COMMISSIE VOOR HYDROLOGISCH ONDERZOEK TNO
COMMITTEE FOR HYDROLOGICAL RESEARCH TNO

VERSLAGEN EN MEDEDELINGEN No. 25
PROCEEDINGS AND INFORMATIONS No. 25

THE RELATION BETWEEN WATER QUANTITY AND WATER QUALITY IN STUDIES OF SURFACE WATERS

TNO

THE RELATION BETWEEN WATER QUANTITY AND WATER QUALITY IN STUDIES OF SURFACE WATERS

COMMISSIE VOOR HYDROLOGISCH ONDERZOEK TNO
COMMITTEE FOR HYDROLOGICAL RESEARCH TNO

Verslagen en Mededelingen No. 25
Proceedings and Informations No. 25

THE RELATION BETWEEN WATER QUANTITY AND WATER QUALITY IN STUDIES OF SURFACE WATERS

PROCEEDING OF
TECHNICAL MEETING 35
(October 1978)

VERSL. MEDED. COMM. HYDROL. ONDERZ. TNO 25 – THE HAGUE – 1979

CONTENTS

AUTHORS

Dr. P.E. RIJTEMA	Institute for Land and Water Management Research, Wageningen
Dr. L. LIJKLEMA	Twente University of Technology, Enschede
Ir. J.H.A.M. STEENVOORDEN/ Ing. H.P. OOSTEROM	Institute for Land and Water Management Research, Wageningen
Ir. G. VAN STRATEN	Twente University of Technology, Enschede
Drs. G. VAN WIRDUM	Research Institute for Nature Management, Leersum
Ir. B.A. BANNINK	RWS – Delta Department, Environmental Division, Middelburg

GENERAL SURVEY OF THE RELATION BETWEEN WATER QUANTITY AND WATER QUALITY

P.E. RIJTEMA

SUMMARY

Problems related to quantitative and qualitative water management generally show a strong relationship. The water quality in a river is not only determined by the quantity of pollutants dumped into the river, but also by the river discharge.

Particularly in coastal areas and river deltas with salinity problems, the mutual relationship is very strong. The more so, when the delta forming rivers are considerably polluted.

The Netherlands are an example of such a situation, where river waters are used to control the quantitative and qualitative water management in 51% of the area of the country. The effect of the river water on the water management is demonstrated with a number of data.

Finally a brief review is given of the processes in surface waters and the consequences for water management problems.

1. INTRODUCTION

It is common in quantitative water management to think in terms of volumes and fluxes. In qualitative water management it is use to define the standards for water quality for various purposes in terms of maximum acceptable concentrations. The problems in quantitative and qualitative water management show nevertheless a strong interrelationship. The quality of surface waters is not only determined by the type and quality of the waste water discharge in the watercourse, but it is also dependent on the mass of water flowing through the tributary.

Both aspects show particularly strong interrelationships in coastal and deltaic areas when also salinization takes place. This becomes even more important, when the delta-forming rivers are considerably loaded with pollutants. The Netherlands are a perfect example of such a situation.

The water manager very often has to face quantitative water problems in order to reach certain criteria for water quality. Moreover, it must be realized that the quality of the inlet water for quantitative purposes ultimately decides on the water quality in the region. It must be realized that in national water management both water quality standards and quantitative water requirements of the various regions determine the final distribution of the water supply in the various regions.

In an introductory survey on the relation between water quantity and water quality it is good to discuss briefly, as an example, the situation in The Netherlands. It is necessary in such a discussion to differentiate in regions with and without external water supply.

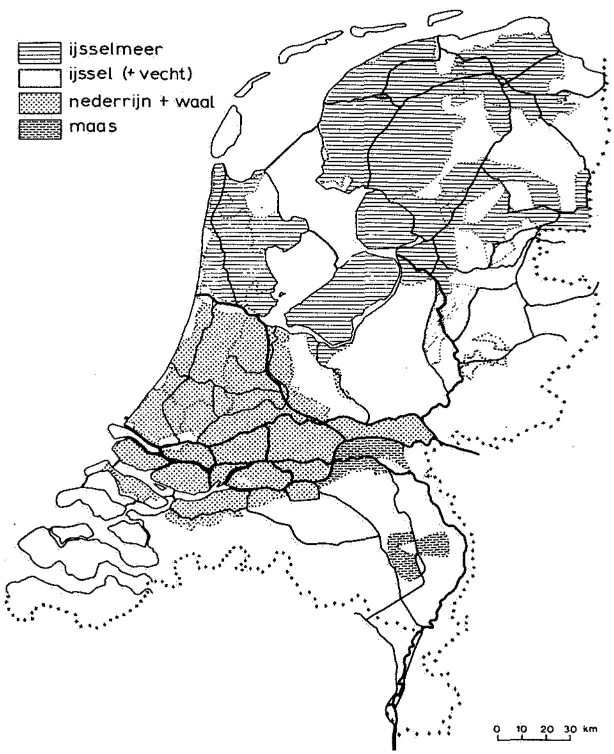


Fig. 1 Regions with external water supply classified by the origin of the inlet water

2. REGIONS WITH EXTERNAL WATER SUPPLY

The discharges from the rivers Rhine and Meuse are the main sources for the additional water supply to the surface waters in The Netherlands. The river waters are transported to the different regions of The Netherlands by the following main surface waters and their branches as IJssel, Lek, Waal, Hollands Diep and Haringvliet. The water intake in a number of regions occurs directly from this main system. For regions which cannot draw directly from the main system, the water is transported from the main system through canals to the region concerned. Fig. 1 gives the regions with external water supply, differentiated according to the origin of the main supply system. It appears that 51% of The Netherlands must be considered to have an external water supply. A quantitative indication of the areas with external water supply and the origin of the supply water is presented in table 1. From this it can be seen that only 4% originates from the river Meuse, while the main source is directly or indirectly the river Rhine.

An analysis of the distribution of the external water supply in The Netherlands in the dry years before 1970 has been made by Van den Berg (1970). In that study a com-

Table 1 Percentage of surface area with water inlet and the origin of the supply water (Van Boheemen, 1977)

Lake IJssel	57%
IJssel	4%
Lek, Waal, Hollands Diep, Haringvliet	35%
Maas	4%

parison is given of the actual quantities with the calculated data of the water supply requirements in a dry year with a 5% probability of exceedence (Rijkswaterstaat, 1968). Van Boheemen (1977) gives data of the external water supply in the dry year 1976. The results of the various investigations are given in table 2. The data show marked differences between the figures calculated by Rijkswaterstaat and those realized in dry years. Particularly the fresh water supply in the regions with a higher elevation is considerably less than the calculated data.

The data show, however, that The Netherlands are a typical example of a delta area, where the rivers dominantly determine the quality of the surface waters in an important part of the country. In the Northwest region the quantity of water supply during the summer is calculated by Rijkswaterstaat as $510.10^6 m^3$ and realized in 1976 was $386.10^6 m^3$. With a total area of surface waters in this region of 8600 ha, with a mean depth of 1 m, it means that the calculated and realized figures equal respectively 6 and

Table 2 Water requirements calculated by Rijkswaterstaat (1968), water requirement data given by Van den Berg (1970) and by Van Boheemen (1977) in $10^6 m^3$; W = water supply, F = flushing

	Rijkswaterstaat 1968		Van den Berg 1970		Van Boheemen 1977	
	W	F	W	F	W	F
Polder areas						
North-Netherlands	1000	250	500	50	444	222
Northwest-Netherlands	260	250	200	50	156	230
Midwest-Netherlands	375	250	300	140	384	128
Southwest-Netherlands	165	15	–	–	5	7
River district	355	0	250	0	308	0
Sub total	2155	765	1250	240	1297	587
Areas with higher elevation						
Northeast-Netherlands	500	0	–	0	153	0
Southeast-Netherlands	565	0	–	0	21	0
Zeeuws-Vlaanderen	50	0	–	0	0	0
Sub total	1115	0	–	0	174	0
Total	3270	765	–	240	1471	587

Table 3 Mean summer values of some quality data of inlet water and at the end of the transport path in Northwest-Netherlands

Parameters	Mean summer figures in $g.m^{-3}$			
	Cl^-	Kjeld.-N	Tot.-N	Tot.-P
Inlet water (Lake IJssel)	282	1.05	1.56	0.13
End transport path	346	3.10	4.34	1.14

4.5 times the total water volume in the region. The amount of water used for flushing during the summer was 3 times the total water body.

The surface in the region is polluted by internal sources, so its quality decreases from the intake point at Lake IJssel to the outlet point. Table 3 gives some quality data at the intake point and the corresponding figures at the end of the transport path. It appears from this table the Cl^--concentration of the water of Lake IJssel dominantly determines the quality level in the region, whereas the N- en P-concentrations are strongly determined by the internal pollution in the region.

Investigations from the Working Party (1976) for the midwestern region of The Netherlands, show that the replacement of the water body in Rijnland is in the order of 8.8 times a year and in Delfland, with the main centre for greenhouse horticulture in The Netherlands, it is 54 times a year. Calculations from the Working Party indicate that in summertime the Cl^--concentration for about 70% is determined by the quality of the discharge of the river Rhine. Here again, the situation for the N- and P-concentrations is less simple. The influence of the river Rhine seems to be less pronounced, mainly due to internal pollution and processes during the transport path.

Data from Bots, Jansen and Noordewier (1978) concerning the ion-composition in the northern region indicate that the main part of the surface waters shows marked differences between summer and winter. During summer, with water intake from Lake IJssel, a strong increase occurs of surface waters with an ion-composition which can be characterized as a $NaCl$-type of water. This increase of $NaCl$-type waters coincides with the decrease of the $Ca(HCO_3)_2$-type of water, which in large areas in this region is normal during wintertime. Waters with sulphates as dominant salts in winter, disappear completely during summer and are replaced by $NaCl$-type water. The intake of Lake IJssel water has for a small number of surface waters the consequence that during summer a $CaCl_2$-type of water is present. These changes in ion-composition during summer might have consequences from the viewpoint of nature conservancy.

The examples given demonstrate that the quality and watertype of the surface waters in an important part of The Netherlands are determined by the quality of the discharge from the river Rhine. Reduction of the pollution load in the Rhine basin will result in an important improvement of the water quality of the surface waters in about 50% of The Netherlands.

3. REGIONS WITHOUT EXTERNAL WATER SUPPLY

About 95% of the sandy soils in The Netherlands belong to the area without external surface water supply. Moreover, no external supply is present in the loess-loam soils in southern Limburg and in the clay-soils in the Southwest of The Netherlands, the Wieringermeer and the northern part of Groningen.

The problems with the management of water quality and quantity for these regions can be best demonstrated for the sandy soils. The enormous extraction of groundwater for domestic and industrial water supply apart from river training, improvement of land drainage and level control by weirs in the water courses, has changed the natural dischange pattern in these regions. It was shown by Rijtema (1974) for areas with groundwater extraction in eastern-Gelderland, that 50% of the precipitation surplus in the intake area of the pumping stations is withdrawn, resulting in a considerable reduction of the discharge via groundwater flow to the surface waters.

The main part of this groundwater, however, returns after domestic or industrial use to the surface waters, in the form of effluents of sewage treatment plants or of a direct discharge of waste water. It is clear that the water coming from these point sources has a poor quality compared with the original groundwater, resulting in a reduction of the water quality in the tributaries. Data of Steenvoorden (1978) for instance, show that in area of the Barneveldse Brook, in dry periods in 1977 in some parts of the catchment area more than 70% of the total discharge originates from the effluents of sewage treatment plants. The effect of the effluents on water quality is under these conditions predominant. Periods of drought are extremely important for the determination of the water quality standards for the effluents of sewage treatment plants. The water manager must realize that under these conditions dilution is not a solution for pollution.

4. RELATIONSHIP BETWEEN WATER QUALITY AND WATER QUANTITY

The water quality discharged from a tributary is principally determined by the quality and quantity of the inflow and the quantity of pollutants brought into the system.

The transport of pollutants from a point source occurs by the displacement of the medium, in this case water, in which the pollutant is present. Due to the flux distribution in the tributary and dispersion and diffusion, the pollutant is not displaced like a fixed body, but the concentration decreases with increasing distance from the source. Moreover, the pollutant can be subjected to a number of interaction processes, i.e. processes in the surface water by which solved material disappears from the system, for example by precipitation, adsorption or biological degradation or enters the water body by resolution from the bottom sludge.

For quantification of these water quality aspects research must be directed towards correct physical descriptions of the processes, resulting in physical-mathematical models which can be used for forecasting the effects of certain measures in water quality management. Theoretical aspects of such an approach are given by Lijklema (1979).

For the analysis and quantification of processes affecting the surface water quality it is necessary to formulate water and matter balances. For these balances a distinction can be made between external and internal sources. The external sources can be, for instance, quality and quantity of external supply from the river Rhine and Lake IJssel, inflow from the North Sea by leakage of sluices. The internal sources in the catchment area can originate for example from seepage, natural gas-sources, sources for cooling water, domestic and industrial waste water, fertilizers, precipitation, the use of de-icing chemicals on roads and pollution by recreation. The analysis of these sources is not only a basis for the determination of measures to reduce directly the effect of the main sources, but it yields also important information for the manager to decide on his flushing policy to reduce their disadvantageous effects. Data of Steenvoorden and Oosterom (1979) give examples of water and matter balances for polders and catchment areas with special reference to the N- and P-load.

In the analysis of the consequences of water quality and quantity management one has to deal, apart from transport of matter, with aerobic and anaerobic biological degradation of organic matter, mineralization of fertilizers, N-uptake by organisms, growth and dying-off of organisms. These processes determine to a large extent to the oxygen-management and the growth of algae in surface waters. A further quantification of these processes is given by Van Straten (1979).

Variation in eutrofication of surface waters by changing the phosphate and nitrogen load and possibly the ion-composition when external water supply in a region is present, may affect growth, development and composition of plant associations in surface waters. The influence of trofic gradients on plant associations is discussed by Van Wirdum (1979).

The execution of civil and hydraulic works in the delta area has important consequences for the water management in the Southwest of The Netherlands. A number of basins in this region are closed and have no longer a direct connection with the North Sea. The changes in water composition and the ceasing or reduction of the tidal movements strongly affects the natural conditions in the surface waters. The consequences of this closure and the resulting problems in the management of these basins are discussed by Bannink (1979).

REFERENCES

BANNINK, B.A. (1979) – The relation between water quality and water quantity in the changing delta area of Southwest-Netherlands. Comm. Hydrol. Onderzoek TNO, Den Haag. Versl. en Meded. 25.

BERG, C. VAN DEN (1970) – The water requirement of agriculture and horticulture. De Ingenieur 82, 17 (in Dutch).

BOHEEMEN, P.J.M. (1977) – Water consumption by agriculture and horticulture in the dry summer of 1976. Nota 975, Institute for Land and Water Management Research, Wageningen. 82 pp (in Dutch).

BOTS, W.C.P.M., P.C. JANSEN and G.J. NOORDEWIER (1978) – Physical-chemical composition of surface water and groundwater in the northern Netherlands. Regionale Studies 13, Institute for Land and Water Management Research, Wageningen. 111 pp (in Dutch).

LIJKLEMA, L. (1979) – Theoretical aspects of the relation between water quantity and water quality of surface waters. Comm. Hydrol. Onderzoek TNO, Den Haag. Versl. en Meded. 25.

RIJKSWATERSTAAT (1968) – The water management of the Netherlands (in Dutch).

RIJTEMA, P.E. (1974) – The agricultural consequences of pumping drinking water in East Gelderland. Nota 696, Institute for Land and Water Management Research, Wageningen (in Dutch).

STEENVOORDEN, J.H.A.M. (1978) – Phosphate and nitrogen balance for the surface water in the catchment of the Barneveldse Beek. Nota 1080, Institute for Land and Water Management Research, Wageningen. 24 pp (in Dutch).

STEENVOORDEN, J.H.A.M. and H.P. OOSTEROM (1979) – Natural and artificial sources of nitrogen and phosphate pollution of surface waters in The Netherlands. Comm. Hydrol. Onderzoek TNO, Den Haag. Versl. en Meded. 25.

STRATEN, G. VAN (1979) – Hydrological aspects of oxygen behaviour and algal blooms in river systems. Comm. Hydrol. Onderzoek TNO, Den Haag. Versl. en Meded. 25.

WIRDUM, G. VAN (1979) – Dynamic aspects of trophic gradients in a mire complex. Comm. Hydrol. Onderzoek TNO, Den Haag. Versl. en Meded. 25.

WORKING PARTY MIDDEN WEST-NEDERLAND (1976) – Hydrology and water quality of the central West Netherlands. Regionale Studies 9. Institute for Land and Water Management Research, Wageningen. 101 pp (in Dutch).

THEORETICAL ASPECTS OF THE RELATION BETWEEN WATER QUANTITY AND WATER QUALITY OF SURFACE WATERS

L. LIJKLEMA

SUMMARY

Whereas the influence of water quality upon the hydraulics of a water system is restricted to density currents and stratification induced by salt and (or) temperature variations, the water quality is generally controlled largely by the water quantity. This coupling is twofold; direct by the loading of the system associated with the water supply and indirect by the influence of flow on detention time, mixing and reaction rates. The influence of different types of loading (constant, impulse, sinus) on systems with different mixing properties (completely mixed, plug flow, plug flow with axial dispersion) is illustrated by three examples.

A short discussion of the effect of flow rate and water depth upon processes such as reaeration, dispersion, algal growth, BOD-degradation, sedimentation and resuspension is presented.

1. INTRODUCTION

It is customary in theoretical treatises to define the terms used. However, the term water quality apparently has no fixed meaning but the sense varies with the use or function of the water involved. Hence there is a reciprocal relation between water use and water quality: the quality controls the use but the actual use or the anticipated use defines the terms in which the water quality should be expressed. For cooling water, recreational water, process water, water for irrigation or as a source for drinking water, the accent in defining the water quality shifts among several physical, chemical and biological characteristics of the water. Water standards vary with water use. In fact the term water quality is a rather general indication of a complex of properties related to the suitability of the water for the use under consideration. Nevertheless certain aspects have a rather general significance for the examination of the usefulness of surface waters. The oxygen concentration is of prominent importance, not only in itself for fishlife etc. but also indirectly as an indication for the absence of major organic pollution, disease-germs, botulism, etc.

The most important quality aspects for surface waters are listed in table 1.

In this discussion the term water quantity can be interpreted as either the quantity present (volume, level) or the quantity transported per unit of time (flow, velocity).

Next it is meaningful to notice that the relation between water quantity and water quality is mainly unilateral: quality is nearly always influenced by quantity but the reverse is less frequent. Major examples of the influence of water quality on quantitative

Table 1 Major water quality aspects

Physical aspects	*Chemical aspects*	*Biological aspects*
Colour	Oxygen	Coliforms
Temperature	BOD, COD, TOC	Fecal bact.
Secchi disk depth	nutrients, N, P, Si	Virusses
Taste/odor	heavy metals	Chlorophyl
Radio-activity	organic toxicants	Diversity-index.
	phenols, oil	
	hardness	
	detergents	
	pH; pE	

aspects are the occurence of density currents caused by gradients in salt concentration or in temperature; therefore temperature differences are the main cause of a coupling quality → quantity is fresh waters. The reduction of the hydraulic detention time of inflowing, warm water in a stratified lake is an example.

As a consequence it is generally possible to achieve an independent description of the water movement in a system and use the output of such a hydrodynamic model after that as the input into a description of the quality. However, if the quality influences the quantity both descriptions should be coupled. In mathematical models the differential equations for motion, diffusion, reaction etc. must be solved simultaneously in such situations. This is exemplified in thermal models and models describing the salt distribution in estuaries.

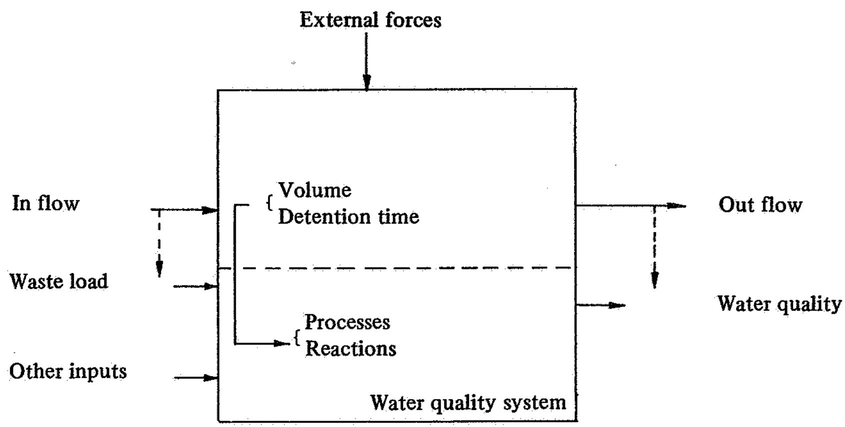


Fig. 1 Block diagram of a water quality system.

Figure 1 is a very elementary representation of the relations in any water system. The water supply by rain, groundwater flow, sewerage systems, pumps, rivers and brooks or runoff is coupled with a loading of the water system by the materials suspended or dissolved in such water. Next to the loading through water also a direct discharge of polluting materials can occur. Release of certain compounds by sediments can also be considered as an input of pollutants if the sediments are not included in the water system.

The hydraulic detention time, volume, water mark and mixing in a water body are controlled by the inflow and outflow of water and often also by external forces, especially wind. The organisms and chemicals in the water are subject to all sorts of physical, chemical and biological transformations, which processes are controlled in a large measure by the degree of mixing and the residence time in the water body and also by external forces (wind, irradiance). Hence the water within the system or leaving the system will have a quality that depends on the quantity in two ways:

a) directly through the load of pollutants associated with the inflow;
b) indirectly through the residence time distribution, mixing, water level and velocity-field induced by the inflow and the influence of such variables upon rates of transformations.

The dynamic character of the inputs, both inflow and loading, and also the variable meteorological conditions generally cause a more or less variable water quality. The question whether in an analysis of water quality and water quantity problems a more detailed, complex description of the system behaviour including the dynamics should be used or a more elementary, stationary approach based on average values or trends depends on several conditions such as the availability of data, the objectives of the study and the accuracy required, the time available for the study and the importance of the decisions to be made. It can be useful to combine both approaches and allow a dynamic description for parts of the system only.

The elements of the scheme in figure 1 will be discussed in some detail in the following.

2. SOURCES

An investigation in the water quality of a system always involves the registration of the sources feeding the system. An inventory of all inflows of water and all loadings of the materials under consideration will give a first indication of the importance of separate contributions.

For systems characterised by long hydraulic detention times the relative contribution of precipitation, evaporation and occasionally seepage to the water balance may be great. For instance in some parts of the peat marshes in the lake district in the North-West of the province Overijssel in The Netherlands the water balance during the summer is dominated by evaporation and seepage. However, usually the water balance is controlled by the discharge of rivers and brooks, drainage of polders and during wet periods also by runoff.

The accuracy with which water and material balances should be known depends upon the quality aspect under consideration and the desired précision of the conclusions drawn. For example: an accurate description of the phytoplankton dynamics in the IJsselmeer (a lake with a hydraulic detention time of more than six month) does not require a detailed water balance nor accurate data on the time varying input of nutrients. In this system the internal cycling of phosphates and nitrogen and the light conditions control the growth of algae. However, for systems with shorter hydraulic detention times the external influences gain in importance. As a rule of thumb it can be laid down that an accurate knowledge of the water balance is required if the hydraulic detention time is equal to or smaller than the time-constant of the slowest process studied. The principle also may serve for guidance in the averaging of time varying discharges. The admissibility of hourly, daily, weekly or monthly averages as input data depends upon the rates of the processes under consideration. The high costs associated with most inventories of inputs and outputs of water and pollutants and the time consuming character of such activities justify a thorough consideration of the conditions to be made on data acquisition. Often agencies collect numerous data without sufficient reflection on the objective(s) and the use that should be made of the information. Such data may prove useless or insufficient for an analysis of the processes controlling the water quality. Partly this may also be true when an unbalance exists between data on inputs and discharges on one side and data on the resulting quality in the water body on the other hand. The feasibility of system identification on the basis of input-output analysis is challenged in such situations. It can be concluded that the location, the number and the frequency of sampling must be evaluated carefully.

Water quality management often focusses on extreme meteorological conditions. Droughts causing reduced dilution of discharged pollutants may attract attention as well as the high loadings associated with heavy rainfall. Statistical processing of meteorological and hydrological data can improve the insight in the probability of such events. Information on the variability of the concentration of pollutants is generally less available than information on the variability of flow. In rivers draining virgin areas the concentration of suspended materials tends to increase with increasing flow, whereas the concentration of dissolved materials decreases. Hence the load of suspended materials increases more than proportional with the discharge and the load of dissolved substances less than proportional to the discharge of water (Meybeck, 1976).

Hydrologists have accumulated useful information on the relation between rainfall and runoff in urban and rural areas and the role of infiltration, storage, slope etc. on the resulting transformations and time shifts of rainfall traces. Human interferences as improving the drainage of agricultural fields and the increase in the percentage imperviousness in urban areas have affected the hydrologic cycle. Not only the rate of discharge of water is increased by such activities, but also the concentration of dissolved and suspended matter is affected.

The day-night variation in dry weather flow of combined sewers may be about a factor three, however, due to variations in concentrations the load of BOD, Kjeldahl nitrogen etc. can exhibit a ten-fold increase during the day with respect to the night. In sewage treatment plants such differences tend to be levelled but the occurence of overflows can cause strong fluctuations in the loading of receiving waters. The pollutional load of such overflows depends on the sewerage system, the spacing of storms, duration and intensity of the rainfall etc. Figure 2 presents an example of such relations as observed in a combined sewer system. The initial stage of a high intensity rainfall after a long dry weather period causes the highest loading through overflows.

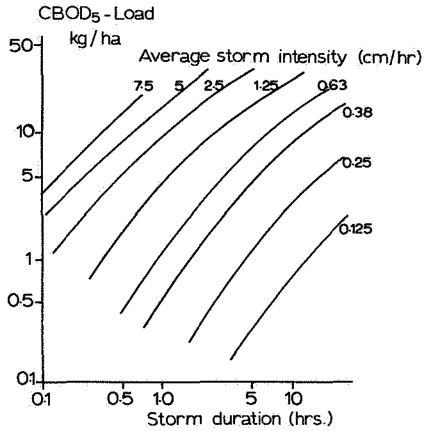


Fig. 2 Biochemical oxygen demanding loading from combined overflow as a function of storm duration and intensity.
Wingehocking sewer, North Philadelphia; drainage 2200 ha; % runoff/rainfall, approximately 20%. (Source: R.V. Thomann; Systems analysis and water quality management, New York, 1972.)

After completion of the construction of the traditional biological waste water treatment plants in The Netherlands in the forthcoming years the remaining loading of surface waters with BOD and nutrients will be due to a considerable extent to overflows. Hence it is important to study the relationship between water quantity and water quality in this respect. The research organization of the joint Water Boards (STORA) sponsors such a research project. The administration will try to select the most effective alternative in the pollution abatement program.

Comparison of the effects of overflows, diffuse sources and the remaining pollution

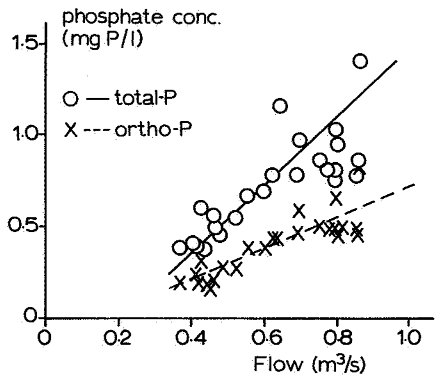


Fig. 3 **Relation phosphate concentration – flow in Kleine Valkse Beek; November 1977.**

from treatment plants will be important for the justification of the selection of alternatives as improvement of the sewerage system, construction of storage capacity, tertiary treatment and other measures. Quantitative information on the cause and effect relations between overflow and water quality is required for a balanced trade-off. The stochastic character of overflow events and the variability of time scales in processes affecting the water quality complicate the analysis.

In agricultural areas the concentration of pollutants is related to the discharged quantities of water as well as in urban areas. Increasing discharge does not result necessarily in dilution of pollutants. Research of the nutrient loading in the Barneveld Brook area (a district characterised by intensive bio-industry) during a very wet period (Beunders, 1978) revealed a positive correlation between flow and both ortho-phosphate and total-phosphate concentration (fig. 3). The increasing difference between total-P and ortho-P with increasing flow can be explained by overland flow of particulate-P during intensive rainfall. Steenvoorden and Oosterom in their review on nutrient balances of polders and catchment areas of brooks suggest resuspension of sediments as an explanation. At this same sampling station, situated downstreams of a rural area, no relation between nitrogen concentration and flow has been observed. This may be due to the fact that ammonia and nitrate percolate through soils in contrast to phosphate, which ion is absorbed strongly on clays and ironoxides. Hence for nitrogen there will be no major difference in concentration between subsurface or overland flow, at least if no major reactions such as denitrification occur in the soil.

