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Hydrobiology and chemistry of the Schelde and Westerschelde

H.Hummel, C.Bakker, A.H.L.Huiskes and A.G.A.Merks

Editors

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HYDROBIOLOGY AND CHEMISTRY OF THE SCHELDE AND WESTERSCHELDE

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1988

INTRODUCTION TO THE SCHELDE SYMPOSIUM

H. HUMMEL

The Schelde river and the Westerschelde estuary are of vital importance to Belgium and the south-western part of The Netherlands with respect to shipping, industrial activities and recreation. The area has great natural potentials and could be of importance to fishing and aquaculture activities. However, due to intensive anthropogenic activities the potentials of the river and estuary are minimized.

To evaluate the present state of our knowledge on the ecology and chemistry of this complex and imperfectly understood environment the Delta Institute for Hydrobiological Research organized the Schelde Symposium on May, 26 and 27, 1987, in Terneuzen, The Netherlands. The purpose of the symposium was to promote future and cooperative research and cooperation.

The symposium proceedings comprise the results of this specific conception and therefore include, beside the keynote lectures, list of participants and opening address, also the summaries and conclusions of discussions and suggestions for future research and cooperation.

The symposium was attended by 93 Belgian and Dutch scientists. The opening address was spoken by Drs. E.H.T.M. Nijpels, Minister of Housing, Physical Planning and Environment. On the first day 6 keynote lectures were presented on aquatic, and semiterrestrial ecology and chemistry, chaired by Dr. Ir. W.G. Beeftink. The second day lectures and queries were discussed in parallel workshops on ecology, chemistry and modelling, presided by Drs. E.K. Duursma, P.H. Nienhuis, J.W. Rijstenbil and W. Zurburg. Finally a general session on the outcome of all lectures and discussion was chaired by Dr. R.W.P.M. Laane. The conclusions from this general session are given in the contribution of Laane. Technical and management assistance were lend by Mrss. E.S. Nieuwenhuize and M.J. van Leerdam.

The more individual queries and remarks are summarized in Table 1a of this introduction. The queries and remarks were numbered and subdivided in those on (aquatic and semiterrestrial) ecology and those on chemistry. Table 1b presents a preliminary suggestion for cooperation between institutes involved in the fields of research mentioned in the numbered queries of Table 1a.

To complete the information the list of participants (Table 2) has been supplemented as far as stated at the symposium with a person's research-discipline and specific research-topics.

Beside these symposium proceedings the keynote lectures were issued in the Hydrobiological Bulletin. Both, the proceedings and the Hydrobiological Bulletin issue mark a succesfull symposium and the importance of future research in the Schelde river and Westerschelde estuary.

Table 1a. Queries and remarks posed at the Schelde symposium.

A. Aquatic and semiterrestrial ecology.

1. Is it possible to partition the species-composition and production of phytoplankton in the Schelde and Westerschelde into 3 zones, as assumed in the model of Hummel ? It should be quantified with C^{14}-measurements and microscopic counts. During the workshops it was suggested that the primary production between Gent and Antwerpen, in the freshwater tidal zone, still could reach values of 1.5 $g.m^{-2}.day^{-1}$.
2. The interrelation between detritus and zooplankton, which according to the model of Hummel seemed to be important in the brackishwater tidal zone, is not clear. More data are needed.
3. The high biomass of the macrozoobenthos near Zandvliet, as observed by Van Impe (1985), is questioned.
4. There is a lack of data on the role of the tidal flats in the carbon flow of the Westerschelde.
5. The effects of bioturbation, morphological (anthropogenic) changes and dredging in the Westerschelde on the organic carbon and nutrient-balance are not known.
6. Are salt-marshes a source or sink for organic matter, nutrients or pollutants. According to the model of Huiskes they are a source. This should be clarified with budget-studies.
7. To study effects of pollutants some representative groups or organisms should be chosen, *e.g. Nereis diversicolor, Macoma balthica, Fucus vesiculosus.*
8. Genetic changes in the biota of the Westerschelde caused by pollutants are not known. Sedentary organisms from salt-marshes are most suitable to study this topic.
9. The following general models are employable in the Westerschelde : a) a comprehensive 'ecomodel', composed by hydrodynamic-, transport-, carbon-flow- and oxygen models, and b) specific models, such as a nutrient (eutrophication) model or a dispersion model (for *e.g.* toxicants).
   Three current models are available : the Eems-Dollard, the Waddenzee and the Oosterschelde model.

