation exchange on to humus is responsible for the enrichment process which concentrates the uranium in a proportion of about 1:10 000 in fossil plant debris and bioliths (SZA 58).

Uranium is considerably more mobile than radium in surface waters containing small amounts of calcium bicarbonate. Radium is readily sorbed by clays as well as by peat. In peat, uranium is bound to the humic and fulvic acids soluble in alkali, whereas the radium is associated with the insoluble residue. Under oxidizing conditions the elements appear to be retained by ion exchange but in the presence of high concentrations of calcium in the water, the radium in the peat and in the insoluble residue becomes nearly unexchangeable. The exchangeability of the radium bound to the humic acid is however retained. These studies have been performed in the taiga permafrost areas. Batch sorption experiments showed an increased uptake of radium by peat when pH increases from 5 to 8, while the uptake of uranium was highest at pH 6, cf. Figure 13. Distribution coefficients are not given in the

paper, but these are said to vary greatly with
the kind of peat. Desorption experiments showed
that much of the uranium could be eluted from
the peat, while radium was less easily desorbed.
For radium the influence of calcium on the
desorption properties was evident; thus the
desorption was much less if the sample was
pretreated with Ca-solution. The table also
gives a comparison of the sorption on some
substances found in peat: humic acid, humins,
floodplain loam, and pure kaolinite. The sorption
of radium on peat is similar to that on humins,
indicating that radium is bound to the humin
fraction. The mobility of uranium observed in
the laboratory experiments is not related to the
observed enrichment of U in peat deposits. A
stronger binding of U(IV) than U(VI) may be
involved since there is a possibility of reduc-
tion of the U in peat (TIT 67).

Lower plants (mosses, algae and microorganisms)
accumulate more uranium than higher plants.
Accumulation of uranium by living plants plays
an unimportant role in its concentration in
peat. A study of the uranium content in plants,
peat water and soil in eastern Siberia has been
performed in areas where the U content of the
water ranged from 1.5 to 2.3 ppb. Little difference
was found in the U contents of soil samples
taken at the surface and at 0.5 m depth. Mosses
were shown to accumulate large amounts of U and
there were correspondingly high concentrations in
peat (LOP 70).

Figure 12. Ni, Pb, U, and ash contents and pH-
          values in a peat bog. Acc to Erämätsä
          et al (ERÄ 69)
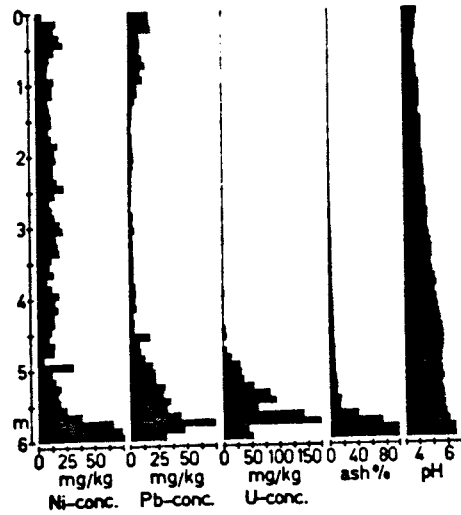
Figure 13. Effect of pH on the sorption of U and
          Ra in peat. Acc. to Titayeva (TIT 67)
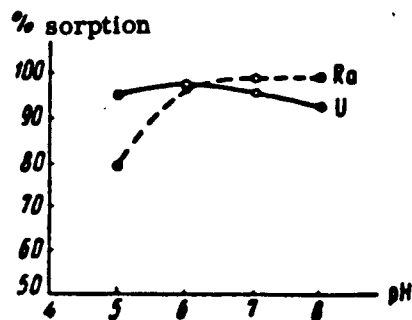
6   CONCLUSIONS

In the evolution of a lake or a Baltic bay into farming land or peatland a wide range of possible chemical conditions may be expected to occur within bottom sediments and in the peat.

There are some parameters in the interstitial water which are of major importance to the behaviour of trace metal ions. The most important of these seem to be pH, Eh, ionic strength and content of complex formers.

pH may vary from 3.5 to over 8. The low values are most likely to occur in peat and in acid soils, while pH values of 7 to 8 are the most common in sediments. In some types of peat pH values up to 7 may also occur. Since the possible sorption of cations usually increases with pH, especially for easily hydrolyzed ions, the systems with high pH may be expected to exhibit a lower mobility of trace cations.

The absolute value of the redox potential, Eh, is a quantity that is difficult (or impossible) to determine. A significant difference between oxic and anoxic conditions may nevertheless be observed. In sediments Eh changes may occur over the year, especially in the surface layer due to variations in the aereration and oxygen consumption of the overlaying lake water. Deep sediments and peats also tend to be depleted of oxygen. The behaviour of redox sensitive metals (e.g. Tc, U, Pu) may be determined by the redox conditions to a very large degree, while for others there is one predominant valence state and the redox conditions will only be of importance if they influence the properties of the solid or water chemistry.

The _ionic strength_ also varies over a wide range for the water types in question. The amount of dissolved ions may be from some mg/l up to 16 000 mg/l. This means that the sorption may vary very much for uncomplexed cations depending on the type of water. The complexation is also dependent on the ionic strength (stability constants decrease with increasing ionic strength).

In all the water types discussed, the content of potential _complex formers_ may be quite high. Among these there are both inorganic species like phosphate, carbonate and sulphate and organic compounds (humic substances). The influence of complex formers may be due to several mechanisms. They form complexes with trace metals that may be both more and less soluble than the uncomplexed ion. The complexes may also be sorbed to a greater or lesser degree than the uncomplexed ion. There may also be a sorption of organic complexants on the mineral surfaces that inhibits sorption of metal ions. Complexation may occur with both macro and trace components and competition has thus also to be considered.

Complexation may occur with solid organic substances. This may in some cases be virtually irreversible.

The properties of the _solid mineral phases_ present are also of great importance to the behaviour of trace metals. Metal ions may be sorbed on the surfaces, causing both reversible and irrebersible incorporation into the solid phase. The interstitial water composition is also influenced by the solid phases. The interactions may however be slow, and equilibrium conditions cannot always be expected.

Some general conclusions on the behaviour of
trace metals in peat may be drawn from
investigations of metal enrichment in nature.
Observations have been made on the difference in
the occurence of uranium and lead in peat
deposits. Conclusions about the plant
availability have also been drawn from this type
of investigations. The investigation show
enrichments of uranium and lead in the bottom
layers of peat, while lead is also enriched on
the surface.