In general the relation between flow and concentration for compounds loading surface waters through distributed sources will depend on physical-chemical properties such as solubility, adsorption, degradability etc. In order to attain an estimate of loading from an area it may be useful to classify compounds in categories with similar properties.

These general remarks on the relations between quantity and quality of sources should be concluded with the observation that downstreams in a river a time-lag may arise between the flow and the expected concentration as based upon correlations between both. This is due to the higher speed of discharge-waves with respect to the average water velocity.

In summary: the loading of surface waters with pollutants depends strongly on the discharged quantity of water and the relevancy of their variation in time depends on the time constants of the processes in the receiving water body in which these pollutants are involved.

3. PROCESSES

It is useful to start this section with a description of the sense of some terms bearing on processes affecting the water quality of a system.

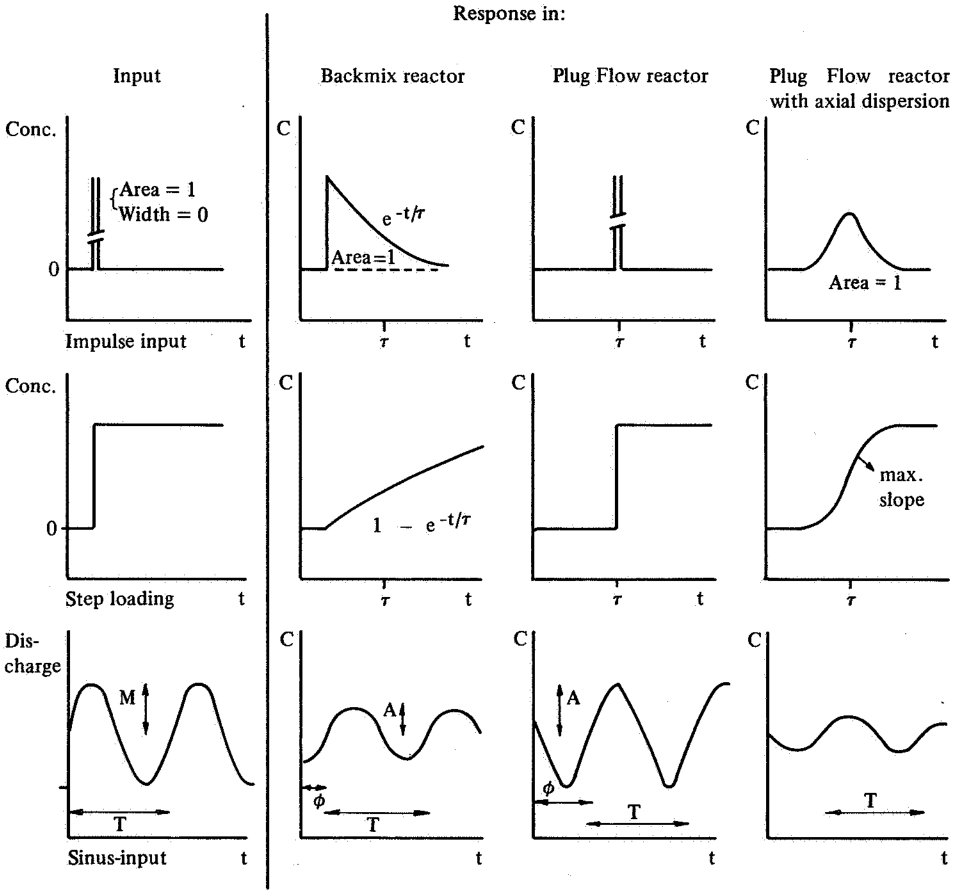
- Conservative materials are compounds which are not subject to degradation, sedimentation, adsorption, evaporation etc. Hence their concentration is affected by transport and mixing only.
- Coupled systems refer to transformations in which a process is forced by the output of a preceding reaction. It is customary to speak of coupled systems although actually the variables are coupled. An example is the BOD-O_2 system in which the oxygen concentration is a function of the BOD distribution in the water body.
- Steady state conditions as opposed to dynamic conditions refer to situations with time independent inputs (forcing functions) and rate constants (parameters), resulting in stationary values for the state variables.
- Further only linear systems will be considered, which means that the effects of separate influences upon one variable are independent; hence the response can be evaluated by addition.

The first process affecting the water quality of water bodies subject to loading of pollutants that should be discussed is the physical process of mixing. For a theoretical description it is appropriate to apply the distinction between completely mixed reactors and plug flow reactors as used by chemical engineers also for water systems. The mixing properties of brooks, rivers and canals can be approximated in the first instance by a description as plug flow reactor. Such a flow is characterised by the absence of differences in velocity for any two elements of flowing fluid in the direction of the main flow; the material marches through the reactor in single file, no diffusion occurs. At the opposite of the plug flow reactor is the total backmix reactor or ideal mixed reactor in which the contents are well stirred and uniform in composition throughout. Sometimes such a description can be applied with success to lakes or reservoirs. However, in reality most situations are best represented by some intermediate form between these idealised types. Although in such systems gradients in concentrations are smoothed to a certain extent, still appreciable differences in concentration in one or more dimensions will persist. Estuaries are an example.

In the chemical reaction engineering literature such systems are often described as plug flow with (axial) dispersion, but also several other representations (e.g. cascade of backmix reactors) can be useful (Levenspiel, 1967).

Figure 4 presents the mixing properties for three input signals. The presented input-output relations apply to conservative materials such as chloride. Certain dyes like rhodamine which are used extensively as a tracer in field studies of transport and mixing are also practically conservative. In case of occurring reactions or other transformations affecting the concentrations, the concentration-time curves will change with a tendency towards lower concentrations as time increases and at higher reaction rates.

For each of the three theoretical reactor models mentioned the response of the system to one type of input will be discussed as an illustration for the relation quality-quantity



$\mathrm{tg}\,\phi = \omega\tau$ $\qquad$ $\phi = \omega\tau$

$$A = \frac{M/V}{\sqrt{\omega^2 + (1/\tau)^2}} \qquad A = \frac{M}{Q}$$

Fig. 4 Mixing behaviour of different types of reactors.

in systems with similar mixing properties. In order to increase the insight simple situations for which analytical solutions to the differential equations are available have been selected. Reactions are assumed to be first order, which is an appropriate approximation for many practical situations.

Example 1 Plug flow reactor; coupled system; dynamic conditions; point source at $x = 0$.

A mass balance for a small elemental slice perpendicular to the flow direction in a reactor with uniform cross section and loaded with $M(t)$ $kg \cdot hr^{-1}$ of substance s_1 at $x = 0$ results in

$$\frac{\delta s_1}{\delta t} = -u\frac{\delta s_1}{\delta x} - k_{11}\, s_1 \tag{1a}$$

The differential equation for substance s_2, formed by degradation of s_1 can be written as:

$$\frac{\delta s_2}{\delta t} = -u\frac{\delta s_2}{\delta x} - k_{22} s_2 + k_{12}\, s_1 \tag{2a}$$

In these equations the symbols s_1 and s_2 represent concentrations of the coupled variables, u is the velocity of the water in the reactor and k_{11}, k_{12} and k_{22} are first order reaction rate constants. The decay rate $k_{11}\, s_1$ of the first variable s_1 does not result necessarily in the formation of an equivalent amount of s_2, because k_{11} may include more processes of which only one results in the formation of the second variable. Hence $k_{12}\, s_1$ instead of $k_{11}\, s_1$ is used in equation (2a). However, in many situations $k_{11} = k_{12}$. Under consideration of the boundary conditions the solutions are:

$$s_1\,(x, t) = \frac{M(t - t^*)}{Q} \exp\left(\frac{-k_{11} x}{u}\right) \tag{1b}$$

and

$$s_2\,(x, t) = \frac{M(t - t^*)}{Q}\, \frac{k_{12}}{k_{22} - k_{11}} \left[\exp\left(\frac{-k_{11} x}{u}\right) - \exp\left(\frac{-k_{22} x}{u}\right)\right] + S_{o,2} \exp\left(\frac{-k_{22} x}{u}\right) \tag{2b}$$

in which $S_{o,2}$ is the concentration of s_2 at $x = 0$ if present and Q is the volumetric flow through the reactor.

Equations (1b) and (2b) show that the effect of the discharge $M(t - t^*)$ at a distance $x = ut^*$ downstreams of the outfall is controlled by the dilution (Q) and an exponential factor expressing the attenuation of the signal by the reactions. The time of travel t^* characterizes which preceding discharge $M(t - t^*)$ affects the concentration at location x and time t.

The phase shift, characterizing the lag between input and output resulting from periodic inputs, is $\phi = \omega t^*$ in which ω is the frequency of the forcing function. The amplitude characteristic, describing the attenuation of the input, is a function of the system parameters but independent of the frequency in this non-dispersive system. This is typical for systems without mixing.

The system described by the equations (1) and (2) with constant flow Q and velocity $u = Q/A$ can be interpreted also as the wellknown Streeter-Phelps equations for the BOD-O_2 system. A is crosssectional area. Variable s_2 represents the oxygen deficit, k_{22} is the reaeration rate and M the time-invariant BOD loading rate. The maximum deficit or critical oxygen concentration occurs at the location where $ds_2/dx = o$ and $k_{22}\, s_2 = k_{12}\, S_1 = s_1\, S_1$, which means that the deoxygenation rate $k_{12}\, s_1$ is balanced by the reaeration rate $k_{22}\, s_2$. Setting the derivative of equation (2b) to zero results in

$$t_c = \frac{1}{k_{22} - k_{11}} \ln \frac{k_{22}}{k_{11}} \left[1 - \frac{S_{o,2}\,(k_{22} - k_{11})\,Q}{k_{11}\, M} \right] \tag{3}$$

in which t_c is the critical time of travel x_c/u at which this minimum occurs. In this model k_{12} is identical to k_{11}.

Equation (3) shows that the time and location of the maximum oxygen deficit are controlled by reaction rates only unless the initial deficit $S_{o,2}$ has a certain value. In this latter case the initial dilution M/Q plays also a role. The magnitude of the deficit can be evaluated by substitution of equation (3) into equation (2b). The effect of the "quantity" (Q or $u = Q/A$) on the quality ($s_2\,(x_c,\ t_c)$) can be calculated after that.

It should be noted however that generally these relations are more complex because the "quantity" also affects the system parameters which have been considered as constants in the preceding discussion. In this example an increasing flow Q will also cause a change in the reaeration constant k_{22} due to increased stream velocity and water depth.

Example 2 Plug flow with axial dispersion, impulse input, single system.

For constant flow Q and cross sectional area A and hence also time-invariant velocity u of the water through the reactor a mass balance results in:

$$\frac{\delta s}{\delta t} = -u\frac{\delta s}{\delta x} + E\frac{\delta^2 s}{\delta x^2} - ks \tag{4a}$$

E represents the dispersion coefficient and has been considered not to be function of x. Analogous to Fick's laws the dispersion term assumes that the mass of material being transferred is directly proportional to the concentration gradient. Hence the dimension of E is identical to that of the molecular diffusion coefficient, m^2/s, but its order of magnitude is much greater.

The solution to equation (4a) for a discharge of M kg at time $t = 0$ at $x = x_o$ is:

$$s(x, t) = \frac{M}{2A\sqrt{\pi E t}} \exp \left[-\frac{(x - x_o - ut)^2}{4Et} - kt \right] \tag{4b}$$

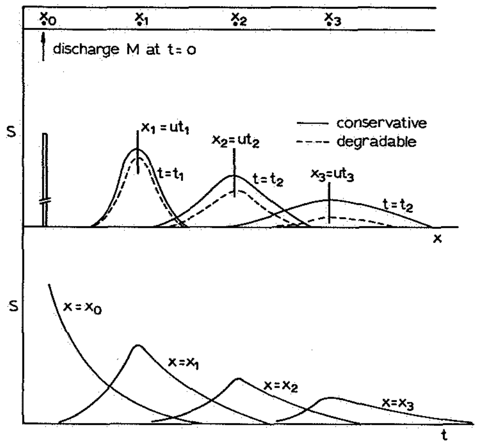


Fig. 5 **Impulsive response of plug flow ractor with axial dispersion.**

This expression represents the normal probability density function and hence the distribution of the concentration is symmetrical with respect to x with a standard deviation of $\sigma_x = \sqrt{2Et}$ and the maximum at $x = x_o + ut$. With t as independent variable the distribution is skewed with a standard deviation

$$\sigma_t = \frac{1}{u}\sqrt{2Et + 8E^2/u^2} \tag{4c}$$

for conservative materials ($k = 0$). Figure 5 depicts these relations.

In practice this description can be helpful to estimate downstream concentrations in a river in which an accident or calamity has caused the sudden discharge of toxic or otherwise polluting substances. The decline of the peak concentration while progressing downstreams is caused by dispersion and reaction. The effect of dispersion is represented by the term

$$\frac{M}{2A\sqrt{\pi Et}} \tag{4d}$$

which decreases with time and the contribution of decay is represented by the term exp-(kt).

The effect of the "quantity" on the water quality is manifest in the parameter u in equation (4b) which is related to the system dimensions A and the flow Q. A hidden effect is through the tendency of the dispersion coefficient E to increase with the dimensions of the system and the shear stress velocity (Bansal 1971). The tendency of rivers to have an expanding cross sectional area toward the sea and the tidal influence in the estuarine part of the river will often require the dispersion coefficient to be considered as a function of distance. Equation (4a) is too simple for such a situation. An alternative is the possibility to simulate the mixing properties of a river or a tidal river by modelling as a series of backmix reactors. The ratio between advective transport and transport by dispersion as characterized by the Peclet number uL/E is decisive for the number of backmix reactors that should be fitted in a stretch of river with length L in order to obtain corresponding mixing properties. A high Peclet number results in a high number of backmix reactors.

Still such descriptions are approximations of reality. Deviations of the idealized theoretical model behaviour will occur, for instance by the presence of dead zone's in the river. However, the information present in an observed impulsive response function of a water body as measured experimentally with a dye or other tracer, is extremely valuable in the assessment of the systems response to other inputs. The frequency transfer function can be calculated from the impulsive response function (Solodovnikov, 1960) and this complex function describes how periodic inputs are transformed by the system. An important application is the prediction of the attenuation of the amplitude of the input as a function of the frequency. The output response of dispersive systems decreases as the frequency of the input is increased.

Physically this is due to the proportionality of dispersive transport to the concentration gradient and high frequency inputs do induce steeper local concentration gradients. Hence an analysis of impulsive response can help to decide whether or not averaging of time variant discharges is appropriate or not. In the following example a quantitative relation between amplitude and frequency will be presented.

Example 3 Backmix reactor, sinus input, single system.

Assuming only minor variations in the watermark and hence a constant volume V the differential equation resulting from a mass balance of the reactor for constant flow is:

$$V\frac{ds}{dt} = M(t) - Qs - kVs \tag{6a}$$

in which $M(t) = M_o + M\sin(\omega t)$ represents the periodic input which may be associated with the flow Q through the reactor or an independent loading. The response of the variable part of the input is:

$$s = AM\sin(\omega t - \phi) \tag{6b}$$

where

$$\tan\phi = \frac{\omega}{k'} \text{ and } k' = k + \frac{1}{\tau} \tag{6c}$$

and

$$A = \frac{1/V}{\sqrt{(k')^2 + (\omega)^2}} \tag{6d}$$

A is the amplitude characteristic of the system, the ratio between the amplitudes of the concentration s in the reactor and the discharge rate M; the dimension of A is therefore time/volume. The phase characteristic ϕ of the system represents the phase lag between input and output, τ is the hydraulic detention time V/Q. Inspection of equation (6d) reveals that the attenuation of the amplitude increases for increasing volumes, higher frequencies of the periodic input and high values of k'. This latter term consists of contributions of decay and dilution rate; a high reaction rate and/or a short hydraulic detention time reduce the amplitude of the output. The admissibility of averaging time variable inputs can be evaluated through such relations.

The phase lag ϕ has a theoretical maximum of 90° corresponding to a time lag of a quarter of the period of the input. In systems with a high detention time and slowly reacting materials the maximum concentration may be observed a long time after the loading reached its maximum value. As an illustration data on the chloride concentration in the drinking water reservoir "De Grote Rug" situated near a branch of the river Rhine at Dordrecht in The Netherlands can be presented (fig. 6). The average hydraulic detention time of the reservoir is about 150 days. With some imagination the chloride concentration of the Wantij, the Rhine branch from which water is withdrawn, can be approximated with a sinus function for a period during 1976 and 1977. The amplitude of the input would be 60 g chloride per cubic meter and the period 660 days. Application of equations (6c) and (6d) results in a time lag of about 100 days and an output amplitude of 34 g chloride per cubic meter.

Figure 6 shows a very good agreement between this calculated response and the field data. It can also be observed that the high frequency variation in the input during july-august 1976 is reflected in the response as well, but with a much smaller time-lag and a stronger attenuation. However, a quantitative analysis of this episode is not appropriate

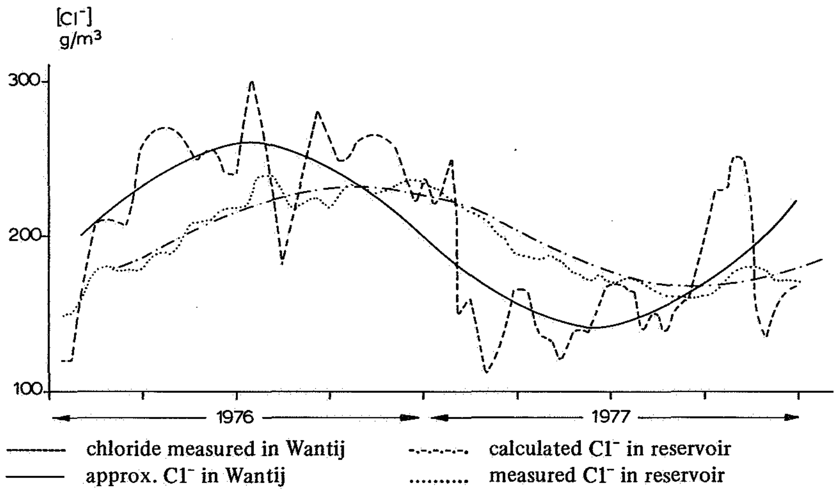


Fig. 6 Response of a reservoir as backmix reactor to a periodic input.

due to the low number of samples during this short period and the irregularities in the flow on this time scale.

This example of a backmix reactor also illustrates the effects of quantity (flow, volume, detention time) on the quality and dynamic effects associated with the system characteristics, notably the mixing properties. The indirect effect of quantity-aspects on the water quality through the influence upon process parameters has been touched upon in passing; examples were the influence of stream velocity and water depth upon the reaeration rate and the dependency of the dispersion coefficient on shear stress velocity and system dimensions. Van Straten in his contribution presents more details on this aspect. The discussion here is concluded with a short description of some other important processes affecting the water quality which are subject to changes by flow rate, water depth etc.

3.1. *Sedimentation/resuspension*

The content of suspended solids in rivers and lakes is a function of water velocity. In lakes the wind induced wave action may be important. It has been observed in laboratory studies that at equilibrium the number of suspended particles of a homogeneous, non-cohesive material is proportional to the averaged shear stress raised to the power ± 0,125. The proportionality constant is a function of particle size and particle density and also differs with the direction from which the equilibrium is attained. In case of a higher initial concentration (sedimentation) the equilibrium concentration will be higher than under conditions where the equilibrium is attained by erosion of the river bed (Graf, 1977). Consolidation of the sediment results in higher values of the critical shear stress at which cohesive material will be eroded. (Terwindt, 1976). Such phenomena are important for the water quality for instance in connection with the sedimentation and resuspension of sludge carried in rivers by storm water. In deep lakes a prediction of regions where erosion, transport or sedimentation will prevail can be made based upon fetch, bottom slope and water depth (Hårkanson, 1977).

3.2. *Biological oxygen demand, decay of bacteria*

It has been observed in laboratory studies that the oxygen consumption rate of suspended bacterial flocs (activated sludge) and microbial films is enhanced by increased turbulence (Hartmann, 1967; Richard and Gaudy, 1968). The BOD decay coefficient in the field can be expected to vary with the stream velocity in rivers or the degree of turbulent mixing in lakes. Apparently the ratio contact-surface to water volume is important for the removal of organics and the decay of micro-organisms attached to macrophytes and bottom materials. Turbulent mixing modifies the transport of substrates and oxygen. The destruction of enteric bacteria also is more rapid in shallow, turbulent streams than in deep, sluggish bodies of water (Fair c.s, 1968).

3.3. *Algal growth*

Owing to the influence of light intensity upon the growth rate of algae, water depth or mixed water depth is an important factor controlling the productivity of a water body. Light intensity is a function of time and climatic conditions, but also an exponential function of water depth. Therefore the time and depth integrated growth function is dominated by the so-called extinction depth ϵH, in which H represents the water depth and ϵ the light attenuation coefficient (Lorenzen, 1973).

Maintenance of sufficient mixing depth is an important alternative in the management of reservoirs suffering from algal Blooms. The Biesbosch reservoirs in The Netherlands have been specially equipped for this option.

4. CONCLUSIONS

Quantitative aspects are of paramount importance for the water quality. The pollutional loading, the dilution and mixing and the rates of several relevant processes and reactions are affected by the water quantity.

In The Netherlands the quantity is controlled intensively, sometimes with a view to the quantity itself (control of level in rivers, canals, polders), sometimes with a view to quality (flushing of polluted canals, control of salt intrusion) and often with a view to both quantity and quality (storage in reservoirs, infiltration in dunes). The close relation between quality and quantity in the physical reality requires also an integration in the areas of research and management. An increasing interaction between hydrologists, environmentalists and specialists in the field of water purification and water quality management will be beneficial for both research and management. Likewise the spatial integration of water management on the local, regional and national level should be promoted because activities in one area will bring on their effects in other areas.

REFERENCES

BANSAL, K. (1971) – Dispersion in natural streams. J. Hydr. Div. ASCE, November 1971, p. 1876.

BEUNDERS, B.A.J. (1978) – Phosphate budget and algal growth in the Barneveld Brook drainage area. MSc-Thesis, Twente University of Technology, Enschede, The Netherlands (in Dutch).

FAIR, G.M., J.C. GEYER and D.A. OKUN (1968) – Water and Wastewater Engineering, Volume 2, Chapter 33: Ecology and management of receiving waters. John Wiley; New York.

GRAF, W.H. and G.C. PAZIS (1977) – Les Phenomenes de déposition et d'érosion dans un canal alluvionnaire. J. of Hydraulic Research 15 (2), 151–166.

HÅRKANSON, L. (1977) – The influence of wind, fetch and water depth on the distribution of sediments in Lake Vänern, Sweden. Can. J. Earth Sciences 14, 397–412.

HARTMANN, L. (1967) – Influence of turbulence on the activity of bacterial slimes. J. Water Poll Control Fed. 39, 958.

LEVENSPIEL, O. (1967) – Chemical Reaction Engineering, John Wiley; New York.

LORENZEN, M. and R. MITCHELL (1973) – Theoretical effects of artificial destratification on algal production in impoundments. Env. Science and Technology 7 (10), 939–944.

MEYBECK, M. (1976) – Dissolved and suspended matter carried by rivers: composition, time and space variations and world balance. In: Interactions between sediments and fresh water; proceedings of an international Symposium, Amsterdam, September 1976. Junk-Pudoc.

RICKARD, M.D. and A.F. GAUDY (1968) – Effect of mixing energy on sludge yield and cell composition. J. Water Poll. Control Fed. 40, R 129.

SOLODOVNIKOV, V.V. (1960) – Introduction to the statistical dynamics of automatic control systems. Dover publications Inc.; New York.

TERWINDT, J.H.J. (1976) – Deposition, transportation and erosion of mud. In: Interactions between sediments and fresh water; proceedings of an international Symposium, Amsterdam, September 1976. Junk-Pudoc.

NATURAL AND ARTIFICIAL SOURCES OF NITROGEN AND PHOSPHATE POLLUTION OF SURFACE WATERS IN THE NETHERLANDS

J.H.A.M. STEENVOORDEN and H.P. OOSTEROM

SUMMARY

Nitrogen and phosphate pollution of surface waters by various potential sources is discussed in general. Attention is paid to the more or less natural sources: precipitation, leaching of natural soil constituents, seepage of saline groundwater and to the artificial sources: domestic waste water, inlet water from storage canals, leaching of fertilizers, surface runoff and discharge of dune-water.

For three polders and three brook catchment areas the total nitrogen and phosphate load has been calculated. Polders and catchment areas differ with respect to the geo-hydrological situation, soil type, agricultural soil use and population density. The contribution by the more or less natural sources of pollution is much higher in the polders than in the brook catchments because of the influence of seepage and peat soil in the former. For polder waters the role of agricultural sources is not important neither for phosphate nor nitrogen.

In the sandy soil of the brook catchment areas on the other hand, the nitrogen pollution originates for the largest part from scattered agricultural sources. The contribution to the phosphate load by domestic sewage depends on the population density and the treatment method of waste water, but generally speaking it is an important factor. The use of P-fertilizers in agriculture up to now does not lead to an increase in the leaching of phosphate to the groundwater. Via surface runoff from badly drained grassland, P-fertilizers can be transported to surface waters. The quantification of this source needs further investigation.

For the explanation of the water quality information must be available about geology, hydrology, soil type and human activities.

1. INTRODUCTION

The fight against pollution of surface waters is heavily focussed on the removal of phosphate and nitrogen compounds because of their adverse effect on water quality. Each addition above the natural load causes undesirable changes in the biological equilibria of the receiving waters. Small or large fluctuations in the oxygen content of the surface waters and even anaerobic situations can occur as a result of increased algae growth. Pollution of surface waters with ammonium or organic nitrogen compounds can lead to a considerable consumption of oxygen when oxydation to nitrate occurs. The suitability of surface waters for some functions will be lowered by the presence of certain compounds. For example when surface waters are used for the production of drinking water, the nitrogen concentration should be lower than 11 $g.m^{-3}N$ (Ministerie van Verkeer en Waterstaat, 1975).

In many surface waters in The Netherlands high concentrations of nitrogen and phosphorus are periodically or constinously recorded (table 1). The question arises which

Table 1 Short and long term water quality aims for nitrogen and phosphate in surface waters in The Netherlands (Ministerie van Verkeer en Waterstaat, 1975) and the recorded concentrations in two areas

Parameter		Water quality aims		Recorded	
		short term	long term	Barneveldse Beek	Polder near Oranjeplaat
Kjeldahl-N	($g.m^{-3}$ N)	3.0	1.0	3.4	7.8
NH_4^+	($g.m^{-3}$ N)	2.0	<0.5	2.1	6.0
NO_3^-	($g.m^{-3}$ N)	4.0	2.0	5.6	0
Total-P	($g.m^{-3}$ P)	0.3	0.05	1.2	1.4

pollution source is to blame for this and what can be done to minimize pollution. A necessary and useful tool to answer this question is the nitrogen and phosphorus balance for surface waters. The balance for The Netherlands as a whole does not give adequate information to be used as basis for measures to be taken in a specific area, since even in dry summers more than 80 per cent of the phosphate imported by the river Rhine is transported directly to the sea. Moreover, large differences in nitrogen and phosphorus load between areas can be expected to occur due to differences in hydrological situation, soil type and soil use. In this article information will be given about the contribution of nitrogen and phosphate by some potential sources for the situation in The Netherlands. With this knowledge an explanation will be given for the nitrogen and phosphorus load as present in three polders and three catchment areas.

2. POLLUTION SOURCES OF NITROGEN AND PHOSPHATE

2.1. *Precipitation*

Research carried out about the chemical composition of precipitation in general gives information on the total load by dry as well as wet deposition. It therefore is not correct to ascribe this pollution to rain alone.

Table 2 Nitrogen ($g.m^{-3}$ N) and phosphate contents ($g.m^{-3}$ P) of precipitation

Measuring period	Number of stations	Nitrogen		Phosphate		Literature
		mineral*	total**	ortho	total	
'32–'37	1	0.6	–	–	–	Leeflang (1938)
'73–'75	1	2.4	3.2	0.03	0.08	Steenvoorden and Oosterom (1975)
'73–'74	14	2.0	–	–	0.15	Henkens (1976)
'78	12	2.4	–	0.01	–	KNMI/RIV (1978)

*NH_4^+ and NO_3^-; **Kjeldahl-N and NO_3^-; – not analyzed.

The chemical composition of the precipitation (table 2) has changed considerably as a result of increasing agricultural and industrial activities and traffic intensity. In the period 1932 through 1937 an average concentration of 0.6 $g.m^{-3}$ N has been measured. Now a total-nitrogen concentration of 3.0 $g.m^{-3}$ N seems to be an acceptable value. For total-phosphate the average concentration is 0.10^{-3} P. The yearly amount of rain in The Netherlands is roughly 750 mm. If all the precipitation would fall on open water in a certain area this would give a load of 22.5 kg N and 0.75 kg P per hectare.