B. Chemistry.

10. Mixing of marine and fresh water sediments will lead to nonconservative behaviour in ad- or desorption of pollutants to particulates. Conservative behaviour is much time-dependent.
11. Standardization of leaching techniques seems impossible but necessary. To what extent can laboratory experiments be translated into field situations. What does it tell about effects on organisms.
12. What are the main pathways for pollutants in the Westerschelde : via water (dissolved *vs.* particulate) or via sediments.
13. There is a lack of data on the speciation of heavy metals in the Westerschelde. Strong spatial and temporal changes in environmental conditions (pH, O_2, salinity) make the Schelde interesting to study chemical speciation. Only a small portion of trace metals is available to organisms.
14. Results on Kd's of PCB congeners are likely to be influenced by the techniques applied to separate the water-phase and particulate matter.
15. A lack of knowledge exists on the net output of material from the Schelde into the North Sea.
16. The content of organic matter in the bottom sediments is so high that an anaerobic-aerobic transition will not influence the desorption processes of heavy metals to the water.

Table 1b. Suggestions for cooperation between institutes on the numbered topics from Table 1a.

Topic (Table 1a)	Institutes
1	DIHO, KFA, TNO, ULB, VUB
2	DIHO, VUB
3	DIHO, RUG
4	DIHO
5	DGW, DIHO, EON, RIN, RUU, TNO
6	DGW, DIHO, KUL, RIN, RUG, TNO, UIA
7	DGW, DIHO, RUU, TNO
8	DIHO, VUA
9	BMM, DGW, DIHO, EON, UdL, ULB, VROM, WL
10	TNO
11	DGW, DIHO, ECN, RUU, TNO, VUB
12	DGW, DIHO, RUU, TNO
13	DGW, DIHO, TNO
14	DGW, DIHO
15	DGW, DIHO, RUU, TNO, ULB, VUB
16	TNO

BMM = Beheerseenheid Mathematisch Model, Brussel
DGW = Dienst Getijdewateren, RWS, Middelburg en Den Haag
DIHO = Delta Instituut voor Hydrobiologisch Onderzoek, Yerseke
ECN = Energie Centrum Nederland, Petten
EON = Ecologisch Onderzoek Noordzee, Texel
KFA = Kern-Forschungs-Anlage, Jülich
KUL = Katholieke Universiteit Leuven
RIN = Rijksinstituut voor Natuurbeheer, Texel
RUU = Rijksuniversiteit Utrecht
RUG = Rijksuniversiteit Gent
TNO = Toegepast Natuurwetenschappelijk Onderzoek, Den Helder
UdL = Université de Liège
ULB = Université Libre Bruxelles
UIA = Universitaire Instelling Antwerpen
VROM = Ministerie van Volkshuisvesting, Ruimtelijke Ordening en Milieu, Den Haag
VUA = Vrije Universiteit Amsterdam
VUB = Vrije Universiteit Brussel
WL = Waterloopkundig Laboratorium, Delft

Table 2. Schelde Symposium list of participants, their research discipline and specific research-topics.

Name	Address	Discipline	Research-topic
R. Baert	Aalst	–	
W. Baeyens	Brussel	GC	Sediment-water fluxes of pollutants
W.G. Beeftink	Yerseke	B, MB	Salt-marsh ecology
J.W. Beijersbergen	Middelburg	–	

Name	Address	Discipline	Research-topic
G. Billen	Brussel	Mic, MB	
P. Boere	Yerseke	–	
M. Bogaert	Brussel	MB	Carbon and chlorophyll in the North Sea
A.J. Bruin	Hoek	–	
J.W. Busser	Goes	–	
L. Bijlsma	Middelburg	–	
F. Colijn	Den Haag	MB	Management of aquatic systems
J. Coosen	Middelburg	MB	Macrobenthos, coastal ecology
J.A.M. Craeymeersch	Yerseke	MB	Macrobenthos dynamics
E. Coppejans	Gent	B	Macro-algae, benthic production
N. Dankers	Texel	MB	The impact of mussels in the ecosystem
M.H. Daro	Brussel	–	
J. de Backer	Antwerpen	–	Silt-sampling
K. de Brabander	Brussel	–	
D.J. de Jong	Middelburg	P	Microphytobenthos
F. Dehairs	Brussel	GC	Trace elements, suspension, sediments
M. de Reu	Gent	–	
K. Delbeke	Brussel	T	Organochlorines
P.J. den Besten	Utrecht	–	
J.E. Haarlem	Terneuzen	–	
M. Haks	Hoek	–	
O. Hamerlynck	Gent	MB	Small-fish community, Gobiidae
C.H.R. Heip	Yerseke	MB	Benthos
M.A. Hemminga	Yerseke	MB	Salt marsh ecology
P. Herman	Gent	Mod	
R. Herman	Gent	MB	Meiobenthos
A.M.B. Holland	Middelburg	C	Waterquality
A.H.L. Huiskes	Yerseke	B, MB	Salt marsh ecology
H. Hummel	Yerseke	MB	Ecophysiology of macrofauna
C.W. Iedema	Middelburg	E	Management of the Westerschelde
E. Jaspers	Bredene	–	
P. Kelderman	Delft	GC	Sediment-water fluxes, sedimentation
M.A.T. Kerkhoff	Den Haag	–	
O. Klepper	Den Haag	Mod	Model of the Westerschelde
J.H. Kluytmans	Utrecht	–	
M. Knoester	Middelburg	E	Research management
K. Kramer	Den Helder	C	Chemical speciation, biological effects
D.J. Kuenen	Leiden	E	
P. Kuyper	Vlissingen	–	
R.W.P.M. Laane	Den Haag	C	Chemistry of the marine environment
M.J.M. Langeveld	Biervliet	–	
M. Loyens	Brussel	C	Oceanography
A. Luyendijk	Middelburg	–	
E. Marteijn	Middelburg	E	Birds