Radium seems to be more strongly bound to the
organic solids than is uranium, and here the
competition with calcium is important.

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Anne-Britt Wilkens

HUMAN IMPACT DURING THE EVOLUTION
OF A LAKE OR A BAY

TABLE OF CONTENTS

1   INTRODUCTION

Man's basic relationship to the environment has
always been founded on the uses he can make of
various components for food, shelter etc. At the
same time he himself is a part of the ecosystem,
and he has many opportunities to change the
system so as to fit his needs.

If a lake or a bay is assumed to have been
polluted by some metal ions, for instance, and
this pollution is assumed to be bound in the
sediments or lake biota, which ways can this
pollutant be released to the environment during
and after the terrestrialization of the lake?
How can human impact influence the release and
recycling of the element?

It is almost impossible to predict all the ways
in which man may in the future utilize a lake or
bay area and its resources during all stages of
development.

Consequently in this paper different methods
that have been applied especially during the
20th century are discussed, and mention is also
made of some aspects of applications which are
directed towards energy conservation that may be
relevant in the near future. It is not possible
to deal with all the probable consequences, but
some of the more important effects are treated.

A great deal of the arable land in Sweden has
once been at the bottom of lakes or seas. During
the 19th century the increasing population
necessitated the reclamation and cultivation of
bog areas and major drainage projects.

Nevertheless, more than 20 % of the Swedish land area is nowadays classified as wetlands, and this constitutes a potential resource for both food and energy production, see figure 1.

The eutrophication rate of a lake may be counteracted or accelerated as a result of human impact. On one hand, some of the most eutrophicated lakes have been restored by chemical precipitation and sedimentation, for instance Trummen near Växjö, and in connection with aquaculture the water has to mee certain demans as to quality claims. On the other side it is well known that man accelerates the eutrophication rate as a result of the fertilization of arable and forest soils clear-felling within the catchment area, or the discharge of waste water from industries or households.

NW95 ASG
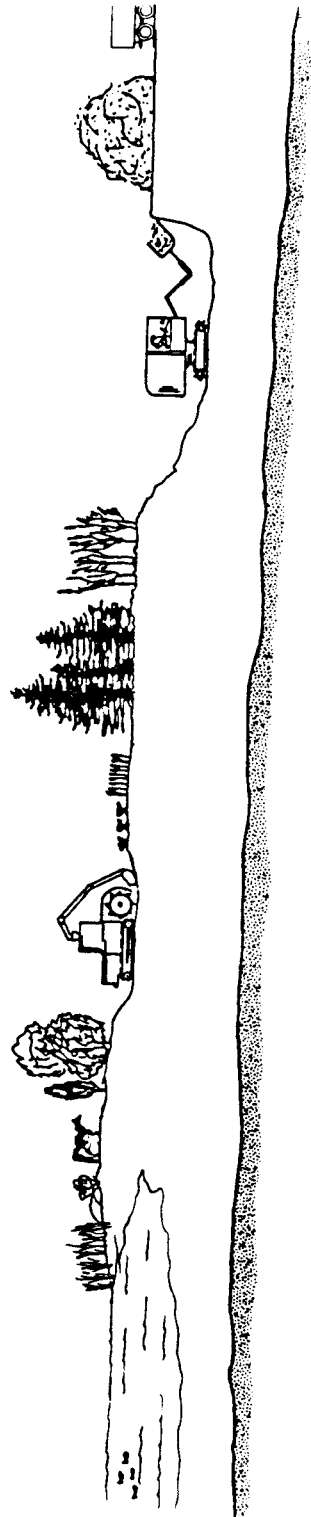
Figure 1. Human impact during the development
of a lake. After /28/

2   DIRECT IMPACT

This paper draws a distinction between direct or intentional impact, and indirect impact.

Indirect impact refers to involuntarily attained effects, usually with a negative influence on the environment, such as acidification and leaching.

2.1   Exploitation of the area

During its development from open water to solid ground the area passes through stages when it can be classified as wetland.

The major categories of wetlands according to Swedish conditions are listed as follows:

-        mires (peatlands)
-        littoral wetlands
-        moist or wet areas without peat
         formation /17/

The interest in utilization of our wetlands has increased considerably during the last decade, and this utilization may have lasting consequences. Hydrology, water chemistry, vegetation and fauna will be affected.

In this paper, the exploitation of an area is assumed to include three main activities: drainage, agriculture and forestry. Drainage is assumed to be a prerequisite for the other two.

The use of wetlands for recreational purposes is a possible application which is not discussed in this paper.

2.1.1   Drainage

The most radical impact on a lake may be assumed to be produced by draining it.

During the second half of the 19th century a great period of drainage and reclamation ventures began in Sweden. Many lakes were drained dry in order to get new arable soils.

For the exploitation of wetlands, sedimented areas or peat soils drainage is the primary procedure.

A requirement for the effective utilization of soils with imperfect or poor natural drainage is the removal of excess soil moisture.

In /23/ it is stated that water moves faster laterally than vertically in poorly humified peat but that the difference in more humified peat is insignificant. Poorly humified sphagnum peat showed the greatest transmission rates.

Two factors are of special importance:

-        the water transmission of the peat
-        the root penetration power in different types of peat.

The objective is to lower the moisture content in the upper layers of the soil, permitting oxygen penetration to the root zone and diffusion of carbon dioxide from the roots.

Surprisingly, even irrigated soils often require an extensive drainage system.