2.2. *Groundwater*

In this paragraph attention will be paid to the natural contribution by groundwater to the load of surface waters. In paragraph 2.3. the influence of agricultural activities on the concentrations of nitrogen and phosphorus compounds will be discussed.

The groundwater not influenced by man's activities already contains a certain amount of nitrogen and phosphorus compounds. Discharge of a precipitation surplus via the groundwater system on open water always causes a natural load. The groundwater discharge not only depends on the local precipitation surplus, but also on the regional geo-hydrological situation. The discharge can be higher because of seepage and can be smaller as a result of infiltration.

Information about the natural chemical composition of the groundwater has been gained from analyses in the upper meter of the groundwater under nature areas. The concentrations found are to a very large extent determined by the soil composition. Especially the organic matter content is of influence with regard to the concentrations of nitrogen and phosphate (table 3). The dispersion of phosphate concentrations for one type of soil was rather wide. For sandy soils total-phosphate concentrations of 0.15 $g.m^{-3}$ P as well as 0.01 $g.m^{-3}$ P have been found. Therefore the calculation of the natural mineral load of groundwater in a certain area should be based on analyses of groundwater from the investigated area. The very high concentrations found under nature areas on a marine clay soil are influenced by the seepage of saline groundwater (table 4).

Table 3 Average nitrogen and phosphate concentrations in the upper meter of the groundwater under nature areas for different soil types (Steenvoorden and Oosterom, 1973; Bots et al., 1978)

Soil type	Number of nature areas	Nitrogen ($g.m^{-3}$ N)			Phosphate ($g.m^{-3}$ P)	
		Kjeldahl-N	NO_3^-	total	ortho	total
Sand	3	0.9	0.3	1.2	0.02	0.05
River clay	1	0.5	0.4	0.9	0.01	0.11
Cut-over high moor peat	4	4.8	0.3	5.1	0.01	0.09
High moor peat	2	5.8	0.6	6.4	<0.01	0.15
Mesotrophic low moor peat	4	5.1	0.5	5.6	0.04	0.28
Marine clay	5	11.1	0.3	11.4	2.6	3.2

Table 4 Nitrogen and phosphate concentrations in the deep groundwater (10 m – 100 m) of some areas in The Netherlands. The number of analyses placed between brackets

Area	Nitrogen ($g.m^{-3}$ N)	Phosphate ($g.m^{-3}$ P)	Literature
Groningen, Friesland*	6.6 (240)	0.9 (156)	Bots et al. (1978)
North-Holland**	18 (88)	2.9 (88)	Toussaint and Boogaard (1978)
Mid-western Netherlands*	8.0 (600)	1.0 (430)	Steenvoorden (1976)
Zeeland (South-west Netherlands)	11.8 (52)	2.3 (49)	ICW

*NH_4^+ and ortho-P; **total-N and total-P

In marine sediments the groundwater generally can be characterised not only by high chloride concentrations but also by high nitrogen and phosphate concentrations. The contents can reach values of 5 $g.m^{-3}$ P and 50 $g.m^{-3}$ N or even more. This eutrophic groundwater can be found in a small strip along the coast of the provinces Groningen and Friesland in the northern Netherlands and in the largest part of the provinces North-Holland, South-Holland (mid-western Netherlands) and Zeeland in the western part of the country. For the provinces Groningen and Friesland only groundwater samples have been used with a Cl^- concentration of 200 $g.m^{-3}$ or higher and the analyses in North-Holland have been performed on water from gas wells and from wells for cooling water. The high concentrations not only are found at depths of more than 10 meters below sea level, but also in the upper meter of the groundwater (table 3). The absence of nitrate and the high concentrations of ammonium in the groundwater have been caused by the long residence time in the subsoil and the anaerobic conditions in the eutrophic marine sediments.

The saline and eutrophic groundwater can reach the surface waters by several ways. A more or less natural way is seepage. Seepage of loaded water takes place when the pressure in loaded deep groundwater is higher than in the shallow groundwater and when between shallow and deep groundwater a soil layer with a low hydrological permeability is missing. Nutrient rich groundwater can reach the surface water also by an artificial way when an impermeable layer is perforated to win gas or cooling water. In the province North-Holland this has been done on a large scale and the average yearly load per square meter of water in the polder Schermerboezem, having an area of 80,000 ha, is roughly 16 g N and 1.7 g P (Toussaint and Boogaard, 1978).

2.3. *Leaching of fertilizers*

The application of phosphate fertilizers on grassland and arable land has not yet had a measurable effect on the transport of phosphate to the groundwater, when the dose is based on soil fertility and crop production. The phosphate concentrations in shallow

Table 5 Total phosphate concentrations in the drainage water of arable land (Henkens, 1971) and in the shallow groundwater under grassland (Steenvoorden and Oosterom, 1977) for different soil types

Arable land	$g.m^{-3}$ P	Grassland	$g.m^{-3}$ P
Sand	0.02	Sand	0.04
River clay	0.04	River clay	0.05
Old cut-over high moor peat	0.02	Low moor peat	0.11
Newly cut-over high moor peat	0.73		

groundwater under grassland and in drainage water from arable land (table 5) are the same as the concentrations in the shallow groundwater under nature areas on a comparable soil type (table 3). An exception must be made for newly cut-over peat soils, where higher phospate concentrations are measured because of the high mobility of the soil organic matter. The application of large amounts of cattle manure, up to 300 $m^3.ha^{-1}.year^{-1}$ with a dry matter content of 9%, during five years on a sandy soil did not have a measurable effect. One can expect, however, that continuously giving a high dosis of manure will, because of the mobility in the soil of organic phosphates, in the long term result in an increased phosphate leaching (Gerritse, 1977).

The consequences of fertilizing for the nitrogen pollution of the groundwater are dependent on:

- soil use (grassland, arable land);
- soil type;
- the level and type of fertilizing (fertilizers, manure);
- the hydrological situation.

The leaching of nitrogen occurs for nearly 100% in the form of NO_3^-. At the same fertilization level the nitrate concentration in the shallow groundwater is higher for arable land than for grassland (fig. 1). This is caused by mineralization of the organic matter of the remainder of crop and roots on arable land and moreover, by the absence in early spring of a growing crop that can take up the mineralized nitrate.

With respect to the results given in fig. 1 one should take into account that the arable land only received manure which causes a much higher nitrogen leaching than fertilizer gifts. The research on grassland and arable land has been performed in different years so that the total amount of leachate is not the same. The calculated groundwater feed was roughly 100 mm for the grassland plots and 300 mm for the ones on arable land. The leaching of nitrogen at the different manure doses on arable land is roughly 30% of the applied amount of nitrogen. On grassland the leaching was 5%.

Soil type plays an important role in leaching. At a comparable fertilization level the nitrate concentrations in the shallow groundwater under grassland are much higher for sandy soils than for soils with some organic matter in the profile and for soils with

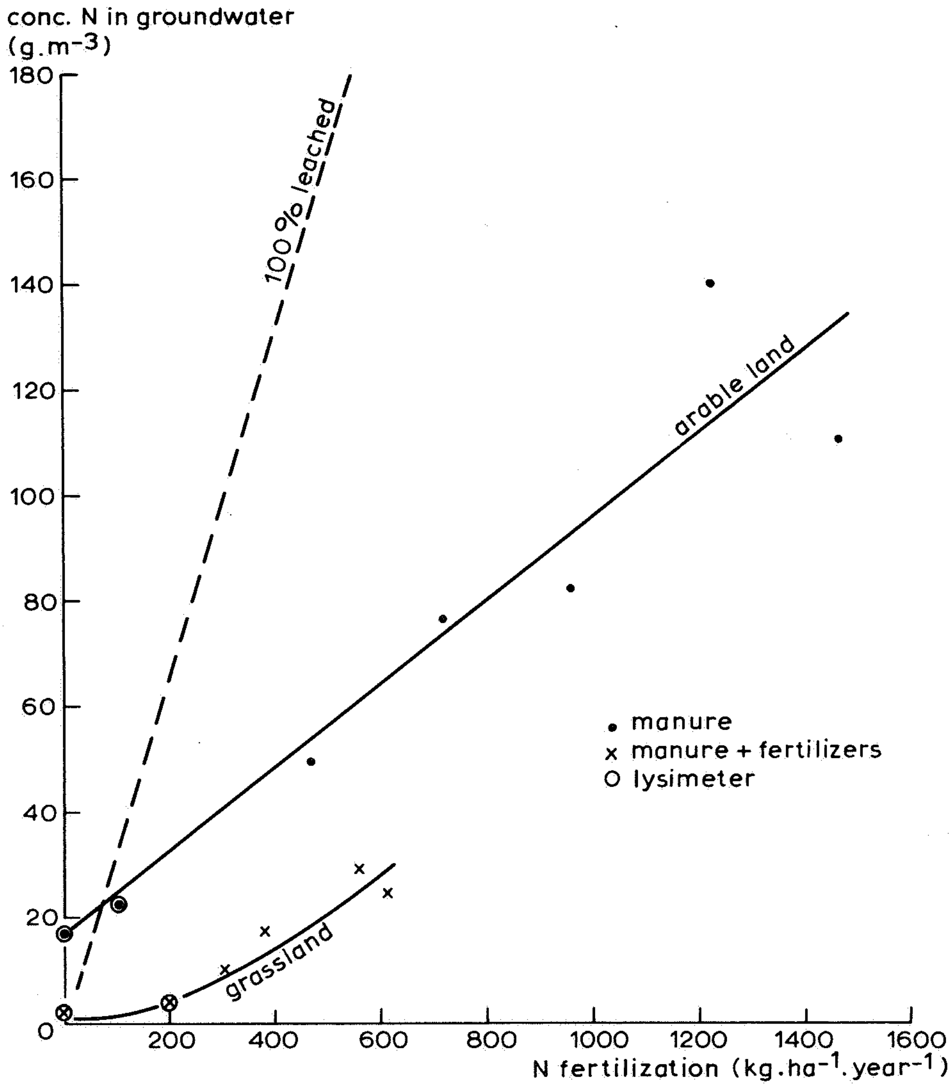


Fig. 1 Nitrate concentration in the upper meter of the groundwater as influenced by nitrogen fertilizing on sandy arable land and sandy grassland in field studies and lysimeter experiments. The line, '100% leached' has been calculated for a net groundwater feed of 300 mm.year^{-1} (after Steenvoorden, 1978b; Kolenbrander, 1978)

a finer texture, like clay soils (table 6). The same effect has been noted for arable land by Kolenbrander (1971).

The drainage situation has a large influence on the leaching of nitrogen. For average climatic conditions in The Netherlands the leaching of sandy soils in a good drainage

Table 6 **Influence of soil type on the nitrate concentration in the upper meter of the groundwater under grassland with a comparable fertilization level (Steenvoorden and Oosterom, 1977)**

Soil type	Fertilization (N $kg.ha^{-1}.year^{-1}$)			NO_3
	fertilizer	manure	total	N $g.m^{-3}$
Sand	350	205	555	29
Loamy and peaty sand	335	165	500	1
Clayey sand	435	170	605	5
Heavy clay	360	130	490	0

situation amounts to a loss of 50% of the mineral nitrogen which is present at the end of the growing season. For a moderate drainage situation the percentage is 80 because of an accelerated transport to open water via surface runoff and interflow (Rijtema, 1977). Before the groundwater reaches surface waters the nitrate content can decrease as a result of denitrification in the groundwater during transport and mixing with less polluted groundwater. Type of soil, residence time and geo-hydrological situation are important factors with regard to these processes.

Transport of precipitation surplusses can occur not only in a downward direction but also horizontally through or over the top soil. This will happen when in a certain period the quantity of precipitation exceeds the sum of evapotranspiration, groundwater feed and storage in and on the soil. The phosphate concentration in this surface runoff water depends on soil fertility and soil type (Kolenbrander, 1977; Sharpley et al., 1977). The more the phosphate dosage exceeds the uptake by plants, the higher the storage in the soil and the higher the phosphate concentration in the soil solution (fig. 2). At the same phosphate dosage the potential contribution on a sandy soil is higher than on a clay soil. For the climatic situation in The Netherlands one can expect that surface runoff will be a rare phenomenon and will be restricted to grassland which in general has a higher groundwater level and the soil therefore a lower storage capacity than arable land.

The nitrogen concentrations in the surface runoff are dependent on the mineral nitrogen stock in the percolated soil layer and on the amount of runoff water. The nitrogen concentration, in contrast to that of phosphate, will be in the same range as the concentration in the shallow groundwater. At higher precipitation surplusses the concentration will decrease because of the limited quantity of mineral nitrogen in the soil (Kolenbrander and Van Dijk, 1972).

2.4. *Domestic waste water*

The quantity of nutrients yearly produced as domestic sewage by the average individual include 5 kg nitrogen (Kolenbrander, 1971) and 1.5 kg phosphorus (Koot, 1970). The part reaching open water heavily depends on the way of handling the waste water. Most of the waste water nowadays is treated in biological purification plants.

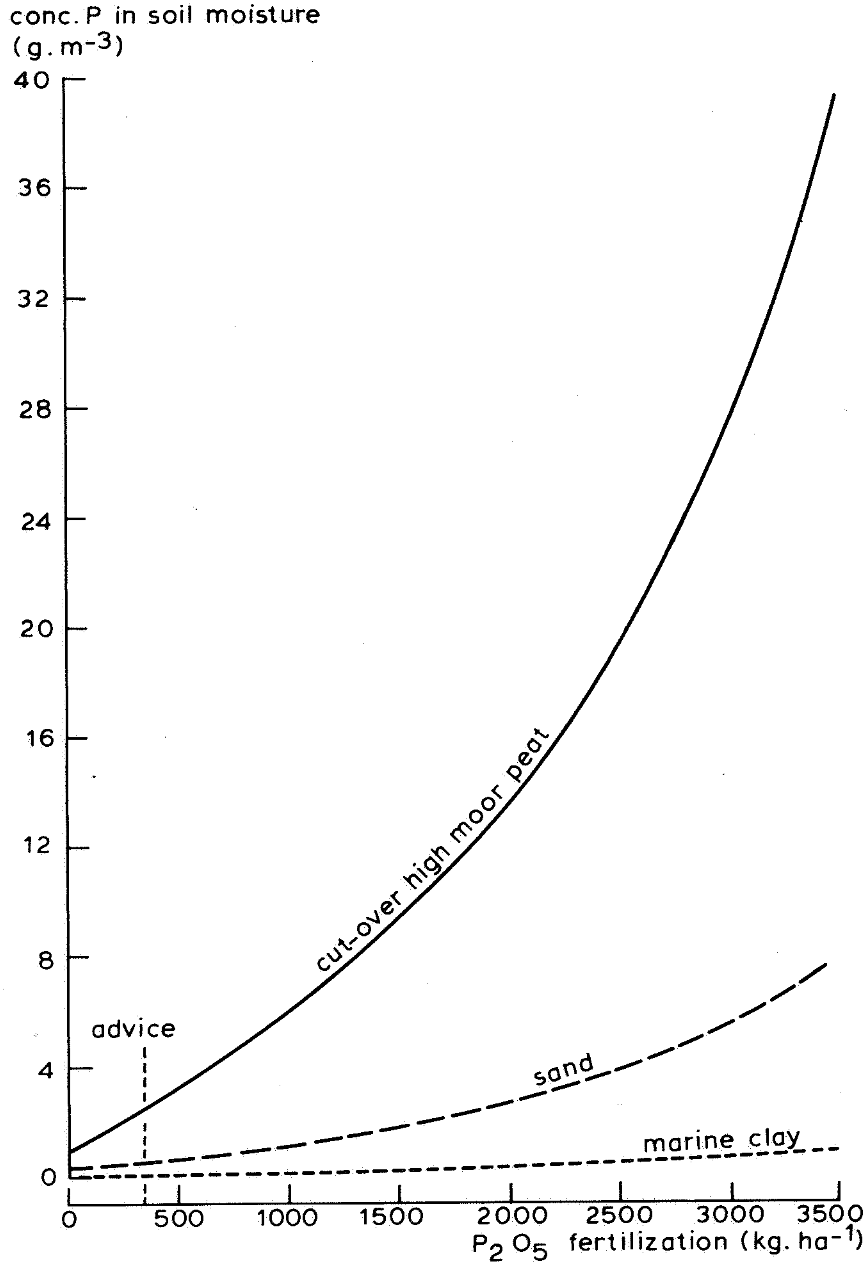


Fig. 2 Relation between total-P concentration in the soil moisture in the upper 20 cm of the unsaturated zone and the cumulative quantity of P_2O_5 added in 5 to 7 years in the form of fertilizers or organic manure (after Kolenbrander, 1977). The broken vertical line indicates the cumulative fertilization advice

An impression of the contribution to the nitrogen and phosphorus load by effluents of biological treatment plants can be given from the results of a research carried out in the area of the Schermerboezem (Hoogheemraadschap, 1976) and the Barneveldse Beek (Beunders, 1978). The concentrations found are slightly influenced by industrial discharge, but in the area of the Barneveldse Beek this applies only to one plant. The average nitrogen contribution per inhabitant equivalent is quite comparable for the two areas (table 7).

For the scattered inhabited sites sometimes a septic tank is used for the treatment, or the waste water is directly discharged on open water. The effect of septic tanks on the reduction of the nitrogen and phosphorus concentrations in effluents is largely unknown.

2.5. *Fresh water inlet*

The inlet of fresh water has several reasons. Rather general motives are the water supply of agriculture and horticulture and the water management for shipping purposes. To combat saline seepage and penetration of saline water near sluices sometimes large quantities of water are used to flush the canals (table 8). Large differences in the quantity

Table 7 Average discharge of nitrogen and phosphate via effluents of water purification plants in the area of the Schermerboezem and the Barneveldse Beek

Item	Dimension	Schermerboezem	Barneveldse Beek
Total-N	N $g.m^{-3}$	35	27
Total-P	P $g.m^{-3}$	–	13.5
Water discharge	$l.inhabitant^{-1}.day^{-1}$	203	200
N-discharge	N $kg.inhab.^{-1}.year^{-1}$	2.6	2.0
P-discharge	P $kg.inhab.^{-1}.year^{-1}$	–	1.0
Number of plants		9	6

Table 8 Quantity of inlet water (mm) to flush saline water ways and used for other purposes in the dry summer of 1976 for different soil types and averaged for a number of water supply areas (Van Boheemen, 1977)

Soil type	Number of supply areas	Flushing	Other purposes
Low moor peat	2	0	123
Marine clay	3	35	36
Low moor peat and marine clay	4	106	120
River clay	3	0	63
Sand	6	0	64

of input water within the same soil type can occur between the water supply areas, this depends on the differences in seepage of saline water, availability of water, etc.

The fresh inlet water in The Netherlands mainly originates from the river Rhine and it has very high nitrogen and phosphate concentrations. The average total-nitrogen and total-phosphate concentrations are roughly 7 $g.m^{-3}$ N and 1 $g.m^{-3}$ P (RWS/RIV/RID, 1976 and 1977). During the transport through channels towards the area where it is needed, important changes in the chemical composition may occur, for example by polluted water discharges. Analyses in the inlet water may be necessary therefore to calculate the total load of surface waters.

2.6. *Discharge of agricultural waste water*

Since the law of 1970 on combating pollution of surface waters is in action, many Water Authorities have played an active role in repelling discharges of agricultural waste waters. It can be presumed that possible illegal discharges will be a scarce phenomenon and thus will give only a small contribution to the mineral load of surface waters. The high concentrations in spread liquid manure, however, can have an important temporary influence on local water quality. These concentrations can be in the order of some thousands $g.m^{-3}$ for nitrogen and some hundreds $g.m^{-3}$ for total-P (Kolenbrander and De La Lande Cremer, 1967). The exact contribution to the total load of surface waters will always remain rather unsure and only a rough estimation.

3. DISCHARGE OF NITROGEN AND PHOSPHATE

The nitrogen and phosphorus compounds added to open water by pollution sources partly will be discharged in either a dissolved form or an undissolved but floating one. Partly the compounds will be involved in physical, chemical and biochemical processes by which products can escape to the atmosphere or can be stored in bottom sediments or in living organisms like reed, fish, etc.

Nitrogen can be lost to the atmosphere by denitrification and NH_3-evaporation. The importance of denitrification mainly depends on the availability of a biochemical oxidizable substrate and on water temperature. Especially in waterways receiving effluents of purification plants favourable denitrification conditions will exist. In such a situation denitrification rates have been measured of 35 $g.m^{-2}.year^{-1}$ N at 4°C and of 330 $g.m^{-2}.year^{-1}$ N at 21°C (Tiren et al., 1976; Van Kessel, 1976).

Little is known about the quantitative importance of NH_3-evaporation. In watery solutions ammonium is dissociated as follows: $NH_4^+ \rightleftarrows NH_3 + H^+$. When the pH-value of the temperature rises, the equilibrium shifts to the right. At a pH of 8 and a temperature of 0°C still all NH_4^+ is undissociated. At a pH-value of 9 and a temperature of 20°C already 30% of the NH_4^+ is in the dissociated form. In surface waters with high ammonium concentrations, for example in eutrophic polders, important quantities

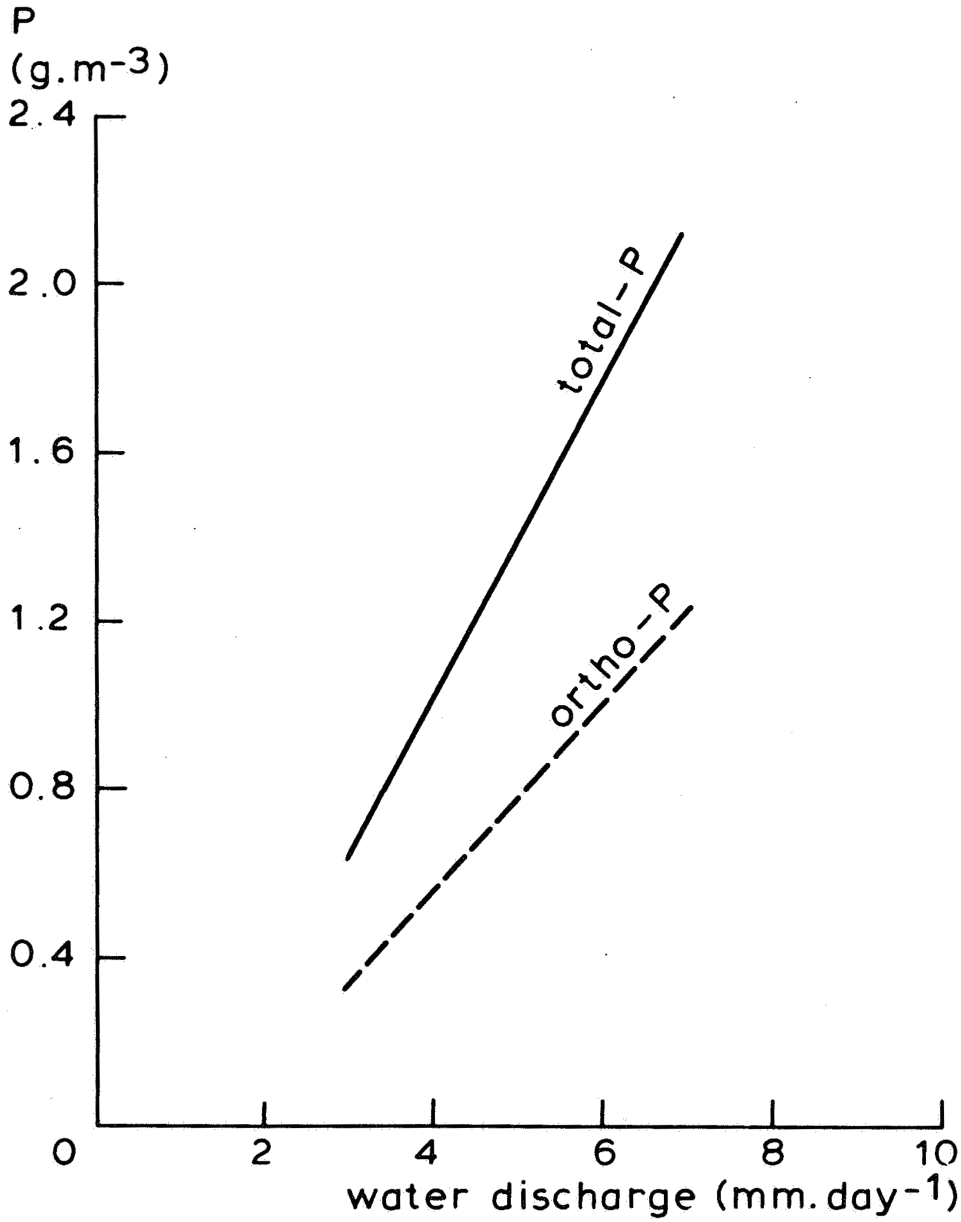


Fig. 3 Relation between the water discharge of the Barneveldse Beek and the concentration of orthophosphate and total-phosphate

of nitrogen may be lost via this process during algal blooms which create favourable conditions by a rise in pH.

Accumulation of nitrogen compounds and phosphate in the bottom sediment can proceed by precipitation of phosphate, adsorption of phosphate and ammonium and by settling of floating particles and dead water organisms and plants.

The disappearance of soluble phosphates from surface waters has been observed in a polder with saline seepage (Steenvoorden and Pankow, 1976) and in the Barneveldse Beek after the discharge of effluent by a purification plant in a dry summer period (Beunders, 1978).

All processes by which accumulation can occur in principle are reversible. When conditions change, nitrogen and partially phosphate can be released from the sediment to the surrounding water. For phosphate this reversibility is proved by the concentration increase during summertime in the lake Veluwe Meer. The only source in that period is the bottom sediment (Hosper, 1978). For the release of nitrogen and phosphate by sediments in some lakes, Vollenweider (1968) mentions values of 0.01 $g.m^{-2}.day^{-1}$ P and 1.2 $g.m^{-2}.day^{-1}$ N. The rate of exchange will depend among other things on the concentrations in the water, the oxygen conditions at the water – sediments interface, and the flow velocity of the water.

During a peak flow of nearly five days a close correlation existed between the water discharge and the concentrations of ortho- and total-phosphate (fig. 3). The concentration of dissolved P in the surface runoff from agricultural land depends on the fertility of the soil (Sharpley et al., 1976), which is nearly constant during a peak flow period of some days, so if surface runoff from agricultural land would have been the P-source the concentration in the water in the Barneveldse Beek would have been constant. It is very probable that in dry periods some phosphate is stored in the sediment, which is resuspended in periods with peak flows.

4. BALANCES FOR NITROGEN AND PHOSPHATE IN SURFACE WATERS

4.1. *Nitrogen and phosphate balances in polders*

Nitrogen and phosphate balances are given for three polders, all situated in the western part of The Netherlands. Industrial activities are not present and population density is low. Two of them, Frederikspolder and Veenderpolder, are under grass and consist of a eutrophic peat soil and a peaty clay soil respectively. The polder near Oranjeplaat consists of an arable clay soil. Other differences are the seepage intensity, the ratio between land and open water and the presence of gas wells. The last aspect is important for the Veenderpolder alone (table 9) (Steenvoorden, 1977).

The diffuse potential agricultural sources of pollution are leaching of fertilizers via the groundwater and disposal of agricultural waste water during the stabling period. In the groundwater nitrate could not be analysed, which is not such a big surprise as the level of fertilization is much lower than the level mentioned in table 6. All the nitrogen in the groundwater is in the form of ammonium and organic nitrogen and originates from

Table 9 Some characteristics of the Frederikspolder, the Veenderpolder and the polder near Oranjeplaat

Characteristic	Dimension	Frederikspolder	Veenderpolder	Polder near Oranjeplaat
Area	ha	68	156	40
Land use:				
grassland	%	85	96	–
arable land	%	–	–	99
water	%	15	4	1
Population density	inhabitants.ha^{-1}	0.20	0.25	0.10
Animal density agric. soil:				
cattle*	cu.ha^{-1}	2.1	2.4	–
total stock*	cu.ha^{-1}	2.3	2.7	–
Seepage	mm.day^{-1}	–0.1	0.1	1.0

*1 cu (cattle-unit) is the added value equivalent with that of 1 milk cow

Table 10 Load and discharge of nitrogen and phosphate in surface waters in three polders

Polder	Nitrogen (N kg.ha^{-1}.year^{-1})		Phosphate (P kg.ha^{-1}.year^{-1})	
	load	discharge	load	discharge
Frederikspolder	22	18	6.0	4.3
Veenderpolder	46	21	12.3	3.8
Polder near Oranjeplaat	110	52	19.3	9.1

natural soil resources. The same holds for phosphate (see chapter 2.3). In grassland polders a part of the dung-water produced in the stabling period can reach the ditches when storage facilities are lacking. From a census of storage facilities at farms in the north-western part of The Netherlands it appeared that roughly 8% of the farms had no facilities at all (Hoogheemraadschap, 1976). For the two grassland polders the discharge of agricultural waste water has been calculated to be 8% of the dung-water production in the stable period. The contribution by other sources could be calculated from hydrological and water quality data collected in the field.