Name	Address	Discipline	Research-topic
J.M. Martin	Montrouge	GC	Heavy metals, radionuclides
P. Meire	Gent	E	Birds, influence of pollutants
A.G.A. Merks	Yerseke	C	Chemistry of water
G. Moerland	Wageningen	H	Literature review
P.H. Nienhuis	Yerseke	B, MB	Elements cycling and food chains
J. Nieuwenhuize	Yerseke	C	Chemistry of sediments
E.H.T.M. Nijpels	Leidschendam	–	
C. Ockeloen	Terneuzen	–	
N. Oskam	Middelburg	–	
M. Ouboter	Delft	GC; mod	Model of the Westerschelde
R. Peelen	Yerseke	MB	Bacteria
A. Ringele	Brussel	C	Heavy metals in sediments
F. Ronday	Luik	–	
P. Ruardij	Texel	MB; mod	Model
J.W. Rijstenbil	Yerseke	B, MB	Phytoplankton
R.J. Scheele	Utrecht	–	
P. Servais	Brussel	Mic	Bacterial biomass and activity
J. Senten	Antwerpen	–	Mud-analysis
A. Smaal	Middelburg	MB	Carbon budget, benthos
M. Smies	Rotterdam	T	
J. Smitz	Luik	–	
K. Soetaert	Gent	E	Meiobenthos
P.J.A. Soons	Den Haag	–	
Mr. Spaas	Hoek	–	
W. Staal	Etten-Leur	–	
R.C.H. Steneker	Utrecht	GC	
F. Steyaert	Middelburg	–	
J. Stronkhorst	Middelburg	O	Primary production, freshwater-discharges
M. Tackx	Brussel	MB	Plankton-seston interaction
J.T. van Buuren	Middelburg	T	PCB's and PAK's in birds and benthos
J.H.M. van der Meulen	Middelburg	–	Waterquality
R. van Grieken	Wilrijk	GC	
R. van Haren	Den Haag	E	Heavy metals in mussels
G. van Hooren	Brussel	H	
D. van Maldegem	Middelburg	–	
W. van Overloop	Brussel	T	Ecotoxicology
M. Veldhuizen	Utrecht	–	
G. Verreet	Oostende	GC	Silt suspensions
F. Volckaert	Zellik	MB	Genetics and physiology of mussels
J.H. Vosjan	Texel	Mic	Bacterial respiration
C. Wauters	Brussel	–	
R. Wollast	Brussel	C	
D.I. Zandee	Utrecht	T	Impact of heavy metals on evertebrates
W. Zurburg	Yerseke	C	Heavy metals, speciation, bioavailability
J.J.G. Zwolsman	Middelburg	GC	Adsorption-desorption of heavy metals

Total		
93	18	C + GC
	33	B + E + H + MB + Mic
	10	T + Mod + O

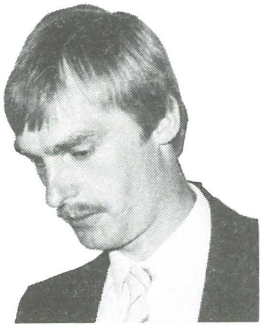
B	= Botany, algology	MB	= Marine Biology	O	= Marine optica
C	= Chemistry	Mic	= Microbiology	T	= Toxicology.
E	= Ecology	Mod	= Modelling	–	= other or unknown
GC	= Geochemistry				
H	= Hydrobiology				

Chairman and invited speakers of the morning session of the first day:

Drs. E. H. T. M. Nijpels

Dr. Ir. W. G. Beeftink
(chairman)

Dr. C. H. R. Heip



Dr. H. Hummel

Invited speakers of the afternoon session of the first day:

Dr. E. K. Duursma

Dr. G. Billen

Drs. R. C. H. Steneker

Dr. A. H. L. Huiskes

Speech of E.H.T.M. NIJPELS, Minister of Housing, Physical Planning and Environment on the occasion of the opening of the Schelde Symposium.

Ladies and Gentlemen,

It gave me great pleasure to accept your invitation to open this symposium on hydrobiology and the chemistry of the Schelde and the Westerschelde.