It is well known that the drying of fresh peat
is attended by shrinkage and that neither the
original water content nor the original peat
volume is regained, when the dry peat is brought
into contact with water again /12/. To a certain
extent the shrinkage is reversible. Consequently
it is found that the total shrinkage on drying
may amount to 80 vol % whereas the reswelling of
a decomposed peat may be limited to 10 % of the
original volume. As shrinkage is generally
greater the more the peat is decomposed it
follows that this property is particularly one
of humified matter. See figure 2.
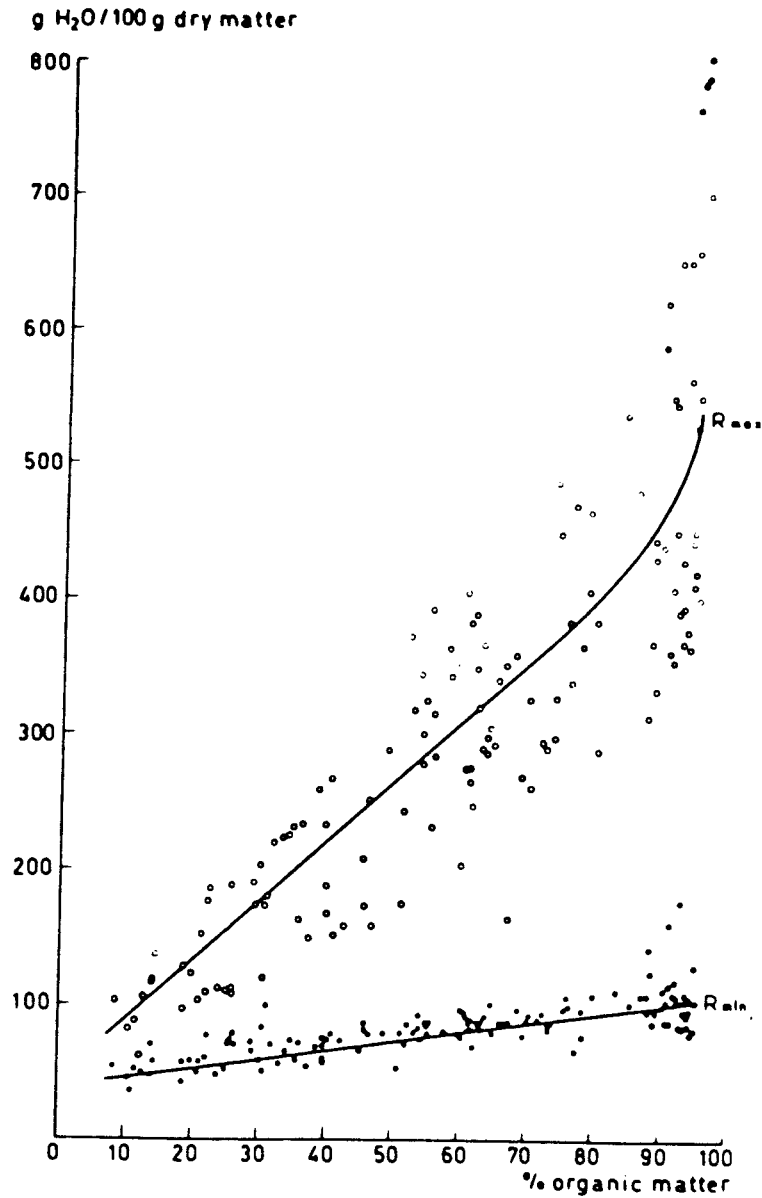
Figure 2. "Centrifuge moisture equivalents"
         (remaining water content after centri-
         fugation at 1000xg) of fresh peat
         (R_{max}) and of peat which has been
         dried, ground, and resaturated with
         water (R_{min}), as a function of organic
         matter content. From /12/.

2.1.2   Agriculture including the nature of
         organic soils
_____

Arable soils are discussed in detail in agricul-
tural and soil literature /5, 8, 18/. On the
basis of organic content, two general groups of

soil are recognized, mineral and organic. Soils
with more than 40 % organic matter are classified
as organic or Histosols /14/. Our knowledge of
organic soils is still imcomplete, for instance
as regards the particle size distribution.

Organic soils are produced by biotic and abiotic
factors. They can be divided into two groups,
sedentic or autoctonic and sedimentic. Peat soil
belongs to the first group but gyttja and dy
belong to the second.

The growth of peat is described in a separate
paper by Karin Andersson, 1984. Gyttja is formed
in lakes or bays of more or less decomposed
plants and animals ("ävja") mixed up with clay
minerals. Dy is a light black-brown earth formed
by chemical precipitation in oligotrophic lakes.
The figure below illustrates the stratification
of different sediments in a Swedish valley.

Figure 3. Vertical section showing the stratifi-
cation of a typical valley in Sweden /34/
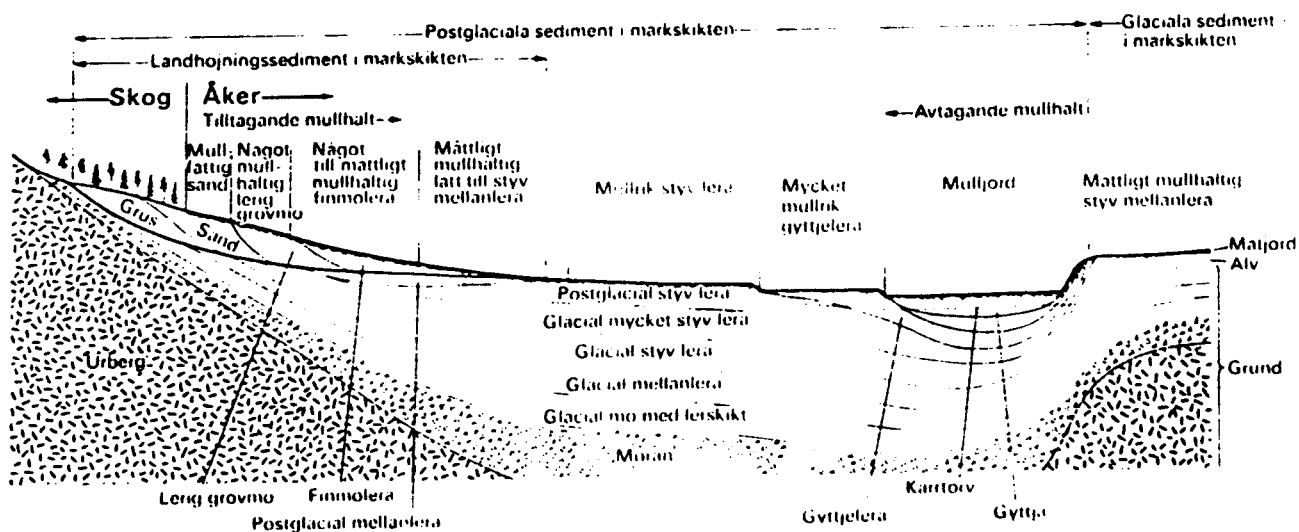
Organic matter accumulates wherever the net rate of photosynthesis exceeds the rate of oxidation of organic matter. The oxidation of organic matter may be reduced or limited by several factors, such as low temperature, the absence of mineral nutrients, lack of water (under aerobic conditions) or the lack of oxygen (under anaerobic conditions).