The nitrogen and phosphorus load of the surface waters in the polders varies largely (table 10), which to a large extent is caused by differences in seepage intensity and the contribution by gas wells (table 11). The discharge of nitrogen and phosphate via the pumping stations controlling the polder water management is much lower than the total input, so that loss of nitrogen to the atmosphere and storage of nitrogen and phosphorus in the sediment will have played an important role. In the Frederikspolder and Veenderpolder the conditions for NH_3-volatilization have been favourable as in summertime pH-values between 8 and 9 frequently have been measured.

The N/P ratio of the total load lies between 3.5 and 5.5, which makes it very probable that in these polders nitrogen is the limiting factor for algal growth.

Sources which cause an inevitable pollution of surface waters and which can not be forced back by water authorities may be named more or less "natural" pollution sources. This background pollution is created by: precipitation which directly falls upon open water, leaching of natural soil components and seepage of saline water. The contribution by these sources is more than 25% of the total load for the three investigated polders and can even reach 99% (table 11). The artificial pollution is mainly caused by the inlet water from storage canals and saline water from gas wells. Discharge of dung-water from stables account for a relative small part of the nitrogen and phosphorus load in grassland polders. The significance of the last mentioned pollution source will increase in polders with a lower natural load. Especially in polders without seepage of saline water and in soil consisting of mesotrophic or oligotrophic peat this source will be of great concern. Even when all artificial sources of pollution would be eliminated it will be impossible to meet the short and long-term water quality aims (table 1) for a great number of polders in The Netherlands.

4.2. *Nitrogen and phosphate balances in catchment areas*

The nitrogen and phosphate balances have been calculated for the brooks Barneveldse Beek (Beunders, 1978; Steenvoorden, 1978a), Hupselse Beek (Kolenbrander and Van Dijk, 1972; Study Group Hupselse Beek, 1973) and Raalterwetering (Steenvoorden and

Table 11 Contribution (in %) by sources of pollution to the nitrogen and phosphate load of surface waters in three polders

Sources of pollution	Frederikspolder		Veenderpolder		Polder near Oranjeplaat	
	N	P	N	P	N	P
Natural sources:						
precipitation on open water	17	2	2	1	<1	<1
leaching of natural soil constituents	8	60	8	13	6	3
saline seepage	–	–	24	23	93	96
sub-total	25	62	34	37	99	99
Artificial sources:						
domestic waste water	4	4	2	4	<1	1
gas wells	–	–	32	29	–	–
leaching of fertilizers	0	0	0	0	1	0
discharged stable water	14	2	7	1	–	–
inlet water	57	32	25	29	–	–
sub-total	75	38	66	63	1	1
Total	100	100	100	100	100	100

Oosterom, 1973), all situated in the eastern sandy part of The Netherlands.

The soil in an important part of the catchment area of the Barneveldse Beek consists of fine loamy sandy soils. In combination with a bad drainage situation this can lead to surface runoff in wet periods. At a depth of 15 to 20 meters a clay layer with a low permeability can be found. This causes a shallow groundwater flow pattern. Roughly 75 per cent of the population lives in dwellings connected to sewage purification plants of which the effluents are discharged into the surface waters inside the catchment area.

In the area of the Hupselse Beek the top soil consists of medium coarse sand which contains some gravel. The thickness of the layer varies from nearly 0 to 10 meters. The underlying sediment is a clay layer with a very low permeability. The flow in the brook reacts very quickly on precipitation.

The top 40 cm of the sandy soil in the catchment area of Raalterwetering is slightly loamy. At greater depths the soil consists of a well-permeable coarse sandy soil. The groundwater flow pattern is rather deep as compared with the two previous catchment areas and the response of the flow on precipitation is slow.

The discharge of the three areas in the years of investigation was 300, 145 and 150 $mm.year^{-1}$ for respectively Barneveldse Beek, Hupselse Beek and Raalterwetering. Information about soil use, cattle intensity and population density can be found in table 12.

In sandy soil areas the cattle intensity normally lies on a much higher level than in other regions. The consequence might be that in periods that the manure can not be spread over the land, part of the dung-water is illegally dumped into the surface waters. Because of this uncertainty the contribution by diffuse agricultural sources will be calculated from the difference between the total nitrogen and phosphorus discharge from the catchment area via the water and the input by the known sources of pollution. By using this method one neglects the processes in the waterways by which nitrogen and

Table 12 Some characteristics of the catchment areas of the Barneveldse Beek, the Hupselse Beek and the Raalterwetering

Characteristic	Dimension	Barneveldse Beek	Hupselse Beek	Raalter-wetering
Area	ha	15,400	650	1525
Land use:				
grassland	%	59	64	94
arable land	%	6	16	5
water	%	1	1	1
other	%	34	19	–
Population density	$inhab.ha^{-1}$	2.8	0.4	0.3
Animal density agric. soil:				
cattle*	$cu.ha^{-1}$	2.7	2.0	2.2
total stock*	$cu.ha^{-1}$	12.4	4.1	4.5

*1 cu (cattle-unit) is the added value equivalent with that of 1 milk cow

Table 13 Annual load and discharge of nitrogen (N $kg.ha^{-1}.year^{-1}$) and phosphate (P $kg.ha^{-1}.year^{-1}$) for the surface waters of the three sandy catchment areas: Barneveldse Beek (BB), Hupselse Beek (HB) and Raalterwetering (RW)

Sources of pollution	Nitrogen			Phosphate		
	BB	HB	RW	BB	HB	RW
Natural sources:						
precipitation on open water	0.2	0.2	0.2	0.01	<0.01	<0.01
leaching of natural soil constituents	3.0	1.5	1.5	0.3	0.07	0.15
sub-total	3.2	1.7	1.7	0.31	0.07	0.15
Artificial sources:						
domestic waste water and industry	9.5	0.5	0.3	3.6	0.15	0.10
leaching of fertilizers	} 13.0	} 23.5	} 0.9	0	0	0
surface runoff and agric. discharges				–0.2	0.03	–0.01
sub-total	22.5	24.0	1.2	3.4	0.18	0.09
Total discharge	25.7	25.7	2.9	3.7	0.25	0.24

phosphorus can disappear from the water. As a first approximation it is allowable, however.

The natural leaching of P from the soil via the groundwater has been calculated from the total water discharge and the average concentration in the groundwater of the catchment areas. These concentrations are 0.10 $g.m^{-3}$P for the Barneveldse Beek and the Raalterwetering areas and 0.05 $g.m^{-3}$P for the Hupselse Beek area. For nitrogen the natural content is roughly 1.0 $g.m^{-3}$N for all three areas. About pollution by domestic sewage of the scattered population very little information is available. For the calculation of this contribution some approximations have been made. The first is that in the summer half-year no domestic sewage reached the open water because of infiltration into the subsoil. For the winter half-year the nitrogen in the domestic sewage was supposed to reach open water for 100% and the phosphate for 50%.

The load with N and P by natural sources of pollution in the sandy soil catchment areas has a much lower level than in polders (table 13, compare table 10 and 11). As the N/P-ratio of the natural load varies between 11 and 24 for the three investigated brooks phosphate seems to be the limiting element for algal growth. An important part of the artificial nitrogen pollution originates from agricultural sources. For the Barneveldse Beek, the Hupselse Beek and the Raalterwetering the share is respectively 51%, 91% and 31%. The much higher contribution by agriculture for the Hupsele Beek as compared with the Raalterwetering can be explained by the higher percentage of arable land and the speedy and shallow groundwater flow pattern. Both factors have a negative effect on the leaching of nitrate. With a higher residence time of groundwater in the subsoil denitrification and immobilization can reduce the nitrate concentration before the groundwater reaches open water.

Despite the very high cattle intensity in the Barneveldse Beek area, the share of agricultural sources in the N-load is much smaller than for the Hupselse Beek with the same total load. Factors which can explain the lower share in the Barneveldse Beek area are: the higher percentage of nature areas, the lower percentage of arable land and the heavier structure of the top soil (see fig. 1 and table 6).

The contribution by agriculture to the N-load in sandy soil areas is underestimated by the method used, because the N-losses to the atmosphere by denitrification of bottom sediments have been neglected. An estimation for the Barneveldse Beek area, where the disposal of sewage-plant effluents give a fair supply of organic material, leads to a yearly loss of roughly 5.5 $kg.ha^{-1}$ N on average for the whole area. Introducing this correction, the contribution by agricultural sources increases from 13.0 to 18.5 $kg.ha^{-1}$ N and the share in the total load from 51 to 61%. The influence of denitrification by bottom deposits in the other two brooks will be much smaller because there is no disposal of sewage-plant effluents. Moreover, in the Hupselse Beek sometimes high peak flows occur which remove the bottom deposits.

The phosphate load of the Barneveldse Beek is highly influenced by the disposal of sewage-plant effluents and domestic sewage of the scattered population. The share of this source in the total P-load of Barneveldse Beek, Hupselse Beek and Raalterwetering is 95, 60 and 40% respectively. The share of domestic sewage is overestimated by using this calculation method because the probable increase of P in the bottom deposits has been neglected. In eleven sediment cores taken over a distance of some kilometers downstream from a heavy disposal of sewage-plant effluents, the total P-storage has been analysed. Per square meter to a depth of 30 cm an amount of 120 g P is present (Hoekstra, 1979). So the contribution of agricultural sources is underestimated in this area. This is confirmed by some analyses in the runoff from grassland in the Barneveldse Beek area. The concentration of total-P ranged between 0.02 at low flows and 0.9 $g.m^{-3}$ at peak flows (Vasak, 1978; Steenvoorden, 1979). The negative values for agricultural P-sources in table 13 must also partly be explained by the errors made in the measurements or estimations of other sources. A more precise calculation of the P-pollution by agriculture can only be achieved when more information is available about changes in storage in bottom deposits. But also the contribution by domestic sewage from the scattered population is an uncertain item in the P-balance.

REFERENCES

BAKKER, H. DE (1979) – Major soils and soil regions of The Netherlands. Soil Survey Institute, Wageningen. PUDOC, Wageningen. 203 pp.

BEUNDERS, B.A.J. (1978) – Phosphate load and algal growth in the Barneveldse Beek catchment. Commission for Water Management in the Province of Gelderland, Arnhem. 82 pp (in Dutch).

BOHEEMEN, P.J.M. VAN (1977) – Water consumption by agriculture and horticulture in the dry summer of 1976. Nota 975, Institute for Land and Water Management Research, Wageningen. 82 pp (in Dutch).

BOTS, W.C.P.M., P.C. JANSEN and G.J. NOORDEWIER (1978) – Physical-chemical composition of surface water and groundwater in the Northern Netherlands. Regionale Studies 13, Institute for Land and Water Management Research, Wageningen. 111 pp (in Dutch).

GERRITSE, R.G. (1977) – Phosphorus compounds in pig slurry and their retention in the soil. In: Proc. Congress Utilization of Manure by Land Spreading, Modena, Italy. Sept. 1976 (Ed. J.H. Voorburg). Commission of the European Communities: 257-267.

HENKENS, Ch.H. (1972) – Fertilizers and the quality of surface water. Stikstof 15: 28–41.

———— (1976) – Nutritious matter or balances of minerals. Stikstof 83-84, 7: 335-362 (in Dutch).

HOEKSTRA, J. (1979) – Institute for Land and Water Management Research, Wageningen. Private communication.

HOOGHEEMRAADSCHAP (1976) – Hoogheemraadschap van de Uitwaterende Sluizen in Kennemerland en Westfriesland, Annual Report Technical Department. Edam. 46 pp (in Dutch).

HOSPER, S.H. (1978) – Nitrogen, phosphate and eutrophication. H_2O 18 (11): 385-388 (in Dutch.)

ICW (Institute for Land and Water Management Research, Wageningen). Archives

KESSEL, J.F. VAN (1976) – Influence of denitrification in aquatic sediments on the nitrogen content of natural waters. PUDOC, Wageningen. 104 pp.

KNMI/RIV. (1978) – Network for analysis of the chemical composition of the precipitation in The Netherlands. Results January to October.

KOLENBRANDER, G.J. (1971) – The eutrophication of surface water by agriculture and the urban population. Stikstof 15: 56-68.

———— (1974) – An estimation of the phosphate accumulation in The Netherlands in 1977. Report 10-74. Institute for Soil Fertility, Haren. 41 pp (in Dutch).

————(1977) – Runoff as a factor in eutrophication of surface waters in relation to phosphorus manuring. In: Proc. Congress Utilization of Manure by Land Spreading, Modena, Italy. Sept. 1976 (Ed. J.H. Voorburg). Commission of the European Communities: 181-197.

————(1978) – Institute for Soil Fertility, Haren. Private communication.

————and L.C.N. DE LA LANDE CREMER (1967) – Manure and slurry. Value and possibilities. H. Veenman & Zn. NV, Wageningen. 188 pp (in Dutch).

————and T.A. VAN DIJK (1972) – Eutrophication of the surface water by agriculture in the catchment of the Hupselse Beek. Study Group Hupselse Beek, Institute for Soil Fertility, Haren. 13 pp (in Dutch).

KOOT, A.C.J. (1970) – The future treatment of domestique sewage. H_2O 22 (3): 551-559

MINISTERIE VAN VERKEER EN WATERSTAAT (1975) – The combatment of the pollution of surface waters. Program 1975-1979. The Hague. 92 pp (in Dutch).

LEEFLANG, K.W.H. (1938) – The chemical composition of the precipitation in The Netherlands. Chemisch Weekblad 35: 658-664 (in Dutch).

RWS/RIV/RID. (1976 and 1977) – Results of water quality analyses in the main surface waters of The Netherlands.

RIJTEMA, P.E. (1977) – An approach of the nitrogen emission by grassland. Nota 982, Institute for Land and Water Management Research, Wageningen. 33 pp (in Dutch).

SCHMIDT-VAN DORP, A.D. (1978) – Nitrogen and eutrophication in shallow Dutch lakes. H_2O 16 (11): 352-357 (in Dutch).

SHARPLEY, A.N., J.K. SYERS and P.W. O'CONNOR (1976) – Phosphorus inputs into a stream draining an agricultural watershed. Water, Air and Soil Pollution 6: 39-52.

STEENVOORDEN, J.H.A.M. (1976) – Nitrogen, phosphate and biocides in groundwater as influenced by soil factors and agriculture. Comm. Hydrol. Onderzoek TNO, Den Haag. Versl. en Meded. 21: 52-69. Techn. Bull. ICW 97. 18 pp.

————(1977) – Sources of eutrophication in the Western Netherlands as illustrated for three polders. Proc. Symposium Effects of Urbanization and Industrialization on the Hydrological Regime and on Water Quality. Amsterdam, Oct. 1977. IAHS-AISH Publication 123: 344-351. Misc.Repr. ICW 210. 7 pp.

————(1978a) – Phosphate and nitrogen balance for the surface water in the catchment of the Barneveldse Beek. Nota 1080, Institute for Land and Water Management Research, Wageningen. 24 pp (in Dutch).

——— (1978b) – Influence of fertilizing on the chemical composition of groundwater. Nota 1043, Institute for Land and Water Management Research, Wageningen. 26 pp (in Dutch).

——— (1979) – Some intermediate results of the surface runoff investigations. Institute for Land and Water Management Research, Wageningen.

——— and H.P. OOSTEROM (1973) – Nitrogen, phosphate and organic matter in ground- and surface water in some areas. Cultuurtechn. Tijdschr. 12, 6: 1-19. Misc. Repr. ICW 143 (in Dutch).

——— and H.P. OOSTEROM (1973) – Nitrogen, phosphate and organic matter in groundwater and grassland of cattle farms. Nota 964, Institute for Land and Water Management Research, Wageningen. 22 pp (in Dutch).

——— and J. PANKOW (1976) – Eutrophication of a creek of the lake Veerse Meer and the influence of discharged polderwater. Nota 910, Institute for Land and Water Management Research, Wageningen. 23 pp (in Dutch).

STUDY GROUP HUPSELSE BEEK (1973) – Second report of the hydrologic research in the catchment of the Hupselse Beek. IHD-project, Wageningen. 37 pp.

TIREN, T., J. THORIN and H. NOMMIK (1976) – Denitrification measurements in lakes. Acta Agric. Scand. 26 (3): 175-184.

TOUSSAINT, C.G. and T. BOOGAARD (1978) – Chloride, nitrogen and phosphate load of polder waters by gas and cooling wells. Nota 1061, Institute for Land and Water Management Research, Wageningen. 19 pp (in Dutch).

VASAK, L. (1978) – Chemical composition of the groundwater in the catchment area of the Barneveldse Beek. Commission for the Management of Water in the Province of Gelderland, Arnhem. 50 pp (in Dutch).

VOLLENWEIDER, R.A. (1968) – Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular reference to nitrogen and phosphorus as factors in eutrophication. OECD, Paris. 159 pp.

HYDROLOGICAL ASPECTS OF OXYGEN BEHAVIOUR AND ALGAL BLOOMS IN RIVER SYSTEMS

G. VAN STRATEN

SUMMARY

A brief introduction in the basic structure of mathematical models for dissolved oxygen and phytoplankton in brooks and rivers is followed by a discussion of the impacts of hydrological characteristics like waterdepth and flow velocity on modelparameters such as reaeration and dispersion. The ratio of light penetration depth to total depth, and the total residence time available, are hydrologically determined factors which influence algal growth and, consequently, the oxygen balance. Illustrations are given for rivers in a Dutch eastern region. Based on the analysis the possible effects of canalization of brooks and rivers with respect to water quality are examined. It is suggested that even rough quality models could be fruitfully used to achieve a better design in such projects.

1. INTRODUCTION

Inevitably, aspects of water quantity are met when studying oxygen behaviour and algal growth in streams. The discharge rate directly determines the time available for biological and chemical processes. In addition, flow velocity and waterdepth influence processes such as mixing, reaeration and dispersion. An analysis and illustration of these aspects is the first aim of this publication. Examples will be taken from practice in the Gelderse Achterhoek region in the eastern part of The Netherlands. Research has been carried out in this area by Twente University of Technology during the past few years, within the framework of a comprehensive regional water management study.

Most brooks in the Achterhoek have been canalized. They are typical low-land brooks, characteristic of many other water courses in The Netherlands and other flat areas. Regulated brooks show drastic differences compared with brooks in a more natural condition. Typically, a tremendous increase in storage capacity is observed due to the expansion of the cross-section and the construction of weirs, resulting in a greater depth and a (much) larger residence time. It is, therefore, our second aim to investigate the response of water quality to these kinds of water quantity management measures, employing the conclusions of the first section. The basic idea is that it might prove helpful to apply even simple water quality models in the design phase of water quantity projects like stream regularization.

2. MODELS FOR WATER QUALITY

In order to be able to evaluate the water quantity aspects in water quality studies it would be convenient to first discuss briefly the basic outline of water quality models. For this purpose a short description of the model used in the Achterhoek region is presented.

Basically a model like this comprises of a set of mass balances. For each arbitrary solute the mass balance can be written as

$$\frac{\partial c}{\partial t} = -\frac{\partial (uc)}{\partial x} + \frac{\partial}{\partial x} D \frac{\partial c}{\partial x} + B - P \tag{1}$$

where c is the concentration, u the cross-sectional mean flow velocity, D the dispersion coefficient and B and P the sources and sinks respectively. Equation (1) represents a one-dimensional model. The assumption of one-dimensionality seems to be a reasonable one, since in rivers the dominant flow is in longitudinal direction. A more detailed discussion follows later. In principle, the concentration on time t and location x can be calculated if the parameters – not necessarily time and space independent – as well as the boundary conditions and initial conditions are known.

It is not always clear a priori which substances should be included in the model. However, the objectives of the model should be paramount to this question. If the water quality resulting from BOD-removal in sewage discharges is of primary interest, dissolved oxygen and BOD will be a natural choice as the model variables. Writing equation (1) for both DO (C) and BOD (L) results in

$$\frac{\partial C}{\partial t} = -\frac{\partial (uC)}{\partial x} + \frac{\partial}{\partial x} D \frac{\partial C}{\partial x} - k_1 L + k_2 (C_s - C) \tag{2}$$

$$\frac{\partial L}{\partial t} = -\frac{\partial (uL)}{\partial x} + \frac{\partial}{\partial x} D \frac{\partial L}{\partial x} - k_1 L \tag{3}$$

The sink term couples both balances. The system is characterized by the parameters k_1 (the BOD-decay constant) and k_2 (the reaeration constant). For steady state and no dispersion the famous classical Streeter and Phelps equations (Streeter and Phelps, 1925) are obtained.

Since the time of Streeter and Phelps the necessity of primary and secondary treatment of waste water has become self-evident in many parts of the world. In such a situation pure DO-BOD-models are no longer of interest, although the modeling of the environment impacts due to the BOD left in the effluents might be a topic in future. Also, specialized, non-stationary DO-BOD models are still required in the judgement of storm water overflows effects, a field which has been little explored up till now.

Ironically enough, it is highly likely that the very results of extreme waste water treatment have contributed to the general recognition of eutrophication as the next major water quality problem. Excessive algal blooms are the most noticeable consequences of the eutrophication process, not only in lakes but also in rivers, especially in imponded river sections. A long residence time and improved clarity, combined with high, sometimes very high, nutrient discharges are responsible for this phenomenon (see below). This is the reason why the primary interest of water quality agencies is shifting from DO-BOD-problems towards questions like: what will be the effect of phosphorus (or nutrient) removal for the receiving water body? Clearly then, if a model is to be used as a tool in the decision process, it must cover not only BOD and DO, but also algae and phosphorus, at least.

So much for backgrounds and trends in water quality model development. One other remark relating to the importance of quantity aspects must be made. The accuracy that will be required for the hydrodynamic terms relates to the accuracy that can be achieved for the biological terms. Our models are, in fact, a very poor description of biological reality. The degree of detail in the model will be a function of the objectives, and of the data available or collectable in order to calibrate a more complex model. To give an example: a better dynamic description of BOD-decay could be obtained if bacteria were also included as a model variable. This seems to be particularly necessary for the correct judgement of the effectiveness of nitrogen withdrawal on the oxygen consumption by nitrification. However, at the same time the data requirements increase tremendously, and this serious drawback has to be balanced against the profits of a somewhat improved predictability.

The considerations above are less significant for our present aim, which is to illustrate the effects of water quantity on water quality. Therefore we confine ourselves to the presentation of the somewhat simplified equations of a model used in the Gelderse Achterhoek study (see table 1; Van Straten, 1977; Van Straten and De Boer, 1979). The model state variables are BOD, DO, algae (as chlorophyll-a) and orthophosphorus. The

Table 1 Simplified Model Equations

		Growth	Death	BOD-decay	Reaeration	Reaction
algae	$\frac{dA}{dt}=$	$+k_{pa}F_nA$	$-k_{da}A$			
BOD	$\frac{dL}{dt}=$		$+Y_1k_{da}A$	$-k_1L$		
oxygen	$\frac{dC}{dt}=$	$+Y_2k_{pa}F_nA$	$-Y_3k_{da}A$	$-k_1L$	$+k_2(C_s-C)$	
phosphorus	$\frac{dP}{dt}=$	$-Y_4k_{pa}F_nA$	$+Y_5k_{da}A$			$-k_sP$

k : rate coefficients
Y : conversion factors
C : dissolved oxygen saturation concentration
F_n : $F.\ P/(P_k+P)$
P_k : Michaelis-Menten factor for phosphorus
F : light factor (see text)

algal growth rate is phosphorus-limited at low phosphorus content (some ten μg P/l) and exclusively light-limited at high phosphorus concentrations.

3. CROSS-SECTIONAL MIXING

An important assumption in equation (1) is that a one-dimensional description is sufficient, i.e. that cross-sectional concentration differences are insinificant. This assumption is not justified directly downstream of a waste outfall. A dispersion experiment with a coloured tracer in the Groenlose Slinge revealed noticeable concentration differences in lateral direction as far as 1000 m from the point of discharge. An estimation of the mixing length, i.e. the length required for complete mixing can be obtained from a formula derived by Fisher (1967). In a somewhat modified form

$$L_{mix} = 1.8 \frac{b^2 Ch}{R\sqrt{g}} \tag{4}$$

where b is a characteristic width (in practice about half the total width), Ch the chezy-coefficient (usually in the range of 30 to 40 $m^{1/2}/s$), g the gravity and R the hydraulic radius.

Note that the flow rate is absent in equation (4). To understand this result consider the case of increasing discharge rate. The mixing time decreases due to the increasing turbulence intensity. But at the same time the total distance travelled during this shorter period will remain about the same due to the higher flow velocity.

For wide rivers, where $R \simeq H$ (depth), a shorter mixing distance is obtained when the water depth becomes larger. At first glance this result is unexpected. An explanation can be found in the eddy structure. The eddy size is essentially depth limited: deeper water allows larger eddies, inducing a stronger lateral mixing and thereby a shorter mixing length.

Application of equation (4) on Groenlose Slinge data results in a mixing length of 2000 m. This is fairly close to our observations, although the result suggests that equation (4) slightly overestimates the required mixing distance. Some typical examples showing the significance of the mixing length are illustrated in table 2. In some instances the applicability of a one-dimensional model is debatable, especially for large rivers, e.g. River Rhine.

Although sufficient mixing in the vertical direction is usually assured due to the much smaller mixing distance, it has been found that occasionally severe vertical gradients can occur during stagnant periods in imponded river sections (fig. 1). This is particularly true for oxygen. Algal growth and associated oxygen production take place predominantly in the surface layers, because of the locally favourable light climate, whereas in the bottom layers near the sediments oxygen consumption often occurs. Thus vertical gradients are formed which sustain under conditions of low vertical mixing. An important consequence is, that dissolved oxygen concentrations obtained

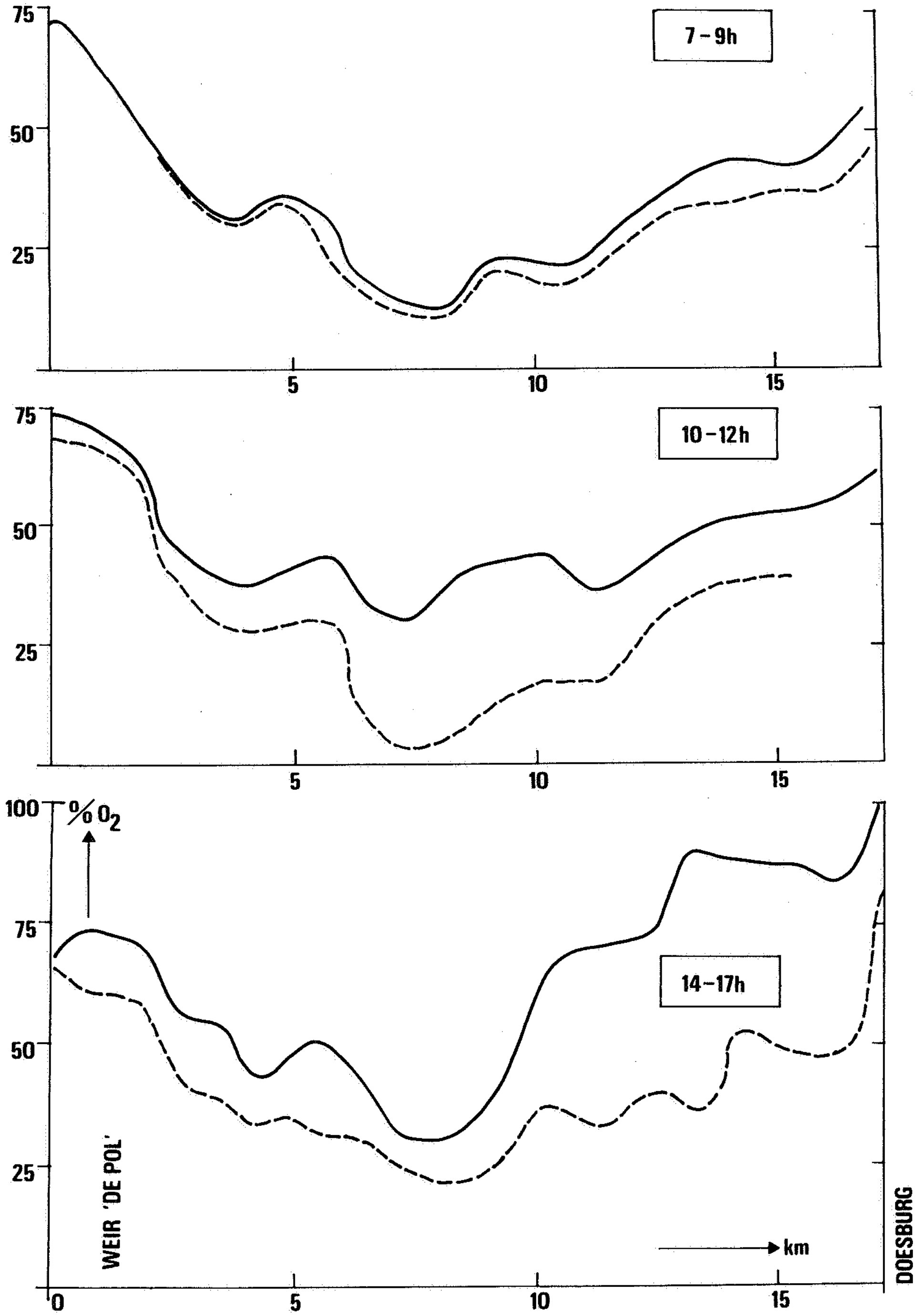


Fig. 1 Development of vertical gradients in the River Oude Yssel, Dissolved oxygen (in % of saturation) early in the morning, at noon and in the afternoon of 19th September, 1974. (Nissink, 1975)

——— 0.5 m below the surface
--------- 1.5 m below the surface

Table 2 Rough values for mixing length and dispersion coefficient

Brook/River	Q (m^3/s)	B (m)	H (m)	u (m/s)	L_{mix} (m)	D (m^2/s)	D/uH (–)
Hupselse brook	0.08	3	0.5	0.06	100	0.04	1
Groenlose Slinge River	1	20	1	0.05	2000	0.08	2
Oude IJssel River	10	40	4	0.07	2500	1.70	6
Bornse brook	0.60	5	0.3	0.40	600	1.3	10
Brielse Meer – channel	60	60	3	0.40	7000	28	25
Waal (Rhine) River	1500	260	5	1.00	250000*	200	40

* The mixing length for the Waal River will be less in practice due to groynes, bents, harbour constructions etc.

from surface samples, or measured with an electrode near the surface, can be considerably biased.