On the way here this morning, glancing again at the letter inviting me here and realising that the Delta Institute has its home in Yerseke, my thoughts automatically went back to what every Dutch child used to be taught at primary school about Yerseke : 'Yerseke is an extremely old, picturesque fishing village on the Oosterschelde with mussel beds but above all oyster beds'. However, the symposium is not being held in Yerseke but in Terneuzen on the Westerschelde, I assume for practical reasons. But one could also attribute a different and more symbolic significance to the choice of location in that it confronts us, in the most literal sense of the word, with the pollution problem of the Westerschelde. The fact is that the water quality plan for the North Sea states that on the basis of the concentration of polluted substances in mussels it can be concluded that the cadmium content in mussels from the Westerschelde is higher than in those in the Oosterschelde estuary and, what's more, that this concentration is higher than any other salt surface waters in The Netherlands. The figures are not much better for other metals and compounds.

It is not so very long ago that research institutes which were concerned with surface water concentrated almost exclusively on the environmental compartment, water, in the narrow sense of the word. Water as liquid containing in solution harmless or harmful substances. It is only in recent years that more interest has developed in the material in suspension in water and also in the underwater soil and the pollutants it contains. More recently still, people have come to acknowledge the possible significance of the atmosphere as a potentially major source of pollution, particularly for the large stretches of surface water.

The logo on the programme for the symposium is a figure with clear interfaces which could well illustrate the interconnection between the environmental compartments of Water, Soil and Air. The planes meet at the top of the pyramid, becoming integrated as it were in a single point. I would say that a great deal of research will have to be done at the interfaces of the different compartments.

An example of research of this kind is that into the scale and significance of the pollution of surface water from the air. I have already referred to it. Not much attention has been paid in the past decade to pollutant substances entering the surface water through this route, neither in the research sphere nor in the policy sphere. However, it is by no means inconceivable that further research will reveal that the significance of this transport route for a number of substances has been seriously underestimated. For policy to combat air pollution it might be extremely useful to know whether the pollution of the water through the atmosphere is on such a scale and has such an impact that it offers an additional argument for anti-pollution measures or, where necessary, for additional measures besides those which have already been taken, or will be taken, with regard to the environmental compartment air as such. Until recently we tended rather to belittle the impact of pollution from the atmosphere on surface waters because of the buffer capacity, because most of Dutch surface waters are running waters and because of the impact of other sources of pollution etc. Slowly but surely we are now becoming worried that we have been too optimistic on this point. In any event research will have to demonstrate what the state of affairs is, and that we have not underestimated the situation. Only a few years ago we would never have thought it possible that the soil and the minor inland waterlands, like the fens, could become acid through substances which ended up there through rainwater or by means of dry deposition. By now we know a lot better as far as that is concerned.

Another example of a research interface between the environmental compartments is the transition from fresh to salt water. Questions which arise are whether the pollutants adsorbed in fresh water silt are released again when the silt ends up in the sea and what effects does this have on the living organisms in salt water. The answers to questions of this kind can play a role in the decisions which are yet to be taken on urgent, practical matters such as which dredging mud may, and which may not, be disposed of at sea in the near future. In more general terms such information can be useful in drawing up emission standards and above all quality standards for salt surface waters.

I am perfectly aware that research into the two fields I mentioned above can be extremely complex. One can see indicentally from the survey of programmes that a number of researchers and institutions took up these items not so very long ago.

A third subject which has commanded more interest in recent years, particularly in an international context, will be one with which you are all probably familiar : the effects of eutrophication in fresh waters have been known and studied for many years but by now these effects have also been identified in salt waters and are already causing problems in some areas. Problems arise with enormous quantities of phosphates and nitrates which in combination with a relatively high water temperature can result in algae bloom in what are, in certain circumstances, virtually stagnant shallow waters and subsequently result oxygen depletion. The policy-makers have a clear need for a soundly underpinned standard for nitrogen and phosphate which will eliminate oxygen depletion and as result of it fish mortality in such vulnerable areas in the future. In the case of phosphate, the main question which arises is whether the measures which have been created to cope with the effects of eutrophication in fresh surface waters ought to be tightened up with a view to the salt water ecosystem. The consequences of quality standards being set for nitrogen are possibly even more serious. It could lead to standards for nitrogen contaminants for agricultural purposes and to large scale de-nitrification in treatment plants etc.

Having mentioned these three examples of pollution of the aquatic environment I may wrongly have created the impression that concern for environmental problems should exclusively concentrate on combatting pollution by all kinds of compounds. That is certainly not the case. There are still many other factors which are important. You need only think of the disturbance in the Wadden area caused by recreation, military exercises, shipping, of the risks attached to oil and gas extraction in the North Sea and the Wadden Sea, of sand and gravel extraction, of the fishing industry or, closer to home, of the closure of estuaries, the raising of dikes and the deepening of channels etc. In other words there are many more factors which can equally well exert a major influence on the salt water ecosystem. In view of the limited funds available it goes without saying that those measures which can be expected to produce the best result and which cost least are the ones which are eligible for implementing first.