As illustrated in the foregoing figure most soils, with the exception of blocky moraine, can have potential interest as arable soils.

It must, however, be noted that organic and mineral soils differ considerably from both a chemical and a physical point of view.

For example, peat soils have a lower bulk density than mineral soils and in /23/ the following average values of bulk density were recorded.

Soil	Bulk density kg m^{-3}
gyttja	375
dy	350
bog soil(kärrjord)	250
swamp soil(mossjord)	150

The low density of an organic soil is a disadvantage in connection with cultivation and crop production. It gives a lower bearing capacity and in agricultural techniques today, most machines are extremely big and heavy.

The application of mineral soils to ameliorate the physical properties of peat has been tried earlier /23/.

The structure of a soil should also be considered.
Intensive cultivation of some organic soils will
produce a powdery structure and thus a poorer
water-sorption capacity.

It should be noted that a large amount of the
water content in a peat soil is colloidally
bound and not available to plants; this is
illustrated in /23/ where a comparison is made
between the wilting percentages of three
different soils.

Soil	Wilting percentage
Sand	1.5 %
Sandy clay	7.2 %
Peat	33.2 %

The colloidal nature of organic soils is more
pronounced than in most mineral soils. Their
surface area exceeds even that of montmorillo-
nite clay by a factor of two to four and the
cation exchange capacity is correspondingly
greater. Cations are adsorbed in the same order
of magnitude in organic as in mineral material:

$$Ca > Mg > K \text{ or } Na$$

Some important differencies should be noted. The
amount of adsorbed Ca is much greater in organic
than in mineral soils, and a higher proportion
of total, mineral cations are exchangeable in
organic soils.

These two facts are shown in the following
table.

Table 1. Cationic conditions in two woody peat soils, one low and the other high in lime /8/

Cation	Exchangeable metallic cations meq/100 g		Percent of cations exchangeable	
	low lime peat	high lime peat	low lime peat	high lime peat
Calcium	39.8	159.7	80	75
Magnesium	9.7	12.0	100	51
Potassium	0.87	1.16	34	27
Sodium	0.80	0.87	-	-

All this is in contrast with a mineral soil, where usually less than 35 % of the calcium, less than 10 % of the magnesium and less than one (1) % of the total potassium are exchangeable.

Hydrogen and aluminium tend to dominate acid soils. Both contribute to the concentration of H^+ ions in the soil solution, adsorbed hydrogen directly and aluminium indirectly through hydrolysis, as follows:

$$Al^{3+} + H_2O \quad -- \quad Al(OH)^{++} + H^+$$
$$Al(OH)^{++} + H_2O \quad --- \quad Al(OH)_2^+ + H^+$$
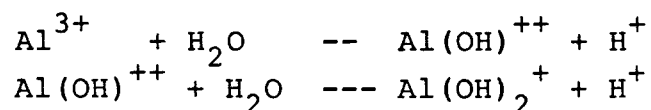
Most of the other cations, exchangeable bases, neutralize soil acidity. The proportion of the cation exchange capacity occupied by these bases is called the percentage base saturation /8/. From an agricultaral point of view this is an important factor.

2.1.2.1 Grazing

An initial step in the utilzation a wetland area, which may not even have been drained, is by grazing.

In earlier times the drove of cattle was far
more important in agriculture than it is
nowadays, and the number of animals on a farm
was often maximized rather than optimized, in
relation to the available area. The consequence
of this was that the utilization of wetlands for
grazing and hay-cutting was a requirement.

Manure was a limited resource preferably used in
arable fields. The grasslands became successively
depleted of plant nutrients. Not until the
industrial fertilizers came, could a restricted
fertilization of natural grasslands start.

When covered by permanent grassland an organic
soil is more slowly and altogether less oxidized
than through crop cultivation. The grasslands do
not need such efficient drainage as arable
fields and can even withstand short periods of
flooding, near rivers or streams.

In forage production it is, however, verified
that a rather low ground water level gives a
better quality as the poor grasses are pressed
back by the better types with deeper root
systems.

2.1.2.2 Crop production

The necessary steps to bring a peat bog or
drained wetland under cultivation, aside from
drainage, vary with the type of peat and the
kind of living and dead matter that covers the
area. The first step is to remove all of the
coarse debris, including stumps and roots.

In some countries it is a common practice to
burn off the remaining coarse material, but the

deeper layers must be sufficiently wet not to burn.

A better technique is to cultivate the soil a few times and to accelerate the oxidation of the upper soil. A newly reclaimed area is usually an unsatisfactory medium for crops. The peat bog has been under anaerobic conditions perhaps for centuries and the roots of cultivated plants do not penetrate this medium. This can be due to toxic amounts of, for instance, sulphuric acid or metal compounds but is partly dependent on the lack of oxygen.

After the initial operations it can be recommended that suitable grasses should be sown and the soil left in pasture for a couple of years.

In considerating the use of peat for crop production it is well to emphasize that there is usually a great difference between raw peat and peat which has been drained and oxidized for some time. In the raw state all types of peat have a low value of nutritive constituents. When the peat is properly aerated, cultivated etc the change from peat to a material ressembling humus may take place rapidly /5/.

In practically all peats the levels of elements are inadequate for crop production, and a succesful conversion to productive soil depends largely on the application of balanced fertilizers. Not only the major elements are needed, but different crops require minor elements in varying degrees, and copper, molybdenum, boron, zinc and manganese appear to be important.

Lime deficiency is characteristic in most
sphagnum and woody peats. The pH values in these
peats commonly lie between 3.0 and 5.0.

Deficiencies in minor elements deficiencies are
common in cultivated peats or mucks, but properly
managed peat soils can be most productive.

After a few years under grass almost all crops
can be cultivated in this type of area. Especially
good results have been obtained with onions,
cabbages, beans and potatoes.

2.1.3   Forestry

In these days the most extensive demands on
wetlands come from forestry. In Sweden these
claims have been quantified to 1.5 million
hectares.

Around 7 million hectares of Swedish peat soils
have today none or little forest production, but
a great deal of this area will presumably become
productive in the future, above all as a result
of drainage techniques, which in recent times
have been greatly developed. However, the
drainage must be supplemented by a nutrient
analysis of the peat, especially as regards
potassium and phosphorus down to the depth where
the roots of the trees mostly stop.