4. THE ROLE OF FLOW RATE AND WATER DEPTH – DISPERSION

As a consequence of the dispersion term equation (1) is a second order partial differential equation which considerably complicates a numerical solution. No wonder then that during the study much attention has been given to the problem under which conditions dispersion effects can be neglected. If dispersion is neglectable a considerable advantage is gained in field work too, since it is then possible to investigate the reaction kinetics by simply following a single plug of water, without fear of interference with "neighbouring" plugs.

The conditions for which dispersion can be neglected have been investigated by De Boer (1973), cf. also Van Straten and De Boer (1979) and Thomann (1973), among others. For a simple BOD model the conditions are

$$D \ll \frac{u^2}{4k} \tag{5}$$

$$\omega \ll \frac{u}{2.5x}\sqrt{\frac{ux}{2D}} \tag{6}$$

Here k represents the first order decay rate for BOD. Equation (5) is the condition for the static component of the discharge. This condition is nearly always met due to the relatively low value of k (range 0.2–0.7 day^{-1}). Equation (6) represents the condition for the dynamic component, where ω is the characteristic frequency (in radians). In sewage treatment plants the day cycle produces a characteristic frequency component of $2\pi/24h^{-1}$, but also higher frequencies occur due to the discrete time switching of discharge pumps. A stormwater overflow event can be considered as a pulse with steep flanks and therefore a broad frequency spectrum. In other words, the pulsetype character of the overflow causes large longitudinal gradients in the recipient river, which means that dispersive fluxes are already significant at relatively low dispersion coefficients.

Clearly, information is required concerning the dispersion coefficient in order to apply equations (5) and (6). Generally speaking, a direct measurement by a suitable tracer technique is preferable. However, a predictive formula would be useful for a first order estimate, and also to clarify the relation between flow rate and water depth.

A rough rule is given by

$$D = c_1 u_* R \tag{7}$$

or for wide rivers

$$D = c_2 u H \tag{8}$$

where u_* is the friction velocity ($u\sqrt{g}/Ch$) and R the hydraulic radius. However, we find that the constant c_2 is by no means universal (nor is c_1) but depends on geometry and bottom roughness. Table 2 lists some roughly evaluated values for a range of different rivers.

Substitution of equation (8) in (5) and (6) eliminates D from the conditions:

$$u \gg 4kc_2 H \tag{9}$$

$$\omega \ll \frac{1}{2.5\sqrt{2c_2}} \frac{u}{x} \sqrt{\frac{x}{H}} \tag{10}$$

The interpretation of the dynamic condition (10) is that dispersion can be less easily neglected if the distance is greater or the flow velocity smaller. Therefore, relatively important dispersion effects are likely to occur during stagnant periods, such as frequently met in summer.

5. THE ROLE OF FLOW RATE AND WATER DEPTH – REAERATION

The oxygen transfer from atmosphere to water is an important process and is, in fact, the basis of the self-purification capacity of streams. Many reaeration formulae have been proposed in literature. An overview is given by Bansal (1973). The fact that a direct determination in the field is extremely complicated by the presence of other oxygen consuming or producing processes is in part responsible for the existence of such a motley assembly of descriptions.

A popular formula with a theoretical basis is the O'Connor and Dobbins formula (1956) which is represented here in a somewhat unusual form

$$k_2 = H^{-2} B^{-0.5} D_L{}^{0.5} Q^{0.5} \tag{11}$$

Here H and B are the mean depth and mean width respectively, D_L is the molecular

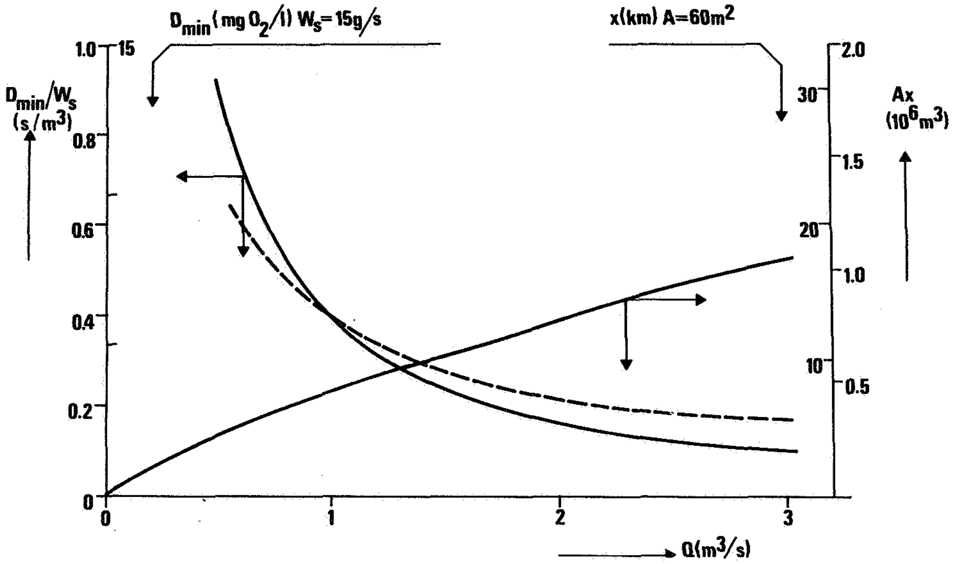


Fig. 2 For a brook *with* weirs: deficit at dissolved oxygen minimum D_{min} per unit waste discharge W_s, and the distance from the source, as a function of flow rate Q. As an example, on the right side of the left axis D_{min} is given for a constant discharge rate of 15 g/s.
Note: Aeration on weirs neglected.
Dashed line: example for D_{min} when W_s increases with Q ($W_s = 10 + 5Q$)
Parameters: $H = 2$ m; $B = 30$ m; $k_1 = 0.2$ day^{-1}

Table 3 Application of Streeter-Phelps for natural brooks and brooks with weirs

Solution Streeter-Phelps

if $k_1 \neq k_2$:

$$D(\tau) = \frac{k_1}{k_2 - k_1}\frac{W}{Q}\exp(-k_1\tau) - \exp(-k_2\tau)$$

$$\tau_{min} = \frac{1}{k_2 - k_1}\ln(k_2/k_1)$$

$$D_{min} = D(\tau_{min})$$

if $k_1 = k_2$:

$$D(\tau) = k_1\tau\frac{W}{Q}\exp(-k_1\tau)$$

$$\tau_{min} = 1/k_1$$

$$D_{min} = \frac{W}{Q}\exp(-1)$$

Reaeration

$k_2 = H^{-2}Q^{1/2}B^{-1/2}D^{1/2}$

in natural brooks:

$H = Q^{2/3}Ch^{-2/3}S^{-1/3}B^{-2/3}$
$k_2 = a_n Q^{-5/6}$
$a_n = B^{5/6}Ch^{4/3}S^{2/3}D^{1/2}$

in imponded rivers:

$H \simeq$ constant
$k_2 = a_s Q^{1/2}$
$a_s = H^{-2}B^{-1/2}D^{1/2}$

diffusion coefficient for oxygen in water and Q is the discharge rate. According to equation 11 the reaeration increases with the square root of the discharge rate, and decreases with the square of the depth.

It is easy to demonstrate the effect of discharge rate on the dissolved oxygen pattern through its influence on reaeration. For this purpose we use the solution of the Streeter-Phelps equations, given in table 3 (steady state, no dispersion). Substitution of an appropriate depth to flow rate relation eliminates the depth from equation (11) enabling the calculation of the maximum deficit as a function of flow rate. Figure 2 represents the result for a river section with weirs, where the depth is practically independent of the flow rate. Values typical for brooks in the Achterhoek have been assigned to the other parameters in the equation.

The computation assumes a constant waste load, which leads to a lower initial instream BOD at higher flow rates. In practice, the waste load very often increases with flow rate, and therefore figure 2 is a little bit on the optimistic side. However, the minimum dissolved oxygen content is a linear function of waste load, and the effect of increasing load with higher flow rates can be readily computed in each individual case. (The dashed line in figure 2 gives an example assuming a linear relationship). On the other hand, figure 2 is certainly too pessimistic, because the reaeration capacity of weirs has been ignored.

To summarize the discussion up till now: in Streeter-Phelps models dissolved oxygen deficit is a linear function of waste load. For weired river sections the minimum dissolved oxygen concentration occurring is lowest for low river flow rates, and in addition to this it is very sensitive to variations in flow rates in that range. The location of the minimum shifts in the direction of the source if the flow rate drops. The location does not depend on the waste loading rate.

6. RELATION HYDROLOGY – ALGAL GROWTH

6.1. *Description of the algal growth process*

The growth rate of algae is determined by the maximum growth rate k_{pn}, attenuated by a Michaelis Menten term for the nutrient limitation, and a light factor F, which represents the effect of the overall light climate (see equations in table 1).

A reasonable description of the light factor comprises the following elements (Van Straten, 1977).

– the relative growth rate as a function of light intensity, for example according to Steele

$$G(I) = \frac{I}{I_s} \exp\left(-\frac{I}{I_s} + 1\right) \qquad (12)$$

where G(I) is the ratio of the growth rate at light intensity I to the maximum growth rate occurring at the optimal light intensity I_s.

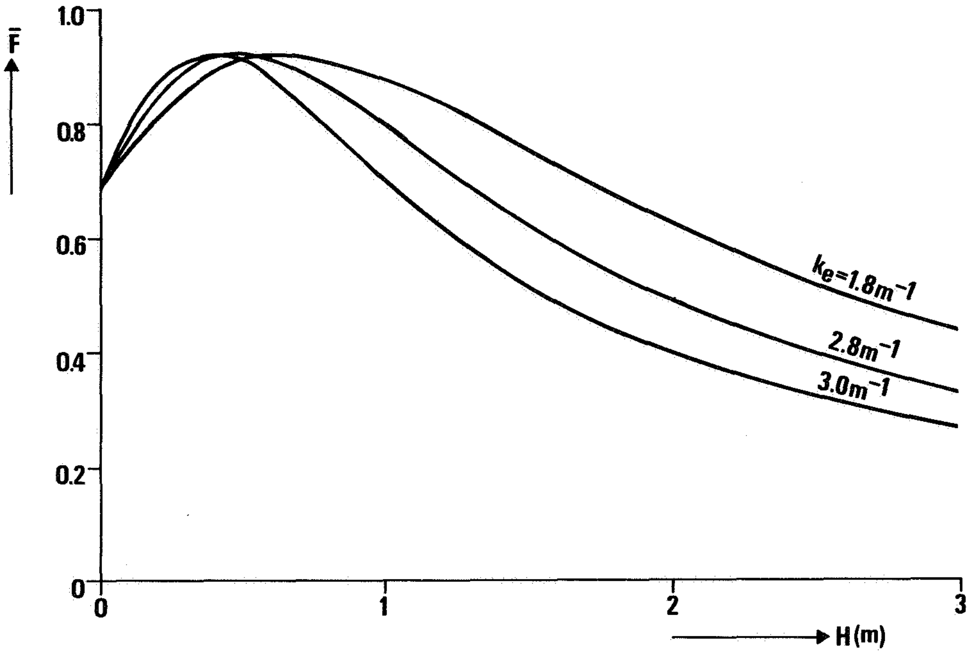


Fig. 3 Lightfactor versus water depth, for three values of extinction coefficient
Parameter: $I_o/I_s = 2.14$ – Berkel River (Van Straten, 1978)

– the light intensity as a function of depth

$$I(z) = I(0) \exp(-k_e z) \tag{13}$$

where k_e is the total extinction coefficient, which is a function of the algae concentration (self-shading) and the remaining suspended solids s

$$k_e = k(s) + \alpha A \tag{14}$$

The depth averaged relative growth rate is computed as

$$F = \frac{1}{H} \int_0^H G(I)\, dz \tag{15}$$

using equations (12), (13) and (14). The factor F ranges from 0 to 1. F is, of course, a function of the time of the day; at night F is zero. Figure 3 shows the light factor as a function of depth, with different extinction coefficients. Parameters have been chosen in accordance with observations in the Achterhoek region (cf Van Straten, 1978). The figure is valid at noon. First, a higher light factor is reached with decreasing depth, because of a more favourable depth average light climate. However, a further decrease in depth reverses the light factor depth relationship, as a consequence of the increasing influence of photoinhibition at higher light intensities.

6.2. *The maximum standing crop*

In lakes, the maximum algal standing crop is reached (Lorenzen and Mitchell, 1975) if

a) the self-shading effect has caused the growth rate to drop so far that it is balanced by the sum of all death rates (including predation by zooplankton)
b) there are not sufficient nutrients available to sustain a higher algae concentration.

Condition a) can be expressed as

$$\begin{cases} \dfrac{d\bar{A}}{dt} = \overline{(k_{pa}F - k_{da})A} \\ \max \bar{A} \text{ if } \dfrac{d\bar{A}}{dt} = 0 \end{cases} \qquad (16)$$

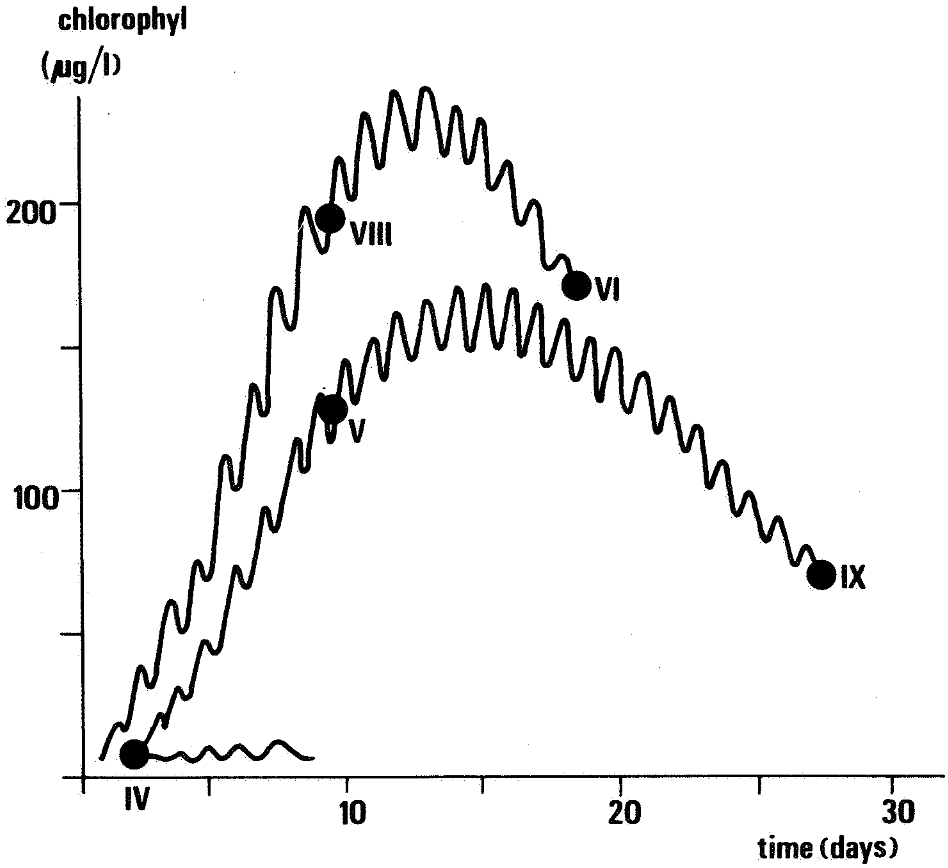


Fig. 4 Development of algae as a function of flow time. Simulations for the River Oude Yssel 1975. The dots represent the residence time in the indicated months. The various curves originate from differences in temperature and light intensity. (Van Straten, 1977)

where the bar denotes averaging over 24 hours. If A does not vary too much over 24 hours (16) is equivalent to the condition

$$k_{pa}\overline{F} - k_{da} = 0 \tag{17}$$

$\overline{F}$ expresses the combined effect of averaging over depth and time. Clearly $\overline{F}$ is a function of the light extinction, which in turn is determined by the self-shading effect of the algae (eq. 14). From eqs. 17 and 14 A_{max} can be solved. It is worth noting that A_{max} is very sensitive for the death term k_{da}, which incidentally can reach high values because of grazing by zooplankton.

In the argument above time did not play a part: in lakes the residence time is usually sufficient to reach the maximum biomass sooner or later. Not so in rivers, however. Here it is quite conceivable that a plug water reaches the river end a long time before the maximum standing crop is reached. It is difficult to derive an analytical solution of equation (16) in order to solve the time required for the maximum to occur. This is a consequence of the complex structure of $\overline{F}$ having A in the denominator. However, a numerical solution can be readily obtained. An example of the effect of residence time is given in figure 4. In this case, the maximum to be expected at the governing levels of phosphorus and light are not obtained with residence times shorter than about twelve days. (Oude Yssel).

6.3. *Discussion*

Based on the sections before the relation of the flow-rate to photosynthesis can be depicted as follows. The biomass developed in a sufficiently long period of stagnation can not be sustained at increasing flow rates, limiting the time available for growth. The population washes out. In addition to this the growth rate usually drops due to increasing turbidity. Run-off from agricultural land sometimes augments nutrient concentration (Beunders, 1978) counteracting the former effect, but only if the nutrients had been limiting before. After the flood wave, initially the transparency increases and the prolonged residence time causes the algal concentration to rise. However, it is conceivable that the composition of the algal population has been changed because of the possible supply of fresh inocculating material from the smaller brooks and ditches during the flood wave. As a consequence, in rivers, the algal dynamics is determined by the hydrodynamics more than it is in lakes. This is apparent in particular in the variability of the DO content, apart from the influence of variable meteorological conditions.

7. POSSIBLE EFFECTS OF CANALIZATION ON WATER QUALITY

Typically, canalization aims at the enlargement of the cross-sectional profile, and at level control by weirs. Consequently the storage capacity is considerably enlarged, giving rise to a serious prolongation of residence time, at equal flow through rate. The water depth

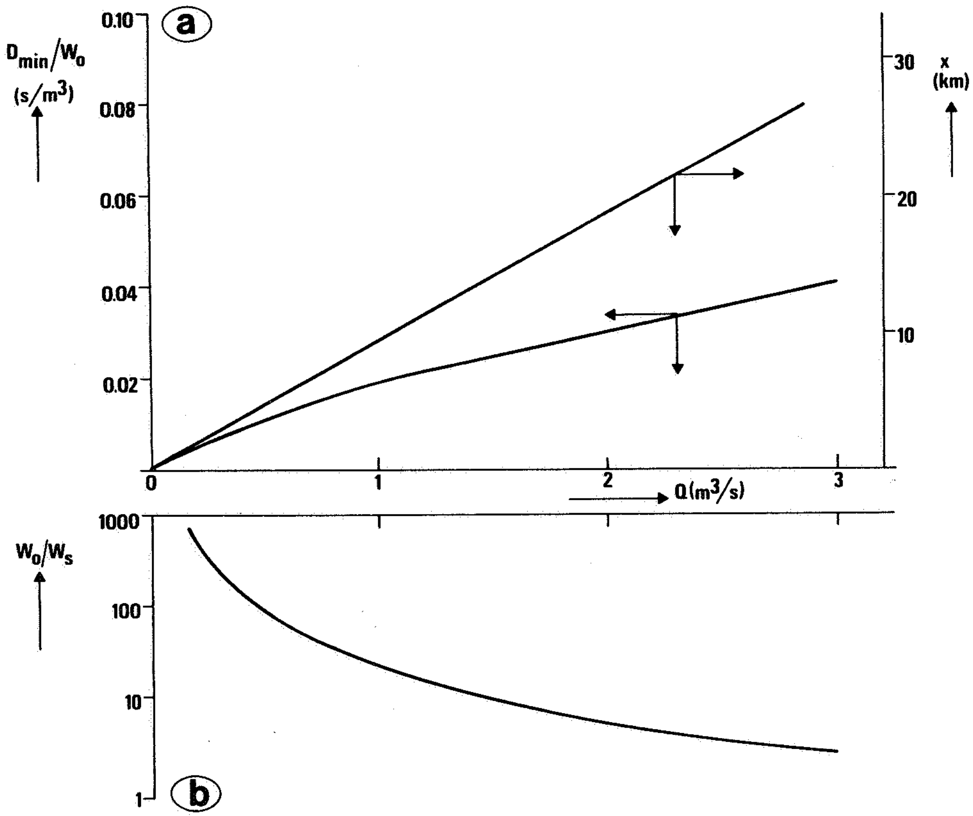


Fig. 5 For a *natural* brook (no weirs):

a) deficit at dissolved oxygen minimum D_{min} per unit waste discharge W_s, and the distance from the source, as a function of flow rate Q.
Parameters:
$B = 10$ m; $Ch = 35$ m $^{1/2}$/s; $S = 2 \times 10^{-4}$ m/m; $k_1 = 0.2$ day^{-1}

b) Ratio of waste discharges permissible in a natural brook and a canalized brook, for a given deficit at equal flow rate. (Compare fig. 2).
Note: Aeration on weirs neglected.

becomes much larger than in the original situation and, furthermore, practically independent of fluctuations in the flow rate. From what is said in the foregoing sections it will be clear that important water quality consequences will have to be expected.

A more detailed analysis involves the application of the computations in section 5 for the situation without weirs. The water depth increases with the flow rate ($H \simeq c\, Q^{2/3}$) and therefore the reaeration coefficient decreases with the flow rate. It should be noted, however, that at a given flow rate reaeration in a river without weirs considerably exceeds reaeration in a weired river section.

Analogous application of the equations from table 3 results in similar plots as examplified in figure 5a. The parameters have been chosen in such a way that a direct comparison with figure 2 is allowed (under reasonable assumptions for the change in a cross-sectional area by canalization. Of course, for any practical case, the true values can be substituted). Inspection of figure 5 immediately reveals a marked difference. In the case of a river without weirs the maximum deficit increases with increasing flow rate. In addition to this the sensitivity to variations in the flow rate is considerably less than in the weired river case. The location of the DO minimum moves faster away from the source with increasing flow rate than in a weired river section.

Because of the much higher reaeration an unweired river has a substantially higher self-purification capacity, at least in the (critical) range of low flow rates. The amount of times the waste discharge could be larger at a free flow for the same deficit to occur is represented in figure 5b. It should be stressed, however, that the aeration capacity of the weirs has not been taken into account. This, of course, renders the picture more unfavourable than is actually justified. It is difficult to properly include this effect because useful knowledge on the aeration capacity of weirs is practically absent. It is our experience that the effect can be substantial. This is also illustrated by a similar calculation for the Doubs River in France (Chalon et al., 1978), where a formula from the scarce literature was employed. Unfortunately, prediction formulae are associated with a high degree of uncertainty. Factors usually not explicitly included but nonetheless important include the construction of the weir and the geometrics of the down-stream river bed where the jet comes down.

Keeping the limitations in mind, a calculation according to the given approach is a useful exercise in river canalization projects. It seems possible to perform at least a suboptimalization with respect to water quality, even at priority of water quantity interests. Decision variables could be the numbers of weirs and their location.

The second aspect of hydrological considerations in water quality that applies to river canalization projects refers to the problem of algal bloom. From figure 3 it can be inferred that the decrease in the growth factor at greater depths will be amply compensated by the prolonged residence time, the more so since river canalization usually involves a widening of the profile of the river bed. In any practical case, simulation can provide an answer as to which of the effects will dominate.

Two final remarks should be made. Clearly, dissolved oxygen behaviour is closely tied to algal growth. If strong oxygen oscillations are considered to be undesirable then a shallow, free-flowing river is in a more favourable position. The stronger reaeration by far equalizes DO oscillations at values close to the saturation concentration. Observations in different sections of the Aaltense Slinge brook demonstrate this point well (Van Iersel, 1977).

A second remark touches on a more fundamental question. It is an implicit assumption in the approach that both weired and unweired river systems can be described with the same set of equations. However, this need not necessarily be the case. For instance, it is known that shallowness frequently promotes the development of benthic algae and rooted plants. Such phenomena are not covered in the models and, in fact, comprehensive hydrobiological research would be needed to provide information necessary for the extension of the model. At present, a rather rough approach has to be accepted. It may be noted that this can be quite satisfactory for management purposes.

8. EPILOGUE

In the preceeding sections the effects of water quantity variables such as flow rate and water depth on dispersion, reaeration and algae growth have been illustrated, using relatively simple models. Examples have been given for situations encountered in the Gelderse Achterhoek region. It has been pointed out that in rivers residence time can be a limiting factor for the algae to reach their maximum standing crop, apart from light and nutrient availability.

Although the approach given above provides a useful picture for many practical cases, certain deficiences have to be noted for subsequent investigation. In short: the role of the sediment, especially in relation to sewerage storm water overflow has not been discussed. No solution has been given for the problems arising from dispersion and incomplete mixing at low flow velocities. The aeration capacity of weirs would merit systematic investigation. No clear cut picture has been obtained as to whether weired rivers can be considered to have their own autonomous algal population dynamics, or whether they are determined more by the irregular inflow of allochtonous material. A modeling approach for the growth of benthic algea, rooted plants or duckweed is lacking. The degree of uncertainty in model description and parameter estimation has not been discussed.

In spite of these limitations it can be stated that integration of present knowledge of water quality processes in water quantity research and management can be expected to be fruitful. Application to river regularization projects is the most obvious example to this effect.

REFERENCES

BANSAL, M.K. (1973) – Atmospheric reaeration in natural streams. Water Research, 7: 769–782.

BECK, M.B. (1978) – Random signal analysis in an environmental sciences problem. Appl. Math. Modelling, 2: 23–29.

BEUNDERS, B. (1978) – Phosphorus household and algal growth in the Barneveldse Brook basin. Masters thesis, Dept. Chem. Eng., Twente University of Technology (in Dutch).

DE BOER, B. (1973) – Quality models for brooks and rivers, in particular the Groenlose Slinge. Masters thesis, Dept. Electric Eng., Twente University of Technology (in Dutch).

CHALON, E., D. MOMAL and C. SABATON (1978) – Effects de la canalisation à grand gabarit sur la qualité des eaux du Doubs. In: Modelling the water quality of the hydrological cycle, IAHS Publication No. 125, pp. 59–67.

DI TORO, D.M., D.J. O'CONNOR and R.V. THOMANN (1974) – A dynamic model of the phytoplankton population in the Sacramento-San Joaquin Delta. Advances in Chemistry, Series No. 106, pp. 131–180.

DYNAMIC ASPECTS OF TROPHIC GRADIENTS IN A MIRE COMPLEX

G. VAN WIRDUM

SUMMARY

The first part of this article deals with the problem of coupling trophic processes to trophic levels of the environment related to the occurrence of organisms. Ecological relationships are grouped into operational (associated with quality), conditional (associated with residence time) and positional or field- (associated with level differences) ones. This is illustrated with a survey of the rise and decline of vegetation of *Stratiotes aloides* L. in a broads area in part 2. The importance of the agricultural use of bordering grasslands is hydrologically stressed.