Environmental protection is more than just regulating flows of substances. And while I am on the subject I should like to refer you to a report which was recently drawn up by the coordinating committee for the implementation of the Pollution of Surface Waters Act, (CUWVO for short), entitled Ecological Standard Objectives for Surface Waters in The Netherlands. The report starts with a description of the different types of waters, and goes on to give three levels which can be chosen for each type of water indicating at the same time roughly what steps one ought to take and what one should refrain from taking for these waters, notably in the sense of morphological interventions, changes in flow direction, institution of buffer zones, damming etc. In fact what this report does is give a kind of integral area-wise approach in which the possibilities of each area are examined. The report also contains a chapter on salt waters such as mud flats, creeks and tidal waters.

To take a local example to which the area-wise approach could apply, look at the Markiezaatsmeer. I think if all those involved could be convinced that an attractive ecosystem should and can be developed, actual steps can be taken to create a high-level ecosystem.

I have encountered the word 'modelling' on a variety of occasions in the programme. I hope that a scientific approach of this kind will also be placed in the context of risk evaluation and a description of the effects of calamities. What brings me to this subject is the recent accidents along the Rhine near Sandoz for one. It will certainly come as no surprise to you if I tell you that in the Schelde delta as well there are a number major Belgian and Dutch companies which could have an enormous impact on the entire area if a calamity were to occur. Looked at in this light, it is likely in the future that there will be a greater need for information of this kind. I would heartily recommend this as a matter for your attention.

Ladies and Gentlemen,

I have tried, very briefly, to give you an impression of a few subjects which might require a further scientific underpinning in the near future. I have not the slightest illusion that, having done so, I have referred to all the major topics. Nevertheless, I hope that I have made a contribution to the thinking on the possible directions in which further research can go.

I should like to close by wishing you every success with your symposium and in your further research work.

Thank you for listening.

HYDROBIOLOGICAL BULLETIN 22(1): 7 - 20 (1988)

EXCHANGE PROCESSES IN ESTUARIES SUCH AS THE WESTERSCHELDE, AN OVERVIEW *

E.K. DUURSMA [1], A.G.A. MERKS [2] and J. NIEUWENHUIZE [2]

KEYWORDS : estuaries; partition; residence times; metals; PCBs.

ABSTRACT

A short overview is presented on the principles of exchange processes of metals and PCBs for the Westerschelde, the estuary between the river Schelde and the North Sea. Critical reference is made to the commonly used interpretation of concentration - salinity mixing curves of dissolved material, and to the application of stability and distribution constants (coefficients) for metal species partitioning between dissolved, complexed and solid phases. A discussion is given on the specific PCB congener distribution, the impact of Kds on the solid/liquid percentual distribution in an estuarine system and the approach to determining the residence times of substances in an estuary.

INTRODUCTION

The Westerschelde is the estuarine part of the mouth of the river Schelde, which discharges into the North Sea (Fig. 1). The estuary suffers from a large anthropogenic stress due to the discharge of organic and inorganic contaminants from various effluents.

The hydrodynamic conditions, with upstream a tide of almost 5 metres for spring tide, are the dominant factors determining the geochemical, chemo-ecological and bio-ecological processes. Generally, estuarine exchange conditions are of a non-equilibrium type, so that the time-factor of the exchange processes should be taken into account.

This paper discusses a number of facets of importance in chemical exchange processes. First a somewhat different approach to the explanation of the much-used mixing curves is given, and next data on partitioning processes are presented.

ESTUARINE MIXING PROCESSES

The classical approach to understanding geochemical distributions of metals in estuaries is to compare the concentrations of these metals in solution with those in suspension over the length of the estuary (MART and NURNBERG, 1986), or to plot these concentrations as a function of the conservative property of freshwater-seawater mixing, salinity (DUINKER *et al.*, 1982). This approach is only correct for a river mixing with sea-water, if the particulate matter transport exactly follows the water mixing processes.

However, in most estuaries, including the Westerschelde, the course of things is different. The major reason is that the distribution of the sediment over the estuary, which is continuously or discontinuously in suspension does not follow that of the brackishwater mixing. The Westerschelde-Schelde river system shows a clear turbidity maximum at the low salinity part of the estuary, consisting of a mixture of both freshwater and marine sediments (SALOMONS and MOOK, 1977). But since the percentual mixing of freshwater and