2.1.3.1 Traditional forest culture

Recent plans for increasing wood production in
Sweden aim at increased drainage operations, but
the nutritional status of the peats often
becomes a delimiting factor.

Forestry differs from agriculture in that the rotational time for the crop is much longer, and forest plantation does not decrease the organic peat layer as quickly or to the same extent as in agriculture.

Forestry today include among other things clear felling, fertilization, and treatment with chemical biocides. Clear felling especially when accompanied by burning, quickly solubilizes the nutrients and an increased eutrophication rate in recipient water is a risk, see also the chapter on leaching, 3.2.

Wood from traditional forest culture can be utilized in many ways, as a building material for instance or as biomass and fuel.

2.1.3.2 Energy_forest

The cultivation of energy forest entails the plantation of fast growing trees in order to produce raw material for energy production.

Usually one can distinguish between intensive and extensive cultivation. The term intensive forest cultivation refers to planting with a quick crop rotation, 1 to 5 years, using different Salix species /13/. The plants are fertilized according to the growth capacity and if necessary the plants are also irrigated. Harvesting is carried out with specially designed machines. New generations are ensured since shoots grow out from the stumps, and these have been calculated to have a life span of 20-30 years.

The extensive cultivation of energy forest utilizes species such as poplar, aspen and

birch. These are suitable for a slower crop
production over 15-20 years. The cuttings are
planted out more sparsely and even thinned out
if necessary. Fertilization is carried out at
the same time as planting and thereafter at
intervals of several years. Felling of the
mature trees is planned using existing
techniques.

Extensive forest cultivation does not make such
demands on ground preparation and drainage as
intensive energy forest.

The expected annual production is calculated to
be 20 tons (DM) per hectare on farm land and 15
tons per hectare on wetlands. In /13/ wetland
areas totalling 2.1 million hectares are thought
be suitable for energy forest, i.e nearly 50 %
of the total wetland area.

The cultivation of energy forest is in its early
stages; nevertheless environmental research has
begun. So far this has been conducted in the
following areas:

- water balance
- water quality
- flora
- fauna
- landscaping.

2.2.1   Aquaculture

The rapid development of aquaculture today can
partly be explained by the increasing pollution
of seas and lakes. For instance, the pH-values
in water are important for the comfort of the
fishes. For fish-production pH-values are
recommended to lie between 6.5 and 8, and

strongly basic as well as acidified waters are
classified as unfavourable /3/.

Pollutants in the water can be accumulated in
the fishes either directly through the water or
through polluted food.

Culture techniques are developing along low-energy
cultivation lines where the nutrients are
recycled /2/. Aquaculture-agriculture-industry
is intended in the future to be more and more
integrated, so that warm effluents and chemical
waste products can be utilized for fish-production.

The environmental consequences of cultivation of
fish, mussels and algae in water bodies and
coastal areas depend on many factors, e g the
volume of production and the techniques used. In
aqauaculture the surrounding water may be loaded
with phosphorus from fodder excretion products.
The environmental consequences of the
cultivation of fish are however estimated to be
small /1/ compared with other activities. The
aquaculture is often judged to be "self
regulating", since negative environmental
effects also damage the cultivation.

The treatment of lakes with lime is sometimes
necessary to counteract the negative effects
which acidification may have on aquaculture
activities.

2.2.2   Reed

Large amounts of organic matter are produced by
aquatic plants in marshes, in littoral zones of
lakes and in brackish water areas /16/.

At present only a small portion is utilized, but as late as 25 years ago reeds were harvested as forage in our country.

The total reed-covered area is estimated to be between 50 000 and 100 000 hectares. Reeds grow on a variety of substrates, from minerogenic to organic soils and they can even grow in anaerobic soils. They are thought to extract nutrients mainly from the soil, but when exposed to nutrient-rich water they can also take up elements through the finely branched roots on the lower part of the culm /16/.

Reeds have a deeply penetrating perennial biomass, composed of rhizomes. The rhizomes have a life span of a few years.

This vegetation rarely suffers from water deficiency and as the plants often grow in nutrient rich medium, a high productivity can be expected.

Aquatic plants have for a long time been utilized as paper, building material etc. Nowadays the greatest interest is associated with the energy content. Reeds harvested in summer can be calculated to have an energy content which corresponds to more than half a litre of oil calculated as biomass m^{-2} /16/.

The use of aquatic plants and macroalgae for energy production has been discussed, for example using:
-      methane fermentation
-      biomass fuel and wood mixture.

Biomass (woodchips, straw and reeds) cannot be burnt as efficiently as oil. There is, however,

little variation in energy content between
different plants, especially when calculated on
an ash-free basis. But while algae have a high
ash content, in some cases over 50 %, winter-
harvested reeds have a mean-ignition residue of
4 %; this should be compared with wood having
0.1-2.0 % of the DM.

The environmental effects connected with the
conversion or combustion of reeds depend to a
great extent on the technology. When compared
with fossil fuels biomass has the advantage of a
low sulphur content, for reeds this varies
between 0.1 and 0.2 %. Reeds also contain small
amounts of heavy metals and for this reason the
ash could possibly be returned to the reed stand
to recycle nutrients /16/.

2.2.3   Peat

Active peatlands are among the few existing
ecosystems which produce a long-term surplus of
energy /23/. The possibilities of utilizing peat
deposits for agricultural and forestry purposes
are discussed in previous chapters. But peat has
unique properties. It has an excellent power of
adsorption and adsorbs water to about 20 times
its own weight, but can also take up organic and
inorganic nutrients as well as pollutants.

Undrained peatlands have also been used for
experimental dumping of sludge /21/.

Presumably future research will find ways to
convert peat into other macromolecular
substances with diversified potentials, such as
waxes and medicinal agents or, into smaller
molecules such as solvents.

The most important possibilites for utilizing
peat can however be summarized as follows:

-        as an energy source
-        as mineral soil fertilization
-        as raw material.

Since peat may contain impurities which could
pollute the environment when it is utilized, the
first two applications will be discussed in this
paper.

2.2.3.1   Peat in energy production

Since the oil crisis of 1973, energy saving and
oil substitution is an important aim in the
Swedish energy policy. For example, gasification
studies have been performed in Sweden for more
than 10 years /24/.