After an introduction into mire typology in the third part it is outlined that trophic differences between mires largely bear upon hydrological interactions, caused by their water balance. The name poikilotrophic is introduced instead of mesotrophic as intermediary between rheotrophic and ombrotrophic. The causes of trophic gradients are compared with the *Stratiotes* hydrological system.

In part 4 the break down of an originally stable trophic gradient in a large mire reserve area is attributed in part to changes in the hydrological system.

1. ECOLOGICAL RELATIONSHIPS AND TROPHIC STATE

It is a tradition in ecology to distinguish communities whose member organisms are said to prefer a different trophic state of the environment (or substratum). In doing so, frequently a single environmental constituent's abundance is used as a key factor. Regarding concentrations of any specified ingredient in their immediate environment, several organisms are indeed known to be confined to a range of trophic states, sandwiched between a certain (required) minimum and a (highest bearable) maximum. This juste milieu or golden mean range defines the milieu of the organisms as far as nutrient concentrations in the substratum are concerned. This range being known, the organisms can serve as indicators of the trophic state of their immediate environment. The reverse, however, does not hold: not just meeting its milieu according to concentrations enables the fitting organism to settle and survive with respect to nutrient demands, the latter imposing dynamic state criteria. Metabolism for instance withdraws nutrients from the substratum and adds excreta to it. Thus, the immediate environment of the organism would tend to exceed the constraints of its milieu, be it by deficiency of nutrients, or by excess of metabolic wastes. This has to be regulated by supply from, respectively discharge into, any remoter environment.

Limnology therefore recently highlights physical, chemical and biological processes ruling sediment-water exchange in lakes. Even these however do not suffice to account for most prolonged steady states in nature. Especially if peat is formed in a confined body of water, the mentioned supply largely has to come from outside, recirculation of

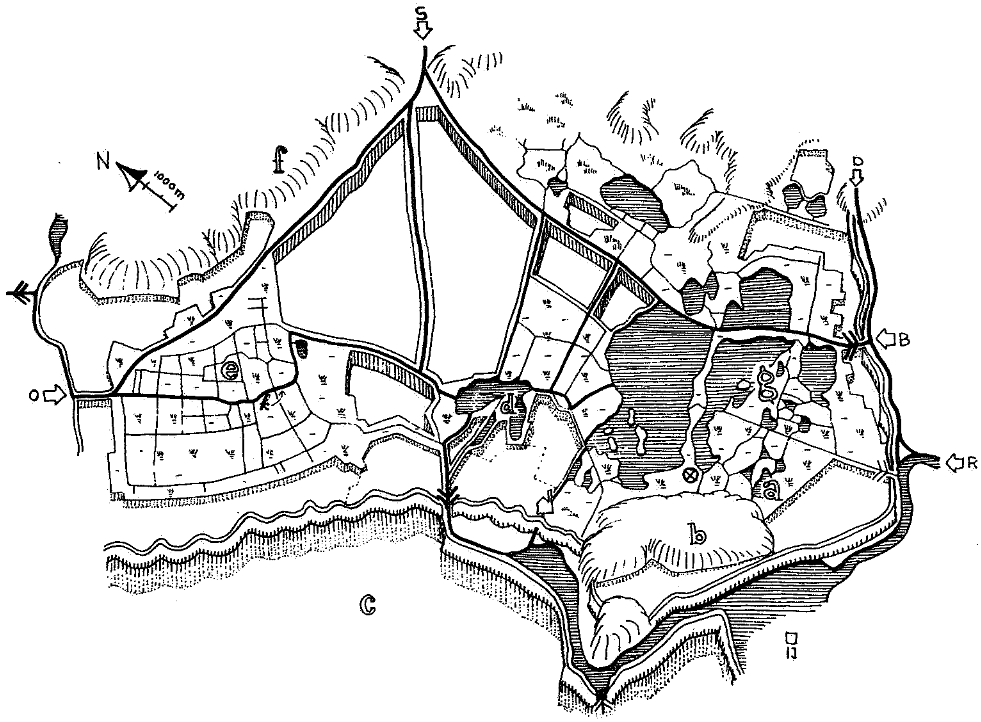


Fig. 1 Hydrographic position of the Northwest Overijssel mire reserves

This map is from 1974 data; no topographic precision is aimed at and any indication of urban area has been omitted.

LEGEND

Formal water levels in polders

- <0.3 m below boezem; partially clay covered
- ca. 0.5 m below boezem; partially clay covered
- ca. 2 m below boezem; reclaimed peat dredgings
- ca. 4 m below boezem; reclaimed from IJsselmeer

Boezem area

- mire vegetation; largely reeds
- grassland
- lakes

Other structural elements

- large canal with locks; points towards higher level
- smaller ditto; most of these omitted
- boundary between polder level classes
- main (former) sea- or riverdike
- pumping station for drinking water withdrawal
- pumping station "Stroink" for boezem discharge

Pleistocene area

- soil surface upto 20 m above boezem

Lettering:
a– Lake Venematen; b – Vollenhove high (boulder clay); c – Noordoostpolder; d – Lakes Zuiderdiep, Giethoornsche Meer and Duiningermeer; e – Nature reserve De Weerribben; f – Paaslo ice pushed ridge (boulder clay); g – Nature reserve De Wieden; k – canal Kalenbergergracht
Arrows:
B – Beukerssluis locks (upto 1973 main boezem inlet); D – Meppelerdiep (Pleistocene discharge); O – Ossenzijl (inflow from Frisian boezem); R – Zwarte Water (locked from rivers IJssel and Rhine); S – Steenwijker Aa (Drenthe Pleistocene discharge); IJ – connects to lake IJsselmeer (Rhine water)

nutrients apparently being curtailed. The deposited peat does act as a regulating structure, interfacing the immediate environmental envelope of certain organisms with the sources and sinks proper of nutrients and poisons. The mentioned relationships will be named and shortly defined as follows:

Organisms, with all their properties, are linked:

a) to their immediate environment by operational relationships;
b) to regulating environmental structures by conditional relationships; and
c) to sources and sinks of the enveloping field by positional relationships.

It should be clear that operational relationships also connect regulating structures to both the immediate environment and sources and sinks in the enveloping field as well as to other regulating structures. Distinguishing additional conditional relationships is worthwhile only if the whole system can be looked upon as contained in a still more extended field, since conditioning is always related to positional factors. In a general hydrological sense, positional relationships in nature reserves bear upon level differences and conditional ones upon residence time or flushing rate of ground and surface water. Operational relationships seem to be made up largely of ingredient supply and disposal. Thus quantity and quality of water are ecologically linked, as will be illustrated here with respect to surface water in nature reserves in the Province of Overijssel, The Netherlands. Some geographic features of the area referred to have been depicted in figure 1.

2. *STRATIOTES* FIELD ECOLOGY

Up to a few years ago, water soldier (*Stratiotes aloides* L.) was an abundant aquatic macrophyte in many lakes in The Netherlands from which it has now nearly gone. In several reclaimed fens with high water levels, *Stratiotes* vegetation is still frequently met with in ditches intersecting the grassland. Water from such places is only rarely found rich in phosphorus (compare De Lange, 1972), nor do the algae actually indicate eutrophic water. Nevertheless such vegetation is known to be quite productive of biomass and formerly it was indeed harvested for manuring arable land. Apparently the plants require an appreciable lot of nutrients although not demanding, and possibly even notwithstanding, high concentrations thereof. Some aspects of the rise and decline of *Stratiotes* and the interpretation of its ecology will be discussed now.

2.1. *Rise and decline of Stratiotes*

According to De Geus-Kruyt and Segal (1973) in 1976 *Stratiotes* plants in Lake Venematen contained twelve times as much phosphorus as the water did (table 1).

As they regarded the lake to have a deficient recirculation of nutrients from the sapropel, they concluded to an exchange with the lake bed. Úlehlová (1970) analysed chemical properties of mud and mud gases of Lake Venematen in the same period.

Table 1 Phosphorus in *Stratiotes* and in water of lake Venematen, July 1967 (according to De Geus-Kruyt and Segal, 1973)

	plants	water
area	2ha	15ha
dry weight	12200 kg	
volume		250000 m^3
concentration of phosphorus	0.4% (P)	0.05 mg/l (phosphate)
weight of phosphorus	49 kg (P)	4 kg (P)

She interprets the muds of the lake to reflect intensive agricultural practice on the bordering Vollenhove high, rendering the lake a higher fertility status than was expected from the relatively low nutrient concentrations in its water. Phosphorus was not studied. Hydrologically in 1974 we calculated seepage to the deep Noordoostpolder and drinking water withdrawals to sum up to 0.25–0.3 mm/24 hrs. Compensation is both by the discharge of the agricultural Vollenhove high and by the infiltration of water from the surrounding "boezem", a surface retention basin into which superfluous polder water is discharged. This exchange of water is highly influenced by wind as is also stated by Lijklema and Van Straten (1977). It is not only in this case that positional and conditional factors governing nutrient levels can contribute to the understanding of the geographical distribution of species and biomass more clearly than operational nutrient concentrations themselves do. This is still more important whenever successional changes in the eco-system outcome are studied, as is dramatically shown by a sequentional analysis of the distribution of *Stratiotes* in the same "boezem" area.

After its disappearance in several ditches in The Netherlands, presumably caused by chemical control (Westhoff and Den Held, 1969), in the past decennium *Stratiotes* has left the scene of nearly all lakes and broads in nature reserves, among which Lake Venematen. At most stations where it still grows in this part of The Netherlands, superficial drainage of grasslands on solid peat or equivalent terrain can be demonstrated to enter the site. In some lakes and canals effluent of houses and small farms is or was effective as well. Most of these stations are relatively remote with respect to the main pathways of the boezem water. Several places where *Stratiotes* has gone now are bordered by grasslands that had been frequently mown for haymaking up to 5–10 years before the plants vanished, some of these grasslands in those days also being grazed by sheep or yearlings and often slightly manured. The trampling resistance had in some cases been improved by fitting a top layer of earth commercially obtained from Frisian terps. At still other stations the decline and disappearance of *Stratiotes* have succeeded to agricultural improvements of the hydrological system. Here peatlands in the retention basin were reclaimed and drained to a lower groundwater level, thus effectively becoming polders. This has been done around the lakes Duiningermeer and Zuiderdiep ca. 1960. Figures 2 and 3 prove the obvious vegetational change in these lakes. Remarkably vital *Stratiotes* vegetation is still met with in this area in the grassland ditches, particularly where formal

Fig. 2 Vegetation in lakes Duiningermeer and Zuiderdiep, summer 1949.
All grassland drained to *boezem* water level.

Fig. 3 Vegetation in lakes Duiningermeer and Zuiderdiep, summer 1971.
As a measure of land improvement, much grassland has been poldered. Lake vegetation is much reduced and more so since 1971. Numbers are formal drainage and water levels, referring to sea level (cm). Uppermost number: summer period; below: winter. Dashed lines roughly indicate polder boundaries.

Fig. 2 and 3 Lettering: B – *Scripus lacustris* L,: D – floating swards of *Typha angustifolia* L.; O – submerged aquatic macrophytes; P – *Nuphar lutea* (L.) Sm.; R – *Phragmites australis* (Cav.) Streud.-reed; S – *Stratiotes aloides* L.

To provide a correct impression of the relief, the photographs have been reproduced with the southern lowering of the water table has been absent or moderate. Admittedly, also there levels have actually gone down, seepage to the Noordoostpolder being effective since ditches have been dammed off or become filled up with vegetation.

Far not always *Stratiotes* had for a very long time been growing at the stations left now. Often a quite explosive rise of the species was noted, like water hyacinth (*Eichhornia crassipes* (Mart.) Solms) is reported to perform in tropical waters. This spread correlates in part (Westhoff and Den Held l.c.) with the increasing use of fertilizers in agriculture, but this is clearly not the only cause in Northwest Overijssel. Here *Stratiotes* reached its maximum spread in the early fifties. Some of its stations were in parts of lakes that had been vegetated with reedmace (*Typha* spec.) before the '40–'45 war. These mats of vegetation had probably disappeared as a consequence of extraordinary flooding during wartime. According to our hypothesizing minerals were freed afterwards and gave rise to the expansion of *Stratiotes* vegetation. Analogously this plant sometimes quickly establishes itself in newly dug pools in somewhat earthened peat soils. However, in none of the recently dredged out old peat diggings in the centre of the Weerribben nature reserve this phenomenon has been convincingly observed, whether the mud had been left on the banks or not.

2.2. *Ecological peculiarities of the water-land interface*

Although in the last paragraph positional relationships of *Stratiotes* to its environment have been suggested, the traditional picture of research and thinking on the rise and decline of this species has focussed on operational relationships. Thus eutrophication of boezem water and infection with viruses or fungi have frequently been stressed as alternative causes. We have in contrast observed that the plant is able to grow normally at some very eutrophicated stations and we are convinced that a possible infection's deteriorating power must be favoured by causes less operational towards the plants. Consequently these operational hypotheses stand in need of positional and conditional support. The slogan 'prevention is better than cure' then urges conservationists to find and fight these supporting deficiencies for managerial practice. Although it is not our aim here to answer the questions after the causes of the decline of *Stratiotes*, some of the mentioned material will be ordered now into a rough interpretative framework.

Figure 4 schematically outlines the hypothesized mechanism of a *Stratiotes* station. Natural, superficial drainage, influenced by the acidity of rainwater and the anaerobic conditions in the slightly manured peat banks, supplies the water around the plants with nutrients (see also Steenvoorden and Oosterom , 1979). The larger volume of this water, however, originates from the excess groundwater in the Pleistocene massives of Drenthe and Vollenhove and typically shows high calcium shares. Now physico-chemical equilibration in the water-mud system, interacting with the plant uptake and carbon dioxide metabolism, will, for instance, produce a steady supply of phosphorus to the immediate plant environment. In fall the young overwintering buds sink down, but the older plants are

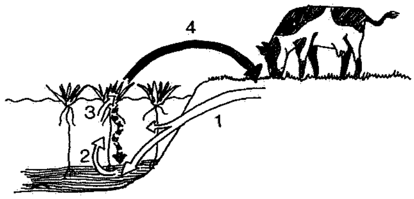


Fig. 4 Environmental relations at a *Stratiotes* station. Explanation in the text.

dredged and spread over the bordering land. Thus a forced recirculation is effectuated, in the mean time maintaining the differences between the lake and the land, and the interfacing functions. In other words: the field gradient is kept in position by managerial practice.

If the old plants are allowed to sink down, processes, characteristic to the thickening mud layer, will once give rise to changes in the vegetation, leading to paludification or, more in particular, to the forming of a fen. The reason conservationists feel the *Stratiotes* decline as a problem is not caused by the mere fact they want to save this species (a very old one by the way). To them it is most alarming that apparently no other plants except the very common yellow and white water lilies (*Nuphar lutea* (L.) Sm. and *Nymphea alba* L.) nowadays take over. Thus no fen is formed. The diagnosis of this illness runs as follows:

a) Changes in positional relationships

The ion composition of boezem water as well as the total amount of solubles has shifted from the calcium rich groundwater type to modern Rhine water, resembling somewhat diluted sea water in the major ions: in a qualitative hydrological sense the present position of the stations is closer to the ecosystem's sink. For the western part of the area only it holds true the seawater type proper to the former Zuiderzee seabosom was more important in earlier days. In connection with the damming off of the Zuiderzee the reclamation of the Noordoostpolder may thus originally have favoured the expansion of *Stratiotes*. Next, the bordering agricultural lands are either kept free of manuring now as a conservationists' measure, or their discharge is diverted from the lakes. In some cases they even attract the lakes' water now, their own levels having been lowered several decimeters.

b) Changes in conditional relationships

Dredging the old *Stratiotes* plants has been stopped, or, in some cases, advanced to the summer. This has been done in years the only serious *Stratiotes* problem was the explosive filling up of lakes and canals, impeding all boat traffic.

Interestingly the species seems to get up at certain places where the field has as it were diametrically turned around the station regarding the supposed phosphorus supply. Thus in nature *Stratiotes* seems to recognize its milieu also at the meeting place of the drainage water of large mire areas and richer, riverine, waters.

A last remark should be made concerning the effects of "impoverishment" of the nutrient supply. This does not always cause a species to leave the scene via the minimum of its milieu. If disharmony in the nutrient supply through deficiency of any ingredient results in diminishing populations or growth, the excessive uptake of another constituent, against which the species has no resistance, might be poisonous.

3. A CHAIN OF MIRES

It is a long tradition again to use the trophic levels concept in the study of mires. Most mire typological systems are based on a complex of geomorphological, botanical and ecological characters. For the purpose of this paper we will consider some aspects of the classification of mires in which peat is formed: peatlands. Morphologically (raised) bogs (hoogvenen, Hochmoore) were early recognized as a separate group. Only far later the raised character reflected in the syllable "hoog" was no longer understood from the name and the group was redefined comprising the peatlands having a raised position above the regional groundwater table. According to this idea, the other peatlands were named low (laagvenen, Niedermoore), or, in the English nomenclature, fens. Thus for instance the Atlantic blanket bogs are included in the first class although they differ morphologically from the Central European plateau raised bogs. Bogs are characterized by a lawn of peatmosses (*Sphagnum* spec.), the abundance of *Ericaceae* and the near absence of trees. Fenlands in general lodge *Phragmites, Juncus* and *Cyperaceae* reeds as well as carr vegetation. Some criteria of the classification systems will be discussed in the next paragraphs in advance of a discussion on the hydrological relationships of an intermediate type. A comparison is made with *Stratiotes* field ecology.

3.1. *Trophic principles in mire typology*

Many authors have stated bogs to be oligotrophic and fenlands eutrophic according to concentrations of minerals in the peat's water. An intermediate category (mesotrophic) is claimed for the so called transition mires, representing a transitory stage, and for a group of peatlands often found spatially wedged in between bogs and fenlands as well. This static concept of trophism is more or less mirrored by the biomass production of peatland vegetation.

In the beginning of the past century, the Danish lawyer Dau (fide Overbeck, 1975) assessed bogs to be exclusively fed by rain water. Despite this point of view being fought and neglected a long time, it has ultimately invaded in nearly all classificatory systems by naming bogs ombrogene, ombrophilous or ombrotrophic. Although ombrotrophy could actually be interpreted from water budget studies, in the practice of peatland science it has been pinpointed as a static trophic level, just preserving the semantic benefit of the ombro-root. This is shown by Du Rietz' (1954) "Mineralbodenwasserzeigergrenze". The chemical reliability of this plant indicated limit however is problematic. Fenlands in consequent contrast are called minerotrophic in these systems.

Only in the last thirty years further hydrological principles have been taken into account in mire typology clearly. Kulczynski (1949) from Poland pioneered. In the then vast and virgin Pripet peatlands he found traditional peatland types pronouncedly tied to the local hydrological cycle. On the watersheds precipitation fed raised bogs were prominent. He named them ombrophilous. Nearer to the draining streams fenlands were met with. Since according to his logic water should pass these peatlands, he introduced the type adjective rheophilous. Moore & Bellamy (1974) suggested to preserve these groups under the names ombro- and rheotrophic, linked by a mesotrophic type. In doing so they wish to underline that nutrient budgets bridge the rather unexplored territory of relations to botanical peatland classes. This is accentuated by the observation of pronounced similarities throughout rheotrophic peatlands, no matter whether their waters are eutrophic or oligotrophic. The other way round, important differences can be seen between peatlands with "the same water" but this having different residence times or flow rates. This criterion is also stressed by Ingram (1967) and others.

3.2. *Position and feeding mechanism in mire typology*

Independent of the afore-cited, Moore & Bellamy gave typological principles based on quantitative hydrology in a geographic sense. This leads to recognition of primary, secundary and tertiary mires at the hierarchically highest level. Primary mires (fig. 5a) are formed in a depression of the land surface under the water level. They fill in a basin and thus by sedimentation and peat deposition diminish the water retention capacity of this basin. As soon as peat formation continues above the original confines of the water level, retention is enlarged and the water level rises. This applies to secundary mires (fig. 5b). A tertiary mire also adds to the retained water mass, but now this is done above the

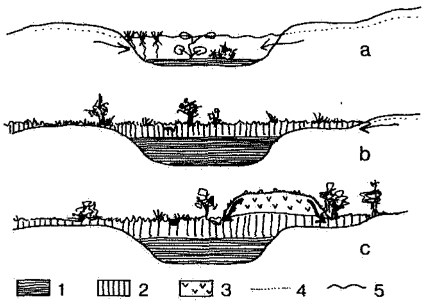


Fig. 5 Mire basin types: primary (a), secundary (b) and tertiary (c)
Arrows indicate general flow directions.
Legend: 1 – peat or sediment formed in primary basin period; 2 – secundary mire peat; 3 – tertiary mire peat; 4 – groundwater level if below surface; 5 – water level.

drainage base of the geomorphologic basin, the origin of the added water being just local precipitation (fig. 5c). This class is identical with the bog's one, and ombrotrophic. The authors are careful to mention that primary and secundary mires both can be rheotrophic as well as mesotrophic. We will now go into some details of differences and similarities in these two classification systems.

All continental water, as far as it is not evaporated, is on its way to deep groundwater systems with long residence times or to the sea. All basin filling mires at first glance therefore will likely be considered rheotrophic. If the catchment area is small, the water may well be mesotrophic or even oligotrophic for mineral concentrations. This occurs in heathpools and bog hollows. As precipitated water will immediately become mixed with standing water in the pools, here no organism formally can exclusively feed on 'individually caught' rainwater. As soon as a floating mat of vegetation has developed in a primary basin, some stations will be given a chance to maintain a proper ombrotrophic character during part of the year at least. In other periods these stations certainly will receive water from under the sward of vegetation. During precipitation excess such stations can indeed show a manifestly perched water table, whilst during deficiency the whole sward sinks down with the water level. This combines to an active primary basin, eventually becoming filled up with peat that has been first deposited above the basin's main water level, in a secundary top layer. Such a floating sward of vegetation can even host miniature tertiary mires.

In a tertiary mire the water level is always at least a bit higher than the immediate surrounding one, causing any excess precipitation not absorbed in the moss layer to be discharged. In large bogs compression of the peat takes place, as it were squeezing out some of the aged peatwater. Sometimes this does lead to "bog bursts" (Casparie, 1972). Because peatmosses actively exchange ions with the precipitation water and because other biological and chemical processes are effective in the inner retention volume of a peat body, this squeezed out water chemically is not similar to rainwater. According to this, a bog functions as a source of qualitatively distinghuishable water in a relatively eutrophic and less acidic environmental matrix. Reversely groundwater cannot intrude the tertiary mire proper: flow of groundwater, if present, passes underneath the mires' "mound of water". Consequently, in between the rheotrophic and ombrotrophic extremes a zone exists, owing its character to this intermediary position. This has been shown in figure 6, illustrating a gradient of the local water quality contrast to general properties of ground- and surfacewater. The contrast is most prominent in the centre of the bog and fades out towards the rheotrophic area. The bog as it were nourishes a potentiometric field of water ion composition: it is a positionally active field source. Hydrologically, water level differences are the positional guide characters. The hydraulic head of the tertiary mire centre is fundamental to the quality gradient regulation. The main conditional element from the quantitative point of view, basic to all further qualitative ones, is the residence time in the peat mass of the secundary mire and thus the peat's structural features.

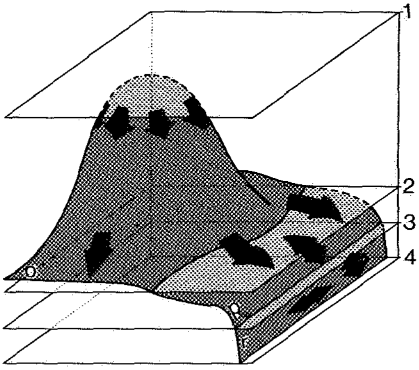


Fig. 6 Water quality contrast in a mire system.
Base is a part of the earth surface. Parallel planes depict quality positions of rain water (1); surface water (high water level) (2); ditto (low water level) (3); and very eutrophic waters (4). Mutual hydrological impacts of the ombrotrophic (o), poikilotrophic (p) and rheotrophic (r) zones are arrowed. Note the similarity to expected contrasts of heights and hydraulic heads!

The way the mesotrophic area has been depicted here as resulting from positional and conditional ecological relationships, more than from operational trophic concentration, merits a different type name. It is proposed here to nominate this poikilotrophic, referring to the Greek root poikil-, meaning "variegated", "wrought in various colours", "diversified", "subtle", etcetera. It is here primarily intended to indicate the two-way positional relationships, although in the next paragraph the applicability to species diversity and vegetation structure of the poikilotrophic mire type will be underlined.

It can be concluded that the discrete classification of mires into primary, secundary and tertiary ones provides a general reference to the overall properties of an actual mire trophic field. Within the so positionally classified basins, each locus in the trophic field is characterized by the effectiveness of feeding sources: rain- versus sea- or groundwater. Although they are not too useful as factor names, the type names ombro-, poikilo- and rheotrophic mire are worthwhile to be retained at this level, extreme ground- and seawater feeding both leading to the rheotrophic type. At a next hierarchical level trophic state with respect to nutrient or other mineral concentrations can keep its place in peatland classification under the names of eu-, meso- and oligotrophy.

3.3. *The rainwater-groundwater gradient*

In general poikilotrophic mires are characterized by a lawn of more or less brown coloured mosses of the *Hypnaceae* family (s.l.) and by many species of slender-leaved sedges. The community names brownmoss and *Parvocaricetea-fen* are appropriate. A lot of rare flowering plants do occur in these communities too. Over 70 plant species per square meter is not exceptional, which means there is a high degree of species diversity.

This richness in species is effectuated by small differences in the ecological peculiarities of neighbouring mire stations. As vegetation variability over the years is very small, we accept the idea of each individual station's specific properties being maintained. This obvious stability of the abiotic factors is thought to be caused by the reliability of the field gradient in which the mire itself and, particularly, its peat, function as "simple" conditional structures. Then, the operational differences between adjacent mire stations result from physico-chemical interactions involving the contacting of waters originating from the rheotrophic, respectively the ombrotrophic pole of the system. Conditions are set by relief determined differences in oxigenation and by exchange processes with the peat itself. By human logic, it must be impossible to actually determine the dynamic state of all the operational factors any specified organism meets at any specified station. However, we can recognize its specificity by the occurrence of the species concerned, as this is bound to be sensitive to these factors. Implicitly, it must be impossible to purposefully design and locally carry into effect any managerial tactic towards the conservation of poikilotrophic mires "at the species' side". The only real solution to this problem is securing the reliability of the field structure. It is there that we need the raised bog as a source regulator in the mire chain. The relatively low species diversity of this type reflects the structural simplicity so characteristic of rugged regulators.

Apparent dissimilarities of the *Stratiotes* field structure and the *Parvocaricetea's* one are in the gradient, the former being a clearcut limes convergens (Van Leeuwen, 1966), the latter a stretched and faint limes divergens. In the *Stratiotes* case history the system's source was the eutrophic field component, dedicated to push forward per se. Only if such a source were a small and local "muck heap" in a vast and oligotrophic environment, it would not persist without minerals being fed into. The moss family of *Splachnaceae* provides a striking example of precise fitness of a lot of rare species, each to the excrements of a specific mammal, to owl's pellets and other such things, presumed these were dropped in an oligotrophic stretch of land, like deserts and bogs are. As from most Western European countries, they have almost disappeared from The Netherlands. In countries such as Ireland and Norway they find a last refugium. The majority of recent *Stratiotes* stations shows the reverse: The Netherlands being turned into a large dunged territory, the gradient's slope towards oligotrophic places is not durably maintained. The poikilotrophic mire zone on its turn is typically preserved by the predominance of an ombrotrophic source component.

4. THE FLATTENING OUT OF A TROPHIC GRADIENT IN NORTHWEST OVERIJSSEL

Field analysis in the paludisphere yields hooks to hang the story of last century's changes of the hydrological system of Northwest Overijssel and their ecological effects. Some main lines will be drawn now to illustrate the application of theories to a conservationist's everyday experience.

In figure 7 quantitative hydrological relations in a part of the nature reserve De

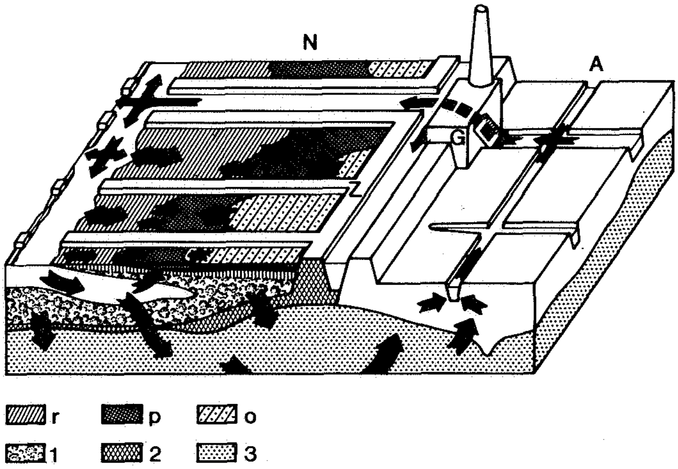


Fig. 7 Hydrological relations of the nature reserve De Weerribben.
Arrows are directions of water flow.
1 – mud; 2 – remaining peat; 3 – mineral bottom; A – agricultural polder area; N – nature reserve; G – pumping station for polder discharges; Z – peat banks between dredging lakes. Other letters: see figure 6.
In general dredging lakes are ca. 30 m wide and 2.5 m deep to the mineral bottom. They can be several hundreds of meters long.