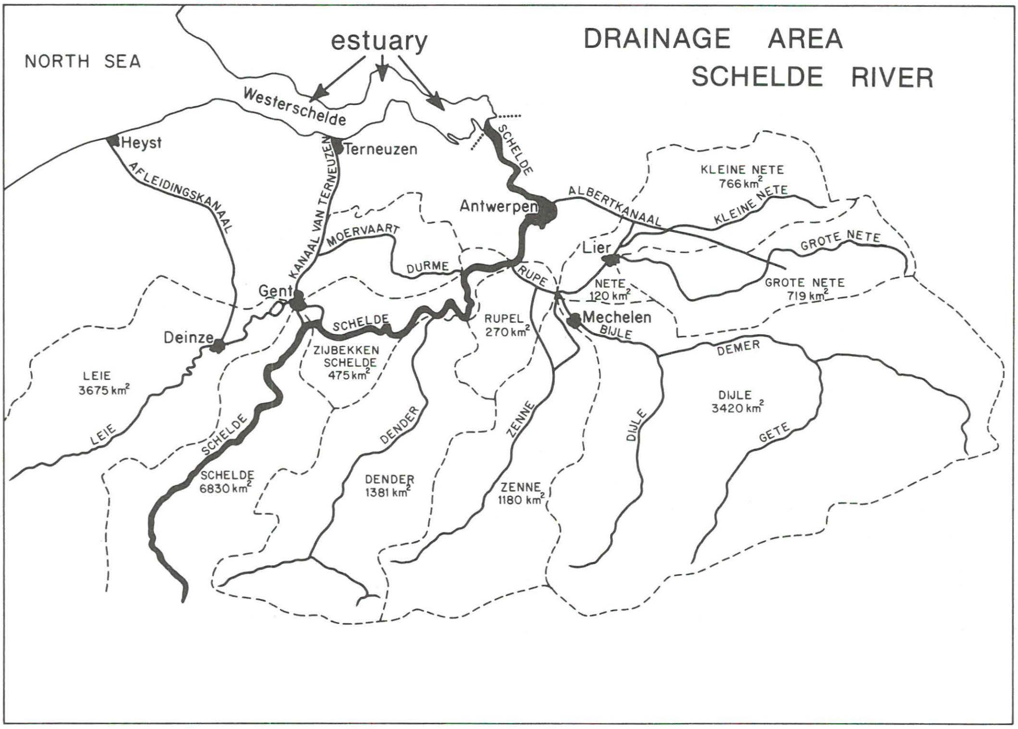


Fig. 1. Schelde and tributaries drainage area.

seawater, and that of freshwater sediments and marine sediments, differ, a disequilibrium exists between the dissolved and particulate substances (see Fig. 2A).

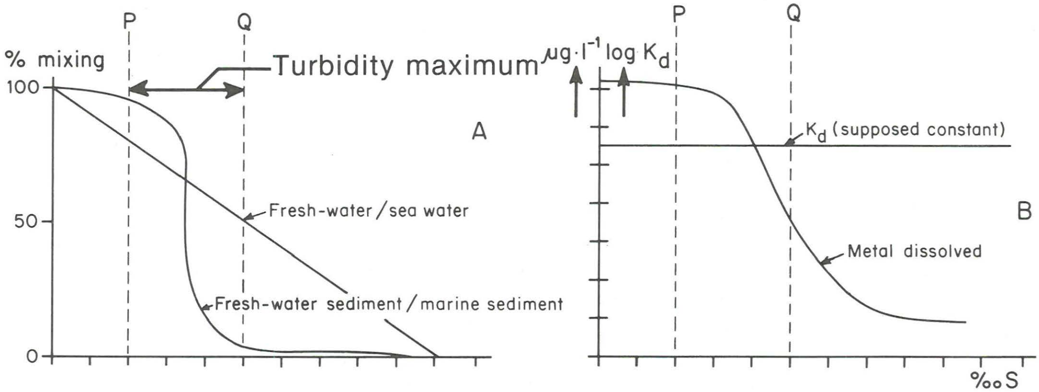


Fig. 2. A : Percentual mixing of freshwater and seawater, and freshwater sediment and marine sediment of a Schelde-type estuary. B : Theoretical concentration-salinity correlation of dissolved metal for a Schelde-type estuary, supposing Kd is constant and sorption-desorption to particulate matter is very rapid.

Assuming (i) that the underwater bottom of the estuary does not play a role, (ii) that the metal involved (for example cadmium or copper) can exchange rapidly between solution and particles, and (iii) that the partition coefficient Kd between particulate matter and solution is the same in the freshwater, brackish and marine system, the theoretical mixing curve for the dissolved and particulate metal can be calculated (Fig. 2B) (DUURSMA and RUARDIJ, in preparation).

Fig. 3 presents recently determined data on dissolved and particulate metals (from the original data of VALENTA *et al.*, 1986). It is clear that all kinds of curves can be made, based on either first, second or third degree correlations, but none of them will easily be explained on the basis of conservative or non-conservative mixing, due to the scatter of data.