Peat may be regarded as a reservoir of stored
solar energy which can be recovered in different
ways. It must however be kept in mind that peat
is a non-renewable energy source, and that the
carbon dioxide emitted through burning of peat
is not balanced by a similar uptake.

The peat reserves in Sweden are estimated at
4000 million tons of oil equivalents, correspon-
ding to the total biomass production in Sweden
during 100 years. Peat is a young material
compared to the fossil fuels, with chemical
resemblances to ignite, see figure 4 /20/.
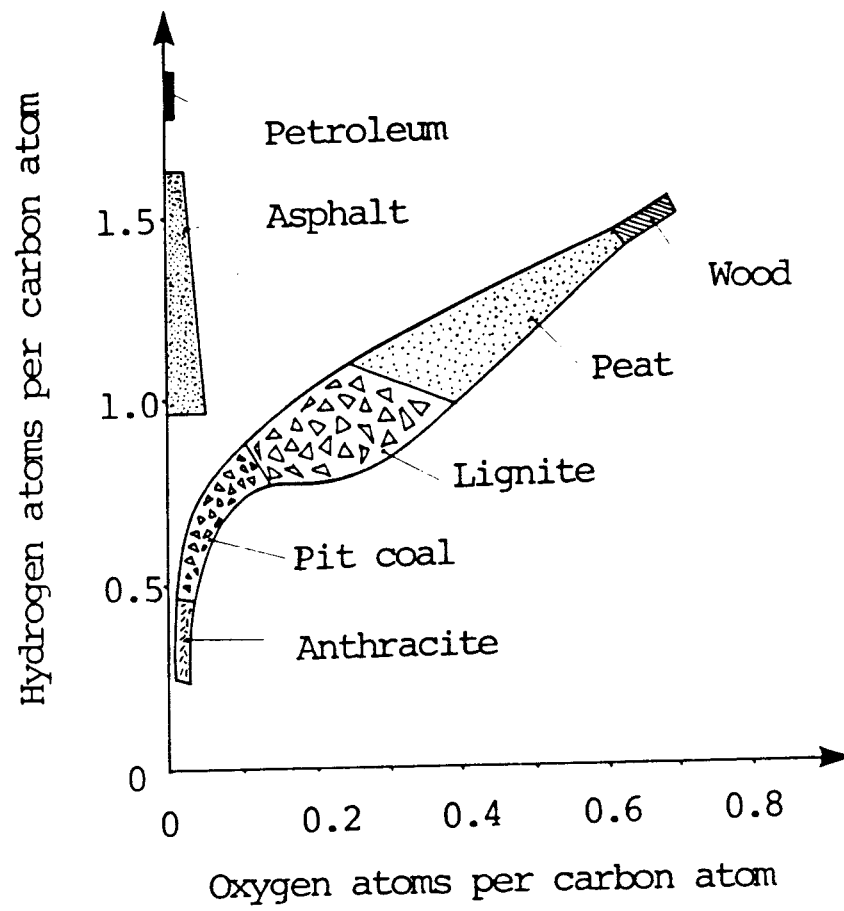
Figure 4. Chemical composition of fuels /20/

Peat is very reactive and can easily be changed
by thermochemical conversion, e.g.

- pyrolysis
- gasification
- liquefaction

to cleaner and more convenient fuels. The
purpose of this conversion is to transform peat
to liquid or gas by reduction. Direct fermenta-
tion of peat to methane has been discussed as a
possible technique.

There are, however, recent plans to exploit
about 10-20 % of the Swedish peatlands for fuel
production.

In table 2 four solid fuels are compared.

Table 2. Chemical composition and properties of
some solid fuels /20/.

		Coal	Lignite	Peat	Wood
Chemical composition:					
Carbon (C)	weight %	76–87	65–75	50–60	48–55
Hydrogen (H)	weight %	3.5–5.0	4.5–5.5	5–7	6–7
Oxygen (O)	weight %	3–11	20–30	30–40	38–43
Nitrogen (N)	weight %	0.8–1.2	1–2	0.5–2.5	<0.6
Sulfur (S)	weight %	1–3	1–3	0.1–0.4	0.02–0.06
Fuel properties:					
Volatile matter	weight %	10–50	50–60	60–70	75–85
Ash	weight %	4–10	6–10	2–15	0.1–2
Melting point of ash	°C	1100–1300	1100–1300	1100–1300	1350–1450
Bulk density	kg/m³	728–880	650–780	300–400	320–420
Eff. calorific value of dry substance	MJ/kg *)	28–33	20–24	20–23	17–20

*) 1 MJ/kg = 0.278 kWh$_{th}$/kg = 238 Kcal/kg = 2092 Btu/lbs

Some points in connection with the combustion of peat should be noted:

- emissions of sulphur dioxide are usually low, as for other types of cultivated biomass, but they can in some cases exceed the current limiting values

- the emission of nitrogen oxides from peat combustion will be of little significance

- emissions of metals are about the same size from the combustion of coal, peat and wood

- as a rule no restrictions are needed as regards organic emissions from large combustion plants

- the residues contain varying amounts of heavy metals.

As a rule it must be recommended that the peats should be analyzed with respect to pollutants, heavy metals, radionuclides etc <u>before</u> they are burned.

2.2.3.2 Peat as soil substitute or soil improvement

The use of peat for gardening or agricultural purposes has increased rapidly during the last years.

One of the reasons is that the sale of vegetable mould or soil in Sweden has been restricted for some time and the demand is increasing. A great

deal of the peat is utilized in vegetable
gardens and greenhouses.

The bulk of the material sold is the reed or
sedge type, mostly because it is suited for uses
in mixtures with soil. Sphagnum peat may also be
used but it is essential that it should be
reinforced by additions of fertilizers and
lime /5/.

Peat or muck is used in mixtures with mineral
soil for four main reasons:

-        to improve the physical conditions

-        to increase the water-holding capacity

-        as a source of humus and nutrients and
         a substitute for animal manure

-        to provide a better rooting medium.

It should be borne in mind that peat is often
deficient in plant nutrients, and the
possibility that the plants will take up soluble
pollutants in the soil-peat mixture should not
be neglected.

3   INDIRECT IMPACT

Under the heading of "indirect impact" three
phenomena are discussed in this paper. They are
all initiated by and dependent on anthropogenic
activities but their consequences are less
desirable.

3.1   Acidification

The acidification problem is too far-reaching to
be treated in detail here and it can suitably be
studied elsewhere.