Weerribben have been schematized. It shows the seepage of water to neighbouring polders, reclaimed from the area of peat diggings. Pumping stations discharge this water back into the boezem system. The water in this system originates from higher Pleistocene regions and from the adjacent Frisian boezem.

The Frisian boezem on its turn, is fed with Rhine water, let in from the large lake IJsselmeer. Via a network of canals and ditches the water can penetrate into even the most distant parts of the mire reserves, "compensating" for seepage and evaporation losses.

In the rectangular lakes left by the peat dredgers, floating mats of vegetation locally

Table 2 Some water analyses; explanation in the text

sampled type		µS/cm	meq/l	mg/l									mg 0/l
	pH	EC_{25}	alk	Cl^-	$SO_4^=$	Ca^{++}	Mg^{++}	Na^+	K^+	P_{flt}	$NO_3^- - N$		COD
ombro	5.1	105	0.0	20	17	2	1	9	9	0.16	n.d		129
poikilo	6.1	115	0.3	8	23	5	1	4	1	0.02	n.d		45
rheo	6.7	400	2.1	56	3	29	4	32	n.d	n.d	–		–
boezem	7.8	715	2.3	88	80	47	13	48	6	0.11	1.57		38

retain some rain water above the surrounding water level. The stable trophic gradient of a bog-sourced field is reproduced here in miniature. In this gradient, poikilotrophic communities occur, and they actually belong to the most praised quagmires of The Netherlands. It should be noted that also the rheo- and ombrotrophic communities are formed on swingmire. To illustrate the correlation of water and mire type differences, in table 2 some water analyses from the area have been gathered.

Most values can be interpreted in the direction of the gradient: they are high in the rheotrophic zone and especially in the boezem system. Filtered phosphorus (0.45 μm pores) is anomalous however. This could be caused by the acidity and the anaerobic conditions in the ombrotrophic area, but it certainly also reflects the high level of biomass production in the concerned rheotrophic sample station, a *Stratiotes* one. If compared to site based measurements, the reported acidity in the ombrotrophic specimen is weak. Nitrate N was only detected in the boezem sample. COD and potassium, and sulphate less so, show the same anomalous trend as phosphorus, suggesting potassium might be a minimum factor in the productive environment, but probably not in the poor one. The boezem sample has been given as a reference. The figures depict means of the concentrations in 25 samples, taken during the period 1972–1977 at Ossenzijl, where the Friesland water and part of the Drenthe water enters the Vollenhove boezem system. In contrast to others, the boezem stations vary highly and irregularly in ion content during this period.

As discussed before, the dissimilarity of peatland sites and stations is maintained by the conditions imposed by the "living", so spreading and thickening, peat body. Thus, in an active mire area, chemical properties of the water continuously diverge. An analysis of the ion content of Weerribben waters during the last six years however, has shown converging tendencies. This concerns the typological distinctness of mire parts with prevailing ombrotrophy, as well as the formerly pronounced differences in the boezem system. These local dissimilarities and regular variations of mineral concentrations in the boezem water were bearing upon the share of waters with a different origin, being larger groundwater reservoirs or the brackish Zuiderzee seabosom. The Zuiderzee was dammed in 1932 and then renamed IJsselmeer. In the early 40's the Noordoostpolder was reclaimed from this lake. So in the old days there was not only the gradient between ombrotrophy and rheotrophy, but within the rheotrophic matrix there was also a limes divergens interfacing brackish water and discharged groundwater. Probably the collaps of this complex field has been favoured by conditional feedback, since the disappearance of dense vegetation of aquatic macrophytes must have enlarged the overall transmissivity of the canals network in the boezem. Moreover, starting from ca. 1950, more than one half of the reeds of the Weerribben reserve is under irrigation by means of windmill or motor driven pumps as a culturing strategy, the reed being mown for thatching and other purposes. Thus, rain water retained in the ombrotrophic nests and matured in the mire system during a dozens of years contact with peat and peatmosses, is spoiled and refreshed with water from the system's sink. The reeds themselves are favoured by this

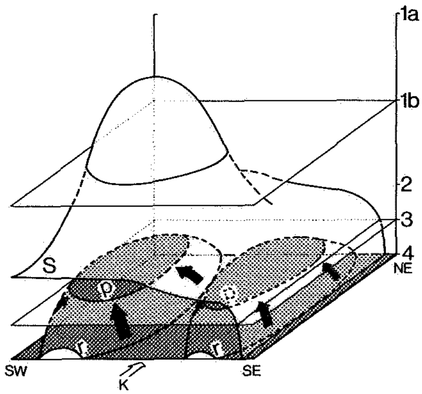


Fig. 8 Water quality contrasts in the Weerribben reserve area, Subboreal quality field compared to present one.
Compass indications in the bottom plane. Lettering: K – Kalenbergergracht (large canal); S – Subboreal field; 1a – Subboreal rain water quality; 1b – present rain water quality; other – see fig. 6.
The present state of the mire's water quality field has been toned. Level 3 is the quality in isolated ditches and lakes. Note hydraulic heads are lower there at present than at stations of level 4 water quality!

practice, common reed (*Phragmites australis* (Cav.) Streud.) being a characteristic species of the rheotrophic zone.

Figure 8 is meant to show the dissimilarity between the trophic gradient in the Weerribben area in the Subboreal period and the present situation. Note that the Subboreal state was taken as representative in figure 6. It is reconstructed from the plant remains in the peat and roughly drawn after soil information in Veenenbos (1950) and Haans and Hamming (1962). The recent state is an abstraction from about 500 water analyses. The dissimilarities can be grouped and shortly discussed as follows:

a) Field positional properties

The Subboreal ombrotrophic pole was characterized by a high hydraulic head, a large and gradually expanding volume, a central position and a low mineral content in comparison to the Subboreal rheotrophic pole and the present ombrotrophic component as well. Now, the ombrotrophic zone even has a lower hydraulic head than the rheotrophic one as a result of the seepage to several polders and the system's resistance to lateral water supply. Besides this, it has a small volume, a position, dispersed over several small and hardly growing nests (the majority even more and more reduced). It is imposed on a rheotrophic matrix of a much higher mineral content than the Subboreal one and of only one water type.

b) Properties of the conditional structure

The Subboreal poikilotrophic area must have been dissected in a zoned drainage pattern. At the raised bog side a lot of small, partly subterraneous and only now and then discharging gullies and bog rivulets according to our imagination existed; at the rivers and sea side larger brooks and lakes have been traced. Nowadays, the whole area is dissected in blocks by navigable canals and each block on its turn by narrower ditches. The presence of large open lakes shows a certain zonation: roughly speaking they are more abundant at the margins of the reserve. The resulting high overall transmissivity of the mire territory is much appreciated with respect to the boezem's function to the agricultural water management of the region. Thus large polder discharge volumes can be disposed of in a reasonable time. However, towards this aim the design and management criteria seem to be needlessly spacious. Towards the conservationists' aims the effect is a deleteriously high refreshing rate.

c) Operational properties

Guessing from the occurrence of critical plant species, at present operational differences within the mire are not pronounced. Moreover, operational environmental complexes deviating from the most common type are rare. The poikilotrophic mire area therefore is less variegatedly shaded.

As to the actual causes of the changes involved in the briefly described historical contrasts, among other things attention can be drawn to the peat dredging industry (now abandoned), the maintaining of fixed water levels in the boezem by letting in Rhine water, the construction of deep polders highering water losses, the irrigation of reeds, and, to some extent, recreational activities. In particular the deterioration of resisting and retaining conditional structures of the system has accelerated the flattening out of field differences.

5. ACKNOWLEDGEMENTS

In the context of this paper, it has not been possible to present detailed data on vegetation, quantitative and qualitative hydrology and theoretical principles fundamental to it. Terrain data have been obtained by the author with the help of many students of several institutes of education in The Netherlands. This would never have been possible without the efforts (finance, equipment, assisting personnel) of those, in particular, who saw the expenses of temporary appointments. Of all of these, I can only mention here the Hugo de Vries Laboratory of the University of Amsterdam, the Department of Agrohydrology of the Agricultural University of Wageningen, the Training Centre for Forestry and Land Development of Velp and the Governmental Forestry and Conservation Service of Zwolle. My own participation and the lift off of the whole enterprise were facilitated by a grant of the Organization of Pure Research in The Netherlands and its spin-off BION. Although my present work at the Research Institute for Nature Management seriously retards concrete occupation with the obtained data, I much acknowledge its Botany Department's scientific atmosphere,

stimulating the draft of the theoretical mainframe referred to. A more careful treatment of data, theory and acknowledgements will be published later on.

REFERENCES

CASPARIE, W.A. (1972) – Bog development in Southeastern Drenthe (The Netherlands). Vegetatio, Acta Geobotanica 25: 1-271.

DU RIETZ, G.E. (1954) – Die Mineralbodemwasserzeigergrenze als Grundlage einer natürlichen Zweigliederung der Nord- und Mitteleuropäischen Moore. Vegetatio, Acta Geobotanica 5/6: 571-585.

GEUS-KRUYT, M. DE and SEGAL (1973) – Notes on the productivity of *Stratiotes aloides* in two lakes in The Netherlands. Pol. Arch. Hydrobiol. 20 (1): 195-205.

HAANS, J.C.F.M. and C. HAMMING (1962) – Over de bodemgesteldheid in het veengebied in het Land van Vollenhove. Mimiogr. rep. Stiboka 583.

INGRAM, H.A.P. (1976) – Problems of hydrology and plant distribution in mires. J. Ecol 55: 711-724.

KULCZYNSKI, S. (1949) – Peatbogs of polesie. Mem. Ac. Sci. Cracovie B: 1-356.

LANGE, L. DE (1972) – An ecological study of ditch vegetation in The Netherlands. Thesis University of Amsterdam.

LEEUWEN, C.G. VAN (1966) – A relation theoretical approach to pattern and process in vegetation. Wentia 15: 25-46.

LIJKLEMA, L. and G. VAN STRATEN (1977) – Water movement in the lake district of North-West Overijssel. H_2O 10 (16): 360-363 (in Dutch).

MOORE, P.D. and D.J. BELLAMY (1973) – Peatlands. London.

OVERBECK, F. (1975) – Botanisch-geologische Moorkunde. Neumünster.

STEENVOORDEN, J.H.A.M.. and H.P. OOSTEROM (1979) – Natural and artificial sources of nitrogen and phosphate pollution of surface waters in The Netherlands. Comm. Hydrol. Onderzoek TNO, Den Haag. Versl. en Meded. 25.

ULEHLOVA, B. (1970) – An ecological study of aquatic habitats in North-West Overijssel (The Netherlands). Acta Bot. Neerl. 19 (6): 830-858.

VEENENBOS, J.S. (1950) – De bodemgesteldheid van het gebied tussen Lemmer en Blokzijl in het randgebied van de Noordoostpolder. De bodemkartering van Nederland, deel V, V.L.O. 55.12.

WESTHOFF, V. and A.J. DEN HELD (1969) – Plantengemeenschappen van Nederland, Zutphen.

THE RELATION BETWEEN WATER QUALITY AND WATER QUANTITY IN THE CHANGING DELTA AREA OF SOUTHWEST NETHERLANDS

B.A. BANNINK

SUMMARY

The dependence of water quality on water management is specifically strong in the Delta area of Southwest Netherlands since the Deltaworks will result in a complex of waterbasins of different types, separated by large dams, but interconnected by water management instruments, like shipping-locks and waterexchange sluices. It is advocated that water quality management enforced by quantitative water management should be based on basincharacteristics, like basinmorphology or basinspecific water quality processes and on consequences for adjacent basins. Examples of the water quality-quantity relationship are discussed for the policy-analysis phase of construction-alternatives, for actual management of the new basins and for the environmental guidance during the construction phase of the dams in the Eastern Scheldt.

1. INTRODUCTION

In recent years the concept of water quality has been extended more and more, notably as a result of the increasing pollution of surface water. It is therefore necessary to examine new aspects of the relation between water quality and water quantity or water management. Initially the only relation considered was that between water management and the chlorideion content; later the relation has been extended to include biodegradable substances in connection with the oxygen balance. From there it was only a small step to algae and dissolved nutrients. At present water quality is also being studied in relation to suspended sediments, xenobiotic substances and thermal pollution.

Particularly in the Delta area of Southwest Netherlands, the relation between water quantity and water quality forms part of the broader relation existing between water management and the natural environment in the area outside the dikes. Not only does water management affect the composition or quality of the water: the water level and its variation can also influence the potential development of the shores, i.e. littoral vegetation, geomorphological processes, birdlife and bottom fauna.

This paper will give some examples of the relation existing between the water management, which is changing as a result of the Delta project, and the natural environment outside the dikes, though the main emphasis will be on water quality. Where possible, model results will be presented, though no detailed discussion of their formulation will be given.

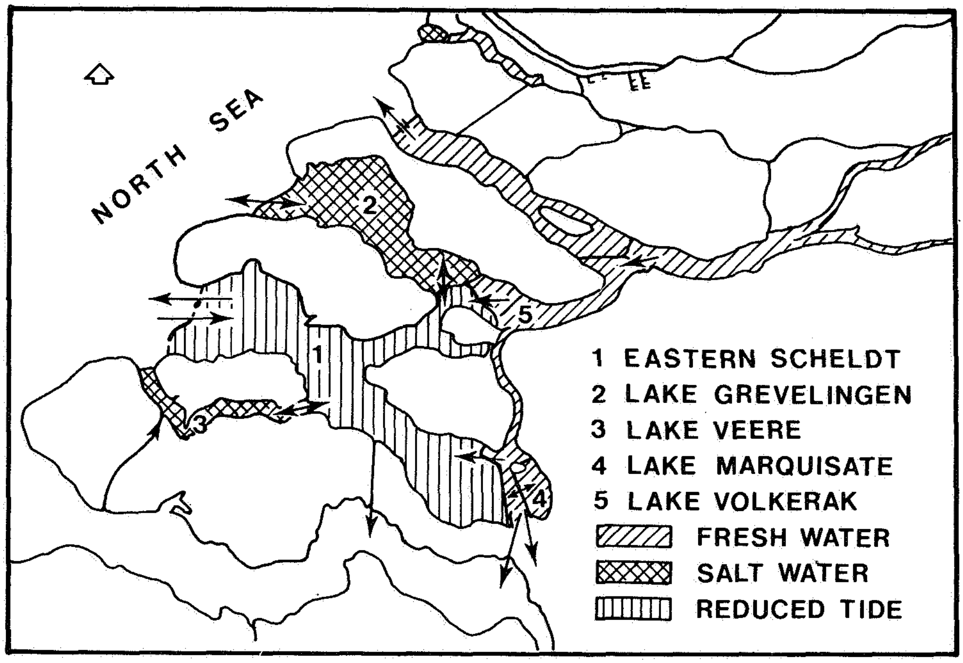


Fig. 1 The present Delta Plan.

2. DESCRIPTION OF THE PRESENT SITUATION

2.1. *The Delta Project in relation to the instruments of water resources management*

Once the Delta Project is completed, the former estuarine area of the Rhine, the Meuse and the Scheldt will have become an interconnected complex of basins, that are diverse in type (fig. 1). It has still to be decided which type two of the basins are to be: in the case of Lake Veere a choice will be made between different brackish water systems, while for Lake Grevelingen the choice lies between salt or fresh water.

The works in the Eastern Scheldt will be the final stage of the Delta Project: a storm-surge barrier is to be constructed at the mouth of the Eastern Scheldt together with two secondary partitioning dams, the Philips Dam, between St. Philipsland and the Grevelingen Dam, and the Oyster Dam, between South Beveland and Tholen; these will serve to seperate the Eastern Scheldt from Lake Zoom (fig. 1 called Lake Volkerak) To serve shipping, traffic locks will be built in the Philips Dam and the canal through South Beveland will be widened. Water management will be facilitated by a discharge canal from Lake Zoom to the Western Scheldt along the Rhine-Scheldt Canal.

The whole system of dams, sluices and canals separates and connects saltwater,

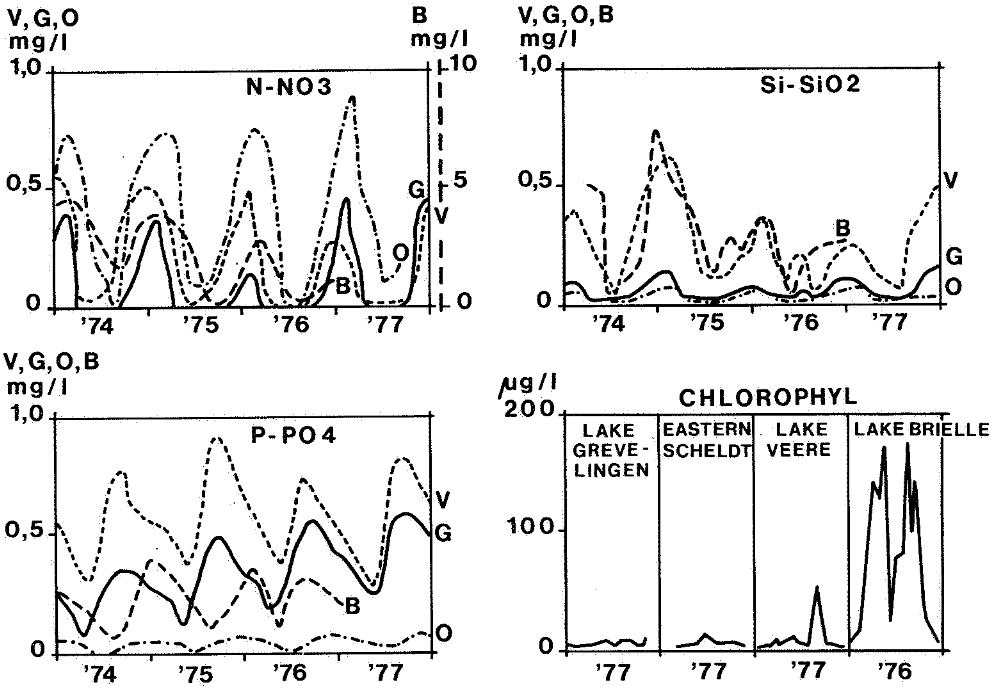


Fig. 2 The averaged pattern of nutrients and chlorophyl in some Delta-waterbodies.
V = Lake Veere; G = Lake Grevelingen; O = Eastern Scheldt; B = Lake Brielle

freshwater, brackish and estuarine areas and will form the basis for water resources management after 1985.

2.2. *The relation between nutrients and the growth of algae*

The information of relevance for the quantitative aspects of water resources management aiming at the optimization of water quality includes routine data on nutrients (fig. 2). The annual pattern of nutrient contents and the relation of these to the annual chlorophyl pattern indicate differences between the various basins. The level of dissolved silicon is mostly minimal in spring in the basins, whether fresh or saltwater. This is due to diatoms, which have a relatively high growth rate in the spring despite the low water temperatures. Since this species is not known to be harmful or troublesome, it is not necessary to take any measures to limit the silicon load in these basins.

The situation with regard to phosphorus in the freshwater Lake Brielle is more or less the normal one, with minimum concentration of dissolved – i.e. directly assimilable – orthophosphate during the summer period.

This may make it possible to normalize the excessive algae bloom by taking measures to restrict the phosphorus load of the lake. Dissolved nitrogen is present in larger quantities and may possibly have a limiting influence on the lake's maximum algae biomass when there is an excessive algae bloom.

In the saltwater basins – particularly those which have been dammed off – it is very probably not the available phosphate but the available nitrogen which is the limiting factor for the development of algae. This can be seen from the minimum concentrations of this nutrient in summer. It is not known through what processes nitrogen becomes the apparent limiting nutrient for algal growth in the salt lakes so this mechanism is therefore an object of current study. For the quality aspects of water resources management it should be taken into account that dephosphatizing the discharges onto the salt basins may well have less effect than quantitative measures to reduce the nitrogen load. It is also conceivable that, when it finally has to be decided whether the Grevelingen basin is to be a salt or a freshwater lake, the limiting influence of the nitrogen content will be an important argument in favour of a salt lake. Another factor as yet unknown is at what chloride-ion concentration the nitrogen-reducing processes become predominant. Thus arises an argument for a provisional policy of maintaining the greatest possible salinity in the lakes and minimizing the entry of nitrogen in order to reduce the growth of algae when they appear to be a nuisance. A high degree of salinity also appears to be a precondition for the presence of organisms of the greatest possible number of species (speciesdiversity).

In order to have a solid basis for both the quantitative and qualitative aspects of water resources management, it is also necessary to have precise knowledge of the nature and distribution of the sources of these substances for the different basins (fig. 3). In view of the above considerations the relative distributions of the phosphorus sources are given for the freshwater lakes and those of the nitrogen sources for the saltwater lakes.

What immediately catches the attention here is the situation in the Hollandsch Diep and the Haringvliet, where the riverflow is by far the most significant factor in determining the distribution of the phosphate sources. Not much can be expected here from measures to dephosphatize the tributaries discharging onto this basin. It is also clear that the load originating from the polders will have a relatively greater influence in Lake Veere than in Lake Grevelingen. Here too a key to the management of water quantities can be destilled if it is understood to include the diversion of polderwater discharges. This tool will be especially appropriate if it can be combined with land rearrangement projects.

After this outline of the present situation, the relation between quality and quantity will be discussed in the light of recommendations related to alternative engineering designs in the policy-analysis phase of the Delta project, to management problems with the new basins and to guidance during the construction stage of the project. An illustration will be given for each of these aspects.

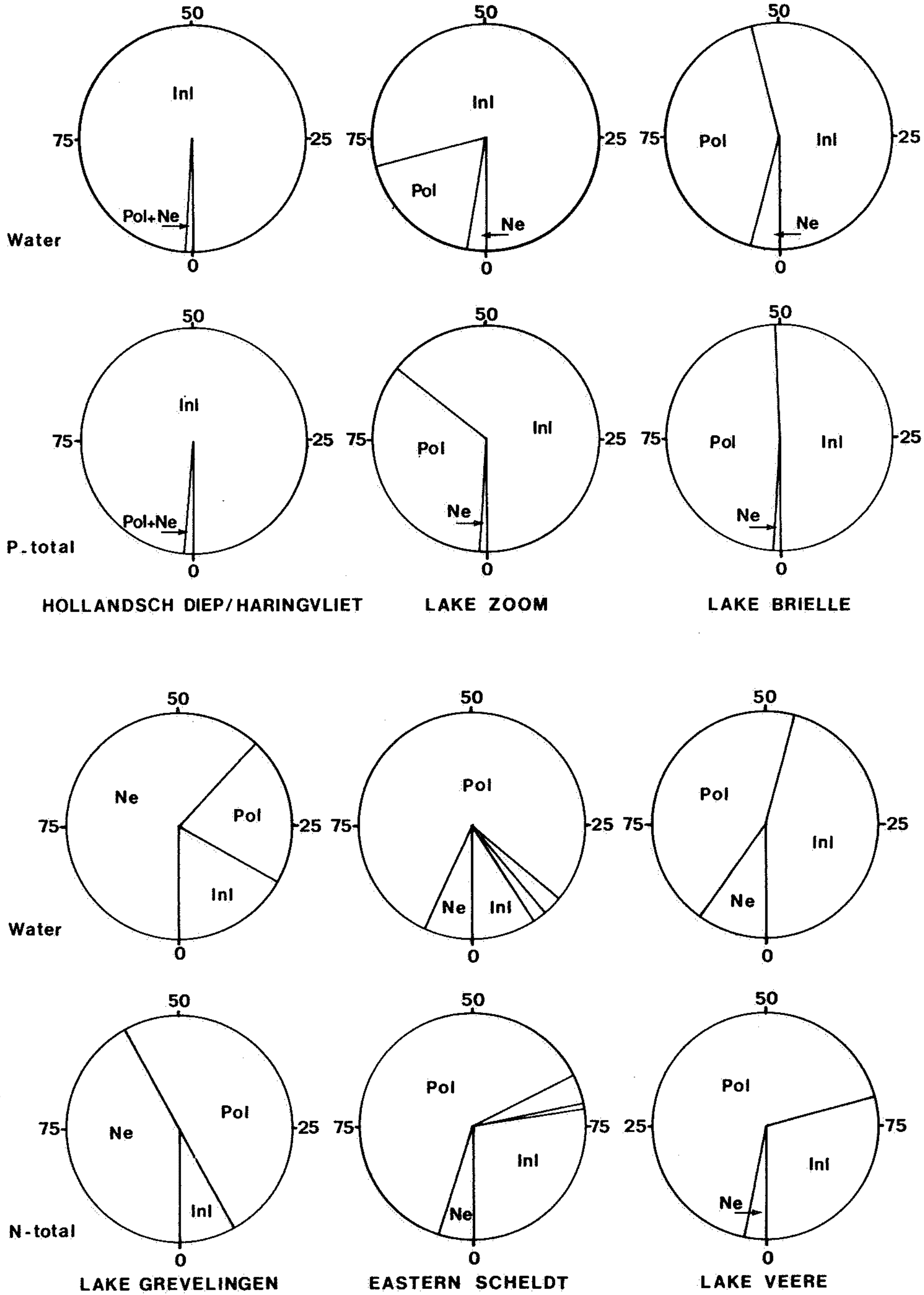


Fig. 3 The load-distributions of water and relevant nutrients on some Delta-waterbodies.
Ne = precipitation; Pol = polderdischarge; Inl = inlet

3. RECOMMENDATIONS FOR CONSTRUCTION DESIGN

A wide variety of questions have arisen during the designing phase of the Delta project, some of which are of significance for the relation between quality and quantity.

The alignments of the secondary dams not only determine the volume and surface area of the new basins, they also determine the extend of physical transport as a result of mixing and flow. The choice of alignment for the Oyster Dam is especially interesting.

In one of the alternative alignments – for reasons connected with flow patterns during its construction phase – one has to seperate a part from the new Lake Zoom. The area concerned is the currently submerged area of the Marquisate near Bergen op Zoom (fig. 1).

This will result in a form of subpartitioning in the future Lake Zoom. In view of the positive developments in Lake Marker – which has also been embanked – an embanked Lake Marquisate offers good prospects as far as the water quality is concerned, partly since water control can be kept completely independent from that of Lake Zoom.

In Lake Zoom itself some means for discharging water is needed even if only from the point of view of regulating the water level. Gravitational discharge (thus avoiding energy costs) of surplus fresh water is in principle possible in three directions: towards Lake Grevelingen, towards the Eastern Scheldt and towards the Western Scheldt. The fact, that the Western Scheldt has an estuarine character with relatively large variations in its salinity, forms one of the reasons why it has been decided to design structures to discharge the surplus water in that direction.

In considering what discharge capacity should be designed, consequences were examined for water quality as well as for water level regulation, together with financial aspects. Much importance has been attached to potential ways of maintaining the lowest possible chloride concentration in Lake Zoom. With respect to the Eastern Scheldt the so called "recovery of fresh water" is of central importance, i.e. the recovery of fresh water by means of the Philips sluices, which are equipped with a system for separating fresh water from salt water. The recovery of this water minimizes the fresh water load in the Eastern Scheldt which aims to maintain the future salinity distribution of the Eastern Scheldt similar to what it is at present. However, the recovered fresh to brackish water will cause extra salt to enter Lake Zoom which is clearly contrary to the policy of minimizing the chloride-ion concentration in this lake. A possible solution here would seem to be to flush Lake Zoom towards the Western Scheldt. Model-calculations have shown that a reduction in the chloride-ion content could be achieved in this way.