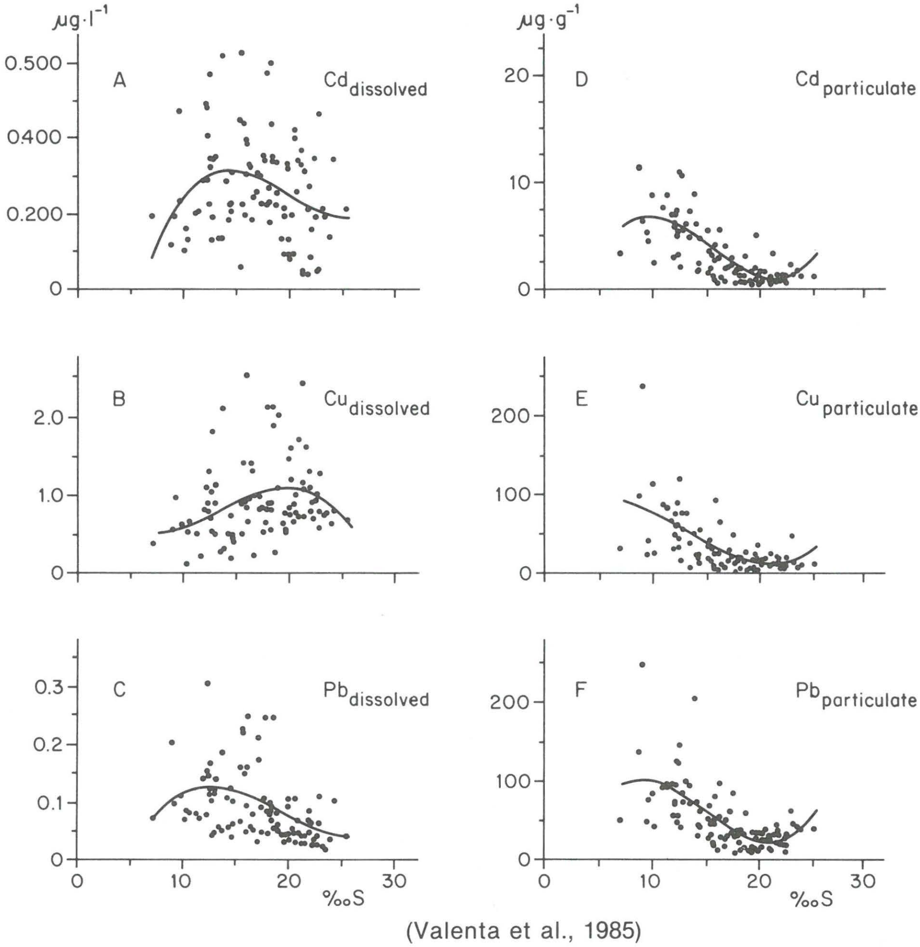


Fig. 3. A to F. Effective dissolved and particulate metal concentrations in the Westerschelde as measured by VALENTA *et al.* (1986) over a year, plotted as a function of salinity. The curves represent 2nd degree correlations.

PARTITIONING PROCESSES

The partitioning of metals, radionuclides and PCBs between the compartments : dissolved, particulate matter (seston), bottom sediments and organisms is determined by a number of basic hydrodynamical and thermodynamical laws, which to some degree can be applied for understanding the phenomena observed. In this context (Fig. 4) we will discuss : 1) partitioning between dissolved inorganic and organic species, where the word species concerns a chemical form of a metal, 2) partitioning of substances between water and particulate matter, and/or sediment or organisms, and 3) partitioning effected by the bottom.