Only a few important points will be mentioned
here.

-        The complexity of most soils reduces
         the possibility to of predicting their
         acidification through acid deposition
         It is known, however, that physical and
         mineral properties of soil are
         parameters involved.

-        The amount and composition of organic
         matter in soil is important as well as
         the content and the activity of
         microorganisms.

-        For several metals, increased mobility
         due to higher acidity in soil is
         accompanied by increased solubility,
         and free metal ions can be available to
         plants, at least to vascular plants.

-        Very little is known about the availabi-
         lity of metals to microorganisms in
         acid soils.

- The complex- or chelate-binding compounds become less stable at low pH-values.

- Acidifying fertilizers can be responsible for at least half of the acidifying effect in some soils /35/. Compare the pH changes in the following table.

Table 3. The pH changes in soil treated with different fertilizers and two strong acids /6/.

Fertilizer	100 kg N		200 kg N	500 kg N		1 000 kg N	ΔpH per 100 kg N
	I	II	III	I	II	III	
Calcium nitrate	+0,05	−0,04	+0,04	+0,23	±0,00	+0,10	+0,02
Calcium ammonium nitrate	+0,04	−0,14	−0,22	−0,08	−0,48	−0,50	−0,05
Urea	−0,07	−0,08	−0,24	−0,19	−0,50	−0,76	−0,08
NPK 20-5-9	±0,00	−0,10	−0,20	−0,24	−0,59	−0,96	−0,10
Ammonium sulphate	−0,13	−0,33	−0,68	−0,37	−0,97	−1,22	−0,34
Nitric acid	−0,06	−0,04	−0,12				
Sulphuric acid	−0,01	−0,09	−0,36				
Manure	+0,02	−0,04	+0,11	+0,23	+0,22	+0,33	+0,05

Period I: 6.6.72−16.8.72; Period II: 6.6.72−8.5.73; Period III: 6.6.72−30.8.73.

3.2   Leaching

Human impact influences the leaching rate in
several ways for instance through fertilization,
irrigation and the choice of crops.

During its passage through the soil the water
dissolves salts, induces ion exchanges and
hydrolytical processes and enhances chemical
weathering. Consequently the composition of the
percolating water is successively changed both
qualitatively and quantitatively.

When the amounts of rainfall or irrigation water
entering the soil become greater than its
water-holding capacity, losses by percolation
will occur. Leaching of cations added to the
soil in fertilizers is affected by the soil's
cation exchange capacity (CEC). Humic matter has
a very high CEC and peat soils tend to hold the
nutrients and prevent their leaching.

As the elements nitrogen and phosphorus mostly
accelerate the eutrophication processes in a
lake, their leaching is of special interest
/25/.

It is hardly possible to generalize the role of
chemical fertilizers in nitrate losses through
leaching. Among other things the leaching rate
of nitrogen varies with the type of crop, the
amount and time of fertilizer applied, the soil
permeability, the precipitation rate and
irrigation, the evaporation hydrology etc.

The leaching rate is also dependent on the
choice of fertilizer. Ammonium-containing
fertilizers cause a depression of soil pH-value
in accordance with the expected acidifying

effect in soil through the microbial nitrifica-
tion of ammonium.

Nitrogen losses from old, unfertilized wooded
ground are usually very low, around 1 to 2 kg
hectare^{-1} year^{-1} /39/. As the need for easily
available nitrogen is the limiting growth factor
for Swedish forests follows that the content of
ammonium and nitrate is low in run-off water.

Increased concentrations of nitrogen in surface
or groundwater can usually be explained by
fertilization, clear felling, drainage, liming
etc. Forest fertilization with ammonium nitrate
has been followed by increased concentrations of
nitrate both in streams and in groundwater /19/.

As discussed earlier, it is typical of most
peats that they contain a large amount of
nitrogen but are poor in phosphorus and potassium.
When fertilizing forests on peatland it is
important to choose a phosphorus fertilizer with
low solubility to minimize the leaching /19/.

Potassium can migrate to the groundwater more
easily than phosphorus, but negative effects of
this element are not observed.

The supply of nitrogen to peat soils, like the
supply to mineral soils, can entail leaching.
When dissolved preferably in the soil solution
the NO_3 nitrogen moves fast down to drainage or
groundwater. Likewise in peat, ammoniated
nitrogen has a great mobility. To minimize the
leaching rate of nitrogen, the type of peat and
the amount of fertilizer applied are of importance
/19/.

When 60 kg nitrogen hectare^{-1} is given, an amount now recommended, the leaching rate is negligible however.

In connection with clear-felling and nitrogen leaching /38/, one factor may be the harvesting technique. The trend is to use more and more of the biomass (removal of the whole tree including branches and stump), a technique which can be expected to have different ecological consequences.

The problems of leaching, both from arable soils and from forest soils, have been considered and discussed in several references, e.g /37, 41, 42/.

The arable land in the south of Sweden is well fertilized and the phosphorus status of the soil is also very high. Lake Ringsjön in Central Skåne is of importance both for supplying drinking water and for recreation and fishing. Today this lake is seriously eutrophicated. In /10, 26, 27/ high nutrient losses were reported. The losses of phosphorus were especially high .

Both surface and drainage waters were analyzed from different points in the catchment area. The nitrogen concentration in the surface water amounted to around 60 % of that in the drainage water. For phosphorus the case was exactly the reverse in this area, and the total phosphorus concentration was three times greater in surface water than in drainage water.

A calculation of the transport of nutrients is given in the following table.

Table 4. Transports of nutrients and eroded material with surface and drainage water to lake Östra Ringsjön (to be compared with transports calculated for arable land and forests of larger areas in Sweden) /10/.

Ground	Run off (mm)	Transport (kg·hectare^{-1}·year^{-1})				
		NO_3-N	Tot-P	PO_4-P	Part-P	Susp
Arable land surface water	60	3.4	0.16	0.09	0.06	44
Arable land drainage water	519	48	0.46	0.28	0.17	46
Forest	579	4.6	0.08	0.02	0.06	23
Arable land[*] The South of Sweden	302	39	0.37	0.19	–	–
Forest[**] The whole of Sweden	–	0.3	0.06	0.02	–	–

[*] from /9/
[**] from /4/

It is well known that the extractibility of most metal elements in soil is greatly dependent on the acidity of the system and the residence time of heavy metals in forest soils is given in reference /35/.