Unfortunately forced flushing of the lake will at the same time mean an increased inflow of polluted Rhine water, increasing the load of toxic materials as a result of increased sedimentation of the silt brought in by the water. As in case of the Haringvliet forced flushing could become the dominant factor determining the phosphate balance. The enlarged influx of water into Lake Zoom might lead to qualitative changes not only in Lake Zoom itself but also in the biotic communities in and around the Western

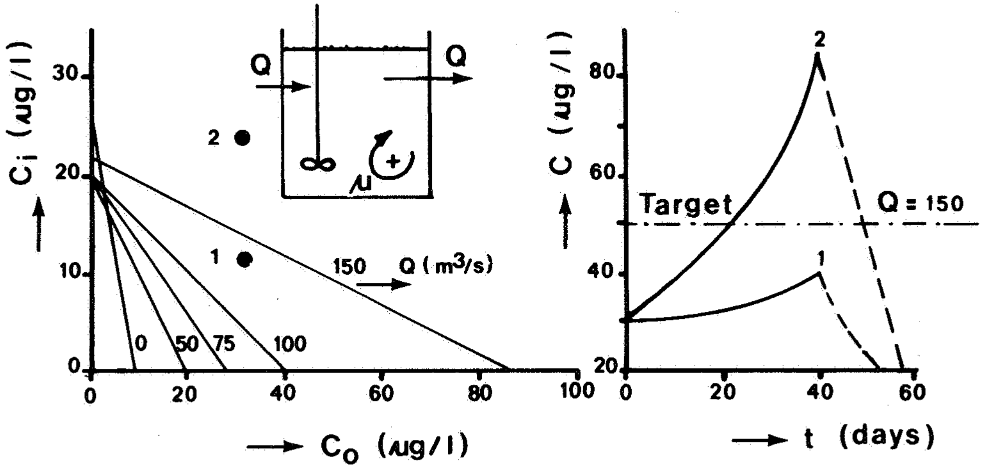


Fig. 4 The savety-area's for algae bloom in the future Lake Zoom at various flushing rates.
C_i = inlet chlorophyll concentration, C_o = in-lake chlorophyll concentration before blooming period.

Scheldt as a result of the alterations in the salinity distribution there.

Another effect which could be of importance, besides the introduction of polluting substances, is the flushing of algae growing in Lake Zoom, as probably also happens in the Haringvliet. The result of a simple model designed for this topic is shown in figure 4.

For a number of Dutch lakes an estimate has been made of the net growth rate of algae, based on the observed increase in chlorophyll during the exponential growth phase. There are of course all sorts of possible influences on these growth rate coefficients, such as consumption by zooplankton, the underwater light climate and the nutrient situation. The objective here was to estimate the growth rate coefficient observed in practice and to compare it with the outflow rate resulting from a continuous extra inflow of 50, 75, 100 or 150 m^3/sec. The observed growth coefficient varied between 6 and 10% per day. In practice the exponential growth phase did not last longer than 3 weeks, but for safety's sake a growth period of 40 days has been taken as a basis for evaluating the models' results. If the chlorophyll content is still below 50 μg/l after this period, it can be assumed that there has been no excessive algae bloom. The model showed to be particularly sensitive to the magnitude of growth rate coefficients: with a growth rate of 10% per day none of the alternative capacities were found to be sufficient to prevent algae growth. In the final recommendations it was therefore stated that no assurance could be given about the extent of a reduction in algal growth through a forced inflow of water, even if this involves very large quantities.

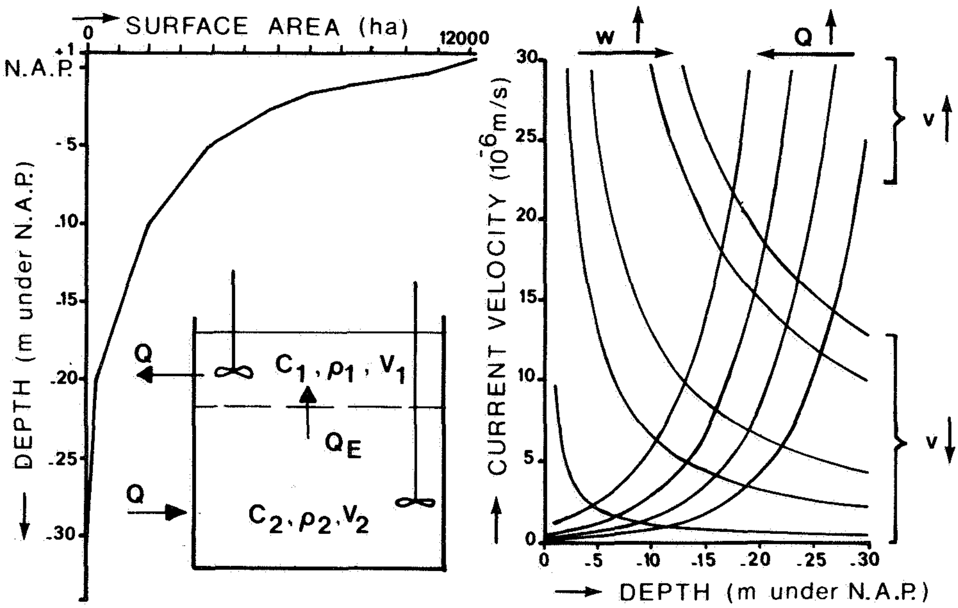


Fig. 5 The hypsographic curve of Lake Grevelingen and the characteristic displacementvelocities of the salinocline.
v ↑ = upward velocity at different flowconditions (Q); v ↓ = downward velocity at different windvelocities (w); w ↑ = increasing windvelocity; Q ↑ = increasing flow; NAP = Dutch reference level.

4. RECOMMENDATIONS FOR WATER RESOURCES MANAGEMENT

The management problems relating to the new basins are also very diverse in kind, such as water level regulation in relation to the development of the biotic communities along the shores, land consolidation and its consequences for polderwater discharges, the disposal of dredged materials and the provision of facilities for the basins' new uses.

Lake Grevelingen in particular offers good prospects as an interesting area from the point of view of natural development, provided that it proves possible to keep it sufficiently saline. This lake is also suitable for recreational purposes, besides having some potential for commercial fishing, especially eel fishing. The new destinative lay-out sketch for Lake Grevelingen gives primary importance to its roles as a natural and recreational area. As noted above, with respect to the former it is important that efforts are undertaken to ensure that it is saline enough to provide a suitable environment for many species. A link between the lake and the North Sea is therefore desirable.

The inflow of sea-water has a series of consequences for the relation between water quality and water quantity; these are consequences which are peculiar to saltwater systems. The density of water is a function not only of temperature, but also of salinity (higher salinity means a higher density). In view of the fact that a difference

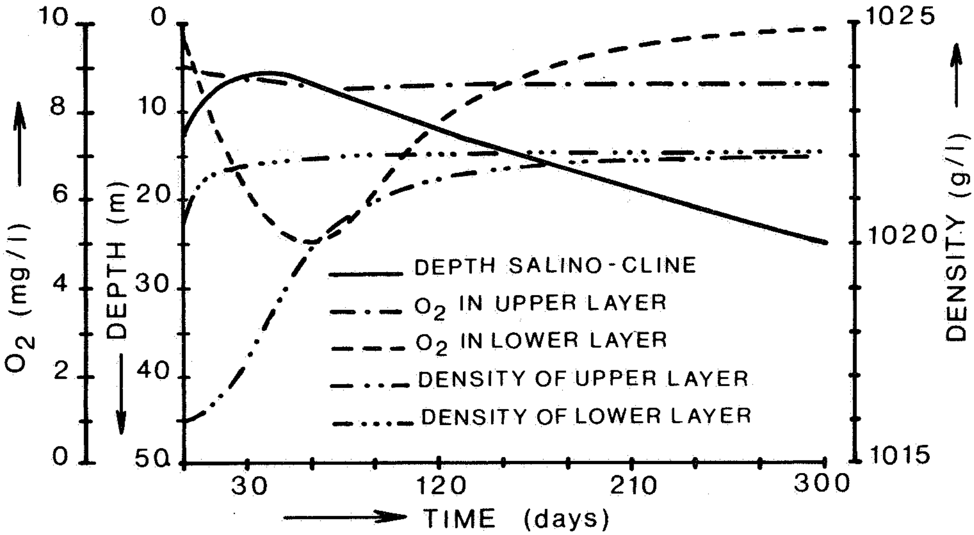


Fig. 6 The densities and oxygen concentrations of upper and lower waterlayer as function of time (simulated).
NAP = Dutch reference level.

of roughly 1 g/l in the chloride-ion concentration causes the about same difference in density as a temperature difference of 10 °C, considering the fact, that a vertical thermal stratification of 10 °C can be regarded as reasonably stable, the influx of sea-water will bring about pronounced vertical stratification since the present difference in salinity averages 4 g chloride per litre.

A very important consequence of stratification is that water from the upper and lower layers no longer intermixe. This means that the possibilities for the mutual exchange of dissolved substances are limited to levels of molecular diffusion or somewhat higher. This has significant effects on the oxygen balance. The upper layer remains in contact with the atmosphere, so that the oxygen can be replenished through reaeration if necessary, while the greater difference in density means that the lower layer is cut off from this source of oxygen. Stratification does not restrict the consumption of oxygen in the lower layer: the mineralization of organic matter in the water phase or at the bottomwater interface causes continuous oxygen depletion of the lower layer, so its concentration will consequently diminish.

This being the case it is important to know how and when sea-water can be allowed into Lake Grevelingen without too great a risk.

Two physical processes are of relevance here, one tending to raise the hypolimnetic boundary layer ("salino-"cline) and one tending to lower it (fig. 5). The first process results from the flux with which the sea-water enters and is determined to a large extent by the morphology of the basin. The second is determined above all by the wind velocity, the source of turbulent energy, as well as by the morphology and the difference in density between the upper and the lower layer.

Studies with a stratification model have shown that if water is permitted to enter it should preferably be done in maximal quantities, since then a reduction in the time during which a salino-cline is present can be achieved and the influx of sea-water (rich in oxygen) will restrict the fall of the oxygen concentration in the lower layer (fig. 6). Inflow should take place at low temperatures: there are three arguments in favour of this, of which the first two formed part of the model. Firstly, in winter the oxygen concentration of the inflowing sea-water is at its maximum as a result of the temperature-dependent solubility of oxygen in water in equilibrium with the atmosphere. Secondly the rate of oxygen consumption is at a minimum in winter due to the retardation of the bacterial decomposition process at low temperatures.

Finally, at low temperatures aquatic organisms are expected to show a greater tolerance to reduced oxygen concentrations because of their slower metabolism. The regulation of water quantities, which will be carried out by means of the Brouwers Dam sluices, completed in 1978, will therefore be limited to the winter period for the time being. The aim will be to restore a maximum exchange of water between the North Sea and Lake Grevelingen, so that ultimately there will no longer be any question of stratification.

The sluice will serve to let water both in and out, so that the exchange will be accompanied by a limited variation in the water level, i.e. about 6 cm per tide. These level variations are within the variations caused by wind, so the procedure of inflow and outflow will not have any harmful effects on the shores.

5. GUIDANCE OF CONSTRUCTION ACTIVITIES

As with the policy analysis for construction alternatives and with the management of the new lakes, there are also many questions to be solved with respect to the guidance during the construction phase. One complex problem here is that of coordinating the construction of the primary dam with that of the partitioning dams (fig. 1). The aim is that they should be constructed concurrently, for reasons connected with the tidal storage capacity.

If the dams are to be completed all at the same time, this will leave various technical possibilities for closing the partitioning dams: these include closure with blocks, with caissons, and with sand. For the latter method it would be necessary to use the floodgates in the storm-surge barrier to reduce the flow velocity in the gaps to be closed. Closure of the secondary dams with sand would cost about 50 million guilders less than closure with blocks or caissons. Closure with blocks can be carried out at normal tide, and would thus have almost no adverse effects on the environment of the Eastern Scheldt (fig. 7).

The original plan for closing the dams with sand envisaged reducing the volume of the tide by gradually closing more of the floodgates in the storm-surge barrier until in the final phase there would have been virtually no tide at all in the Eastern Scheldt for a

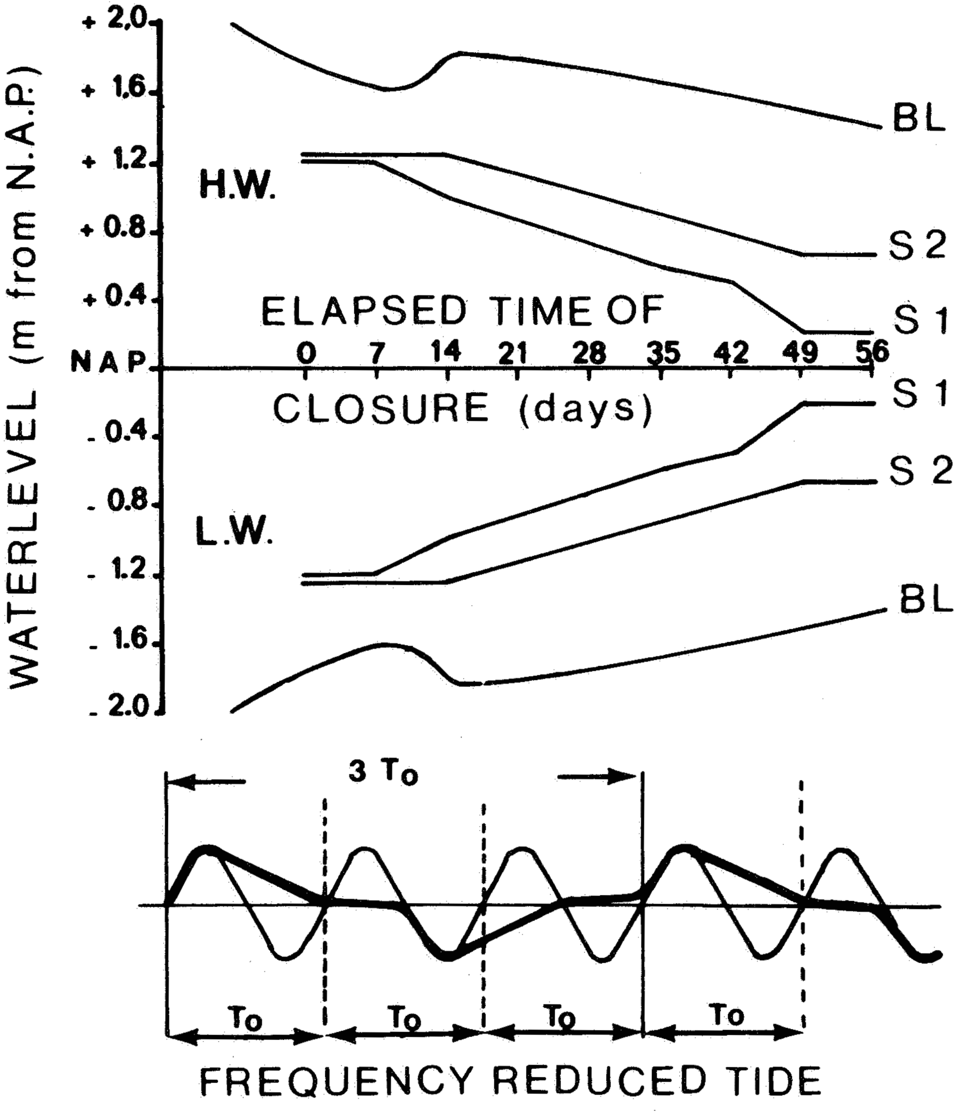


Fig. 7 The expected tidal amplitude time functions of alternative closing methods of the secondary dams, together with waterlevelvariation during "stretched" tide.
B_1 = Blocksclosure; S_1 = original sand closure; S_2 = Sand closure with frequency-reduced tide.

period of 5 days. Provided that the secondary dams were closed in the winterperiod the effects on the aquatic environment and the biotic communities along the shoreline would be limited and a swift recovery could be expected. In view of the fact that it would only happen once this was not considered to be unacceptable.

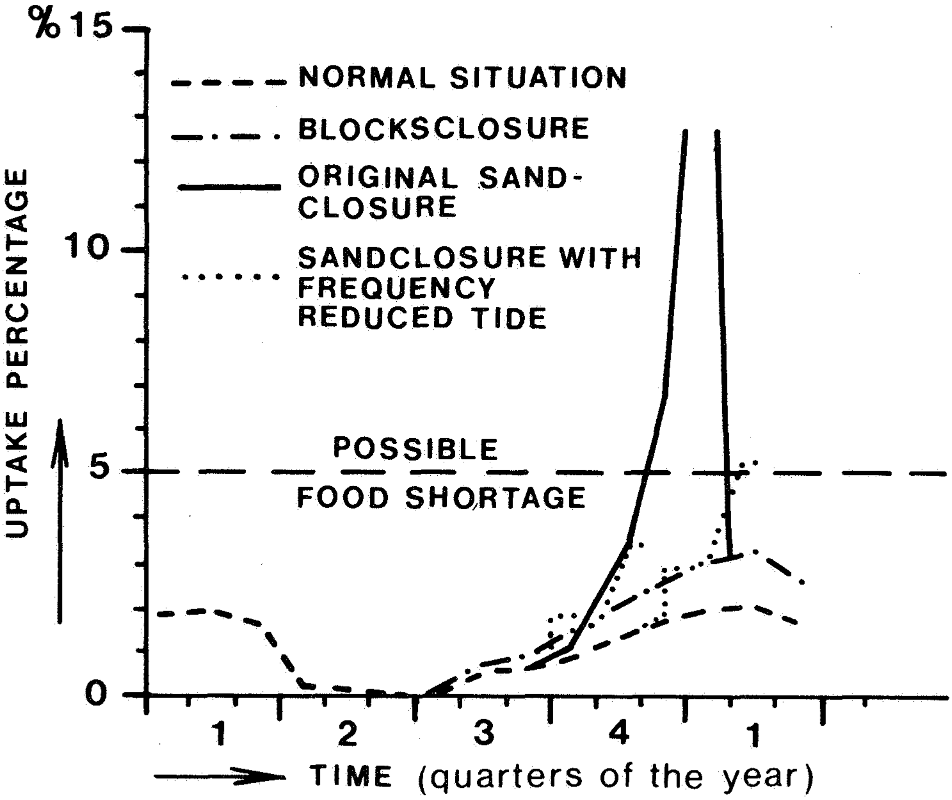


Fig. 8 The expected food uptake of birds during closure alternatives (data from Baptist, unpublished).

Nevertheless, the advisability of this method was questioned both in connection with the food situation of the birds (fig. 8) and with the avoidance of stratification. The absence of all tidal flow could lead to stratification due to the continued discharge of fresh water, which in turn could cause the chloride-ion concentration in the uppermost water layer to drop too far. And it is precisely this layer that is in contact with the rich biotic communities of the shoreline area. Other effects of the discontinuation of inter-mixing by the tide might be increased sedimentation and oxygen problems.

The possibility of closing the gaps with sand by "protracting" the tides is now being explored: here the floodgates in the Eastern Scheldt Dam would be used to artificially reduce not the volume of the tide but rather its frequency (fig. 7). This would mean the volume remaining more or less unchanged, while the flow velocity would be diminished but not reduced to nil. Similarly it is not thought that this method would produce in-surmountable difficulties for the birds (fig. 8). Finally, I should like to point at two relevant details: the first is that the periods of low water should occur as far as possible during the daytime, as it is probably easier for birds to feed then than at night; secondly, the phasing of the closure of the Oyster Dam must be coordinated with that of the Philips Dam in such a way that there is no net southward flow in the Eendracht Canal, as then the relatively fresh water from the Volkerak would flow round Tholen into the main basin of the Eastern Scheldt. As a result of the reduction in the exchange of water between this area and the North Sea, it would be some months before the salinity could again reach the right level.

6. CONCLUSION

Knowledge of the relation between water quality and water quantity is of importance for succesful water resources management. This is particularly appropriate for the delta area of Southwest Netherlands, since the Delta Project changes the possibilities for quantitative water management. It should be noticed that water control measures devised to influence the water quality in one basin may also affect the water quality in an adjacent basin. Furthermore, the possibility of controlling water levels and their fluctuation makes it possible to influence developments on the shoreline: here it would be more correct to speak of a relation between water quantities and the quality of the environment outside the dikes, of which the relation between water quantity and water quality forms a part. Finally the development and use of often quite simple models has a stimulating effect on the development of ideas when alternative actions are to be evaluated and that such models can sometimes indicate the areas requiring further research.

CONCLUSIONS WITH REGARD TO THE CONNECTION OF WATER QUANTITY AND WATER QUALITY STUDIES OF SURFACE WATERS

P.E. RIJTEMA

SUMMARY

The cited literature has made it clear that regarding many aspects there exists a close relationship between surface water quality and quantity.

Changes in water quality and water regime in most cases result from human activities. This implies that multidisciplinary studies of surface waters should be encouraged to ensure that quick progress is made in controlling that part of our environment. In the future it probably will be required to give an assessment of the impact of water management measures on environmental quality, making such studies of even greater importance.

A good insight in the processes that regulate water quality is of prime importance. An operational means to that end is constructing physico-mathematical models which describe the combination of water and substance budgets. Such models stimulate the formulating of new ideas as well as the actual problems encountered and make it possible to weigh in first instance the proposed alternative measures with regard to both pollution control and water management.

CONCLUSIONS

Consideration of the results and discussions given by Rijtema (1979), Lijklema (1979), Steenvoorden and Oosterom (1979), Van Straten (1979), Van Wirdum (1979) and Bannink (1979) of various aspects of the relation between water quality and water quantity in studies of surface waters, leads to a number of conclusions.

It is necessary to have a good knowledge of the processes which determine water quality. For some inert substances as for instance Cl^- a fair impression can be obtained from simple balance studies. For other substances where, apart from the water movement, also other processes play an important role, the studies will be very complex. By combination of water and substance balances with a correct physico-mathematical formulation of the processes concerned, a model approach can be obtained to study the mutual relationship between water quality and water quantity. For the testing of such models in practice it is necessary to define explicitely the aims of the water quality research, to be able to focus place, time and frequency of the required sampling on the goal of the research.

It has been shown that the setting up and operation of still fairly simple models stimulates the forming of views and the formulation of the problems to be dealt with. The models given by the mentioned authors can be used as guidelines for programming research in this field of study, as their models give a clear view of the present gaps in knowledge. They furthermore can be used for a first weighing of alternative measures and to forecast which measures are required to attain the quality standards necessary for the different functions of surface waters. In this connection the following items can for instance be mentioned:

- variation in the required quality standards for effluents of sewage treatment plants, depending on the strong fluctuation of the natural discharge of catchment areas in winter and summer;
- the phosphate load of surface waters originating from natural sources, as well as the load by the flushing water used, when discussing the need and urgency of dephosphatizing the effluents;
- the fact that certain measures in upstream regions may be essential for the quality of downstream surface waters.

Changes in chemical composition of surface waters, as well as changes in discharge pattern are generally caused by human activities. Much attention should be given to the development of system analyses to quantify the consequences of such activities in terms of water quality and water quantity.

In spite of the restrictions of the existing models it must be concluded that by integrating the present knowledge in the field of water quality and quantity research, important achievements can already be made. A close co-operation between the various disciplines is, however, a requirement for rapid progress in this field.

In view of the expected legislation with regard to environmental impact statements, a close co-operation also is urgently needed. It then will be necessary to forecast by means of environmental impact calculations the consequences of human activities, also in the field of water management. In this respect the relation between qualitative and quantitative aspects requires much further research.

The chemical composition of the surface waters in an important part of The Netherlands is mainly determined by the quality of the water discharges of the main rivers. Apart from this, regional variations in natural load exist due to differences in soil composition and in existence of seepage from deep layers. This implies that for certain functions uniform water quality standards cannot be defined; they must be adapted to the local or regional situation.

Finally it is to be concluded that from a functional point of view qualitative and quantitative water management should be closely connected. The results of research on water quality and quantity, as well as the analysis of the sources of pollutants, give the water manager not only a starting point for the weighing of measures to directly eliminate influences of the main sources, but it also yields a basis for quantitative management, as for instance by means of flushing policy, weir control, etc., to indirectly reduce the disadvantageous effects of these sources.

REFERENCES

BANNINK, B.A. (1979) – The relation between water quality and water quantity in the changing delta area of Southwest Netherlands. Comm. Hydrol. Onderzoek TNO, Den Haag. Versl. en Meded. 25.

LIJKLEMA, L. (1979) – Theoretical aspects of the relation between water quantity and water quality of surface waters. Comm. Hydrol. Onderzoek TNO, Den Haag. Versl. en Meded. 25.

RIJTEMA, P.E. (1979) – General survey on the relation between water quantity and water quality. Comm. Hydrol. Onderzoek TNO, Den Haag. Versl. en Meded. 25.

STEENVOORDEN, J.H.A.M. and H.P. OOSTEROM (1979) – Natural and artificial sources of nitrogen and phosphate pollution of surface waters in The Netherlands. Comm. Hydrol. Onderzoek TNO, Den Haag. Versl. en Meded. 25.

STRATEN, G. VAN (1979) – Hydrological aspects of oxygen behaviour and algal blooms in river systems. Comm. Hydrol. Onderzoek TNO, Den Haag. Versl. en Meded. 25.

WIRDUM, G. VAN (1979) – Dynamic aspects of trophic gradients in a mire complex. Comm. Hydrol. Onderzoek TNO, Den Haag. Versl. en Meded. 25.

COMMITTEE FOR HYDROLOGICAL RESEARCH T.N.O.

Proceedings and Informations

No. 1. Proceedings of Technical Meetings 1–6 (with summaries in English), 1952.

1. Investigations into the Water Balance of the Rottegatspolder
2. The Water Supply for Crops I
3. Observations of Groundwater Levels
4. Investigations by Drain Gauges in the Netherlands
5. The Water Supply for Crops II
6. The Problem of the increasing Salinity of Ground and Surface Water in the Netherlands

No. 2. Proceedings of Technical Meetings 7–10 and Report on the evaporation research in the Rottegatspolder 1947-1952 (with summaries in English), 1955.

7. The Study of Precipitation Data
8. Model Research on Groundwater Flows
9. Measurements and improvement Works in Basin of Brooks
10. Geo-electrical Research

No. 3. Proceedings of Technical Meetings 11–12 (with summaries in English) and Report on the Lysimeters in the Netherlands (in English), 1958.

11. The Water Supply of Sandy Soils
12. Quality Requirements for Surface Waters

No. 4. Evaporation Symposium and Report on the Lysimeters in the Netherlands II (with summaries in English), 1959.

No. 5. Proceedings of Technical Meetings 13–14 (with summaries in English), 1960.

13. Groundwater Levels and Groundwater Movement in the Sandy Areas of the Netherlands
14. Water in unsaturated Soil

No. 6. Proceeding of Technical Meeting 15 (with summaries in English and French, 1961.

The Regimen of the Rhine, the Ysselmeer and Zeeland Lake

No. 7. Proceeding of Technical Meeting 16 (with summaries in English), 1962.
The dry Year 1959

No. 8. Proceeding of Technical Meeting 17 (with summaries in English), 1963.
The Laws of Groundwater flow and their Application in Practice

No. 9. Proceeding of Technical Meeting 18 (with summaries in English), 1963.
Water Nuisance

No. 10. Steady Flow of Ground Water towards Wells, compiled by the Hydrologisch Colloquium (in English), 1964.

No. 11. Proceeding of Technical Meeting 19 (with summaries in French and German), 1964.
Geo-hydrological Cartography

No. 12. Proceeding of Technical Meeting 20 (in English), 1966.
Water Balance Studies

No. 13. Proceeding of Technical Meeting 21 (in English), 1966.
Recent Trends in Hydrograph Synthesis

No. 14. Proceeding of Technical Meeting 22, 1968.
Precipitation Data (II) and
Report on the Lysimeters in the Netherlands (III) (both with summaries in English)

No. 15. Proceedings and Informations no. 15 (in English), 1969.
Soil – Water – Plant

– Proceeding of Technical Meeting 23, 1968.
See Page
(Will not be published).

– Proceeding of Technical Meeting 24, 1970.
Flow in the unsaturated Zone
(Will not be published).

No. 16. Proceeding of Technical Meeting 29 (with summaries in English), 1975.
Hydrological Investigations for Masterplan for the future Watersupply in the Netherlands

No. 17. Proceeding of Technical Meeting 25 (in English), 1973.
Automatic Processing of Hydrological Data

No. 18. Proceeding of Technical Meeting 26 (in English), 1974.
Hydraulic Research for Water Management

No. 19. Proceeding of Technical Meeting 27 (in English), 1974.
The Hydrological Investigation Programme in Salland (The Netherlands)

— Proceeding of Technical Meeting 28, 1973.
Water Quality of Dutch Rivers with respect to Water Management
(Will not be published).

No. 20. Proceeding of Technical Meeting 30 (in English), 1976.
Salt Distribution in Estuaries

No. 21. Proceeding of Technical Meeting 31 (in English), 1976.
Groundwater Pollution

No. 22. Proceeding of Technical Meeting 32 (with summaries in English), 1976.
Systems Approach to the Management of Water Resources

No. 23. Proceeding of Technical Meeting 33 (in English).
Precipitation and Measurements of Precipitation
(Is not yet published).

No. 24 Proceeding of Technical Meeting 34 (in English), 1978.
Urbanization and Water Management

No. 25 Proceeding of Technical Meeting 35 (in English), 1979.
The Relation between Water Quantity and Water Quality in Studies of Surface Waters

No. 24 Proceeding of Technical Meeting 34 (in English), 1978.
Urbanization and Water Management

No. 25 Proceeding of Technical Meeting 35 (in English), 1979.
The Relation between Water Quantity and Water Quality in Studies of Surface Waters