3.3   Soil decomposition

In some references the settling of peatsoils is discussed with regard to a lake depression in Sweden. In Kvismar Valley in Närke it is observed that not only the peat soils but also clay soil mixed up with gyttja had settled around 50 centimetres during the first 50 years after drainage. During the following 30 years the soils in this area decreased 0.7 centimetres year^{-1} on an average /15, 33/.

Three factors may be mentioned to explain the settling of soils:

-      the shrinkage of peat soils through the drawing off of groundwater
-      the shrinkage of gyttja soils and generation of permanent cracks and fissures
-      oxidation of organic matter.

Increased exploitation of peatlands initiated by drainage will rapidly oxidize the organic matter.

When plants become senescent they are invaded by microbial decomposers but the bulk of decomposition does not take place until the dead vegetation comes in contact with the soil. Once on the ground the plants are subject to attack by microbes, bacteria, yeasts, actinomycetes and fungi and the decomposition rate is accelerated.

As the bacteria and fungi work on the plant debris they assimilate the nutrients and incorporate them in living matter. At this point the nutrients are still unavailable for recycling /30/. Both bacteria and fungi are shortlived and the nutrients are released or mineralized and become available for uptake again.

In terrestrial ecosystems bacteria and fungi play the major role in decomposition /8/. Organic compounds vary greatly in their rate of decomposition. In terms of their ease of decomposition they may be listed as follows

-      sugars, starches, simple proteins
-      crude proteins
-      hemicelluloses
-      cellulose
-      lignin, waxes, fats.

The soil microorganisms must not only have nutrients for their tissue synthesis but energy as well. Organic matter contains considerable potential energy.

The more common simple decomposition products resulting from the activity of soil microorganisms may be listed as follows:

- carbon CO_2, CO_3^{2-}, HCO_3^{-}, CH_4, elemental carbon

- nitrogen NH_4^{+}, NO_2^{-}, NO_3, gaseous nitrogen

- sulphur S, H_2S, SO_3^{2-}, SO_4^{2-}, CS_2

- phosphorus $H_2PO_4^{-}$, HPO_4^{2-}

- others H_2O, O_2, H_2, H^{+}, OH^{-} K^{+}, Ca^{2+}, Mg^{2+} etc.

A lesser amount of carbondioxide reacts in soil and produces i a carbonates and bicarbonates of calcium, potassium and magnesium. These compounds are readily soluble and may be used by higher plants or lost by drainage.

Proteins are split up into aminoacids and by means of enzymic hydrolysis they yield ammonium compounds. A large number of heterotrophic organisms are involved in these reactions.

Under aerobic conditions ammonium is enzymically oxidized to the nitrate form. The organisms in this case are all specialized and autotrophic, i.e. they obtain their energy from the oxidation of mineral constituents. Nitrates are assimilated

by plants and are also subject to leaching
losses. Hydrogen ions are obtained through the
oxidation of ammonium ions.

In poorly drained and in acid soils nitrates and
nitrites may be reduced by chemical as well as
biochemical reactions and free nitrogen or
oxides of nitrogen may be evolved.

Many organic compounds contain sulphur and
in several cases organic subsoils contain both
inorganic and organic sulphur (gyttja). In /33/
the consequences of high sulphur content in soil
are discussed and the chemical reactions between
different compounds are elucidated. By draught
and drainage the oxidation of sulphur will
dominate. It may be oxidized to SO_4^{2-} and the
soils will be acidified.

Both inorganic and organic forms of phosphorus
occur in soils and both are important to plants
as sources of this element. The kind of phosphate
present in soil varies with the pH in the soil
solution. In alkaline soil the HPO_4^{2-} is apparently
most common and at pH-values around 4-5 $H_2PO_4^-$
tend to prevail.

In general $H_2PO_4^-$ is considered more available
to plants than other P-ions. Organic phosphorus
compounds are mineralized by microbial activity
to $H_2PO_4^-$ and HPO_4^{2-}. Small amounts of
phosphorus in complex mineral combinations are
usually very slowly available.

4   CONCLUSIONS

The effects of human impact can as well
accelerate as counteract the development of a
lake or a bay, but most influences increase the
ageing rate during the different stages, for
instance:

-       During the eutrophication: Fertilization
        of arable land in the catchment area
        may lead to nutrient losses, above all
        of phosphorus and nitrogen, with
        increased eutrophication rate as
        consequence.

-       Clear felling of forests combined with
        fertilization and drainage may give
        rapid release of nutrients to
        surrounding lakes.

-       Fertilizers containing ammonium can
        be enzymically oxidized under aerobic
        conditions to the nitrate form, leading
        to a decreasing soil pH with increased
        solubility and mobility of cations.

-       Outlets of polluted waste water from
        industries and households may accelerate
        the ageing rate of the recipient.

-       Aquaculture may result in a release of
        phosphorus and other pollutants in
        inland waters while nitrogen is most
        important as a pollutant in coastal
        areas.

-       Ditching of surrounding areas may lead
        to changes in the water balance and the
        water chemistry in the recipient.

-       In association with the terrestrializa-
        tion: Drainage of a lake to get new
        arable soils leads to oxidation in the
        upper soil layers. Organic and inorganic
        sulphur may be oxidized to SO_4^{2-} and
        the soil will be acidified. The
        solubility and mobility of heavy metals
        and radionuclides can increase.

-       More and more aerobic conditions in the
        upper layers of the soil lead to
        enzymic oxidation of nitrogen compounds
        to nitrate and a lower pH as a consequence.

The release and transport of pollutants from
sediment and organic soils may be influenced by
for instance the following factors:

- Oxidation processes and decomposition
  in organic soils due to drainage and
  cultivation.

- Uptake of metal ions via roots in
  connection with crop production and
  grazing.

- Lowering of pH in soils due to acid
  rain may change the solubility and
  perhaps also the plant availability of
  metals.

- Use of peats in energy production may
  possibly lead to dispersion of
  pollutants in the environment.

To sum up: the effects of human impact which are
estimated to be of predominant importance
in connection with the eutrophication of a lake
or a bay seem to be associated with acidification
and leaching processes. After the terrestrializa-
tion of a wetland area the consequences of
oxidation and biochemical decomposition of soil
substances due to cultivation may be important.

The effects mentioned above are examples of
impacts that are common or may be expected
today. Other human activities in the future may
naturally lead to hitherto unknown consequences.

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