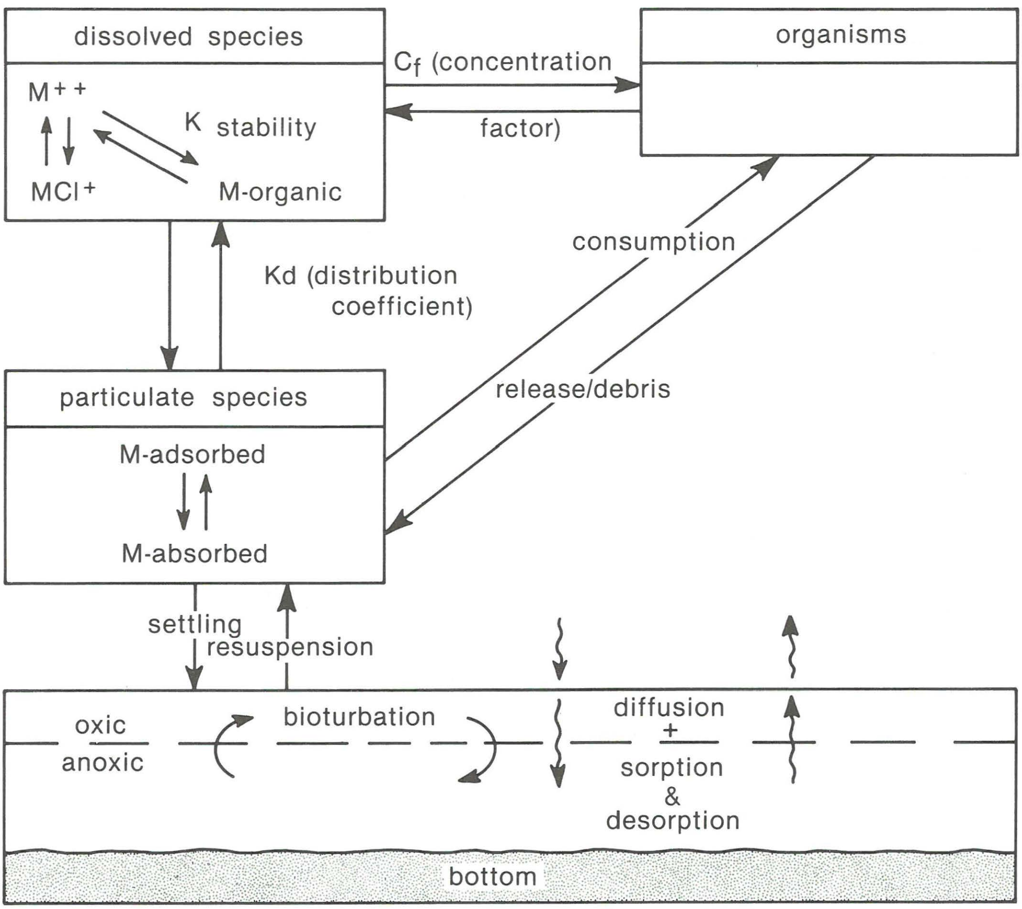


Fig. 4. Diagram of metal compartments in estuary (M=metal)

1) Species Partitioning.

Species partitioning is best illustrated by an example using the metal copper as described by KRAMER (1986). The partitioning of ionic metal to metal-complex is given by the reaction :

$$Mf + L' \rightleftarrows ML' \tag{1}$$

where Mf is the free ionic metal, L' the ligand and ML' the complex.

Thermodynamically the equilibrium of this reaction is determined by the equilibrium constant (or stability constant), K', which is :

$$K' = [ML'] / [Mf] [L'] \tag{2}$$

For natural aquatic systems, however, the situation is much more complicated, since for this type of reaction (1), there is a great number of different Mf ions that compete for perhaps even a greater number of organic ligands L'.

Fig. 5 gives a theoretical example of all metals present in seawater, when complexed with only one organic ligand, EDTA (DUURSMA, 1970). This figure shows that 'becoming complexed or not' depends on the EDTA concentrations and those of other metals. At low EDTA concentrations, only Fe is complexed - due to its highest stability constant - whereas at high EDTA concentrations all trace elements become complexed. Here, the major Ca and Mg metals - with low stability constants but high concentrations - are not yet completely complexed.

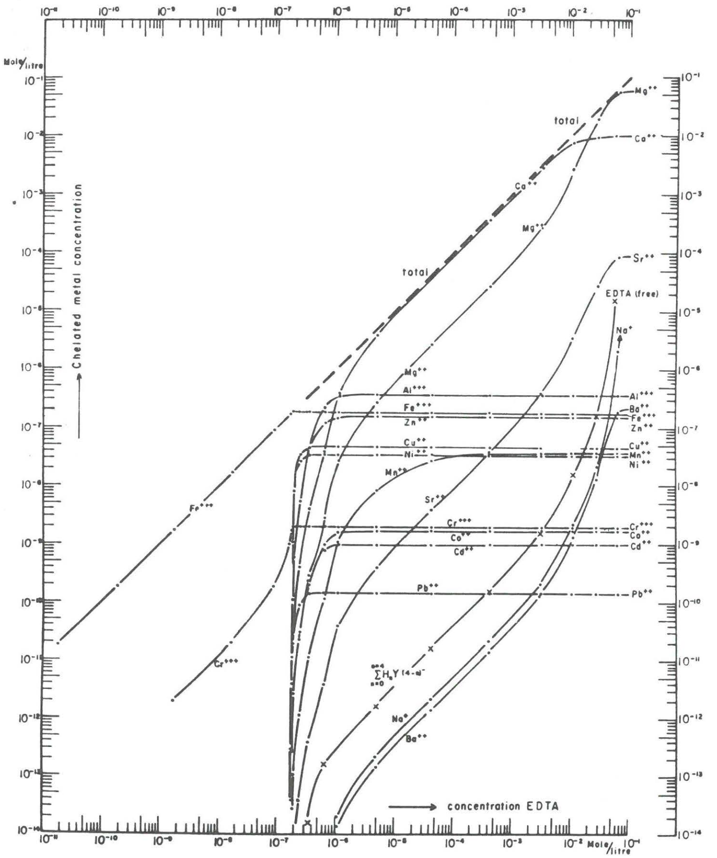


Fig. 5. Plot of total EDTA concentrations (abcis) to metal-EDTA complexes and free EDTA ($H_n Y^{(4-n)}$) (ordinate) in seawater (DUURSMA, 1970).

For copper, however, KRAMER (1986) determined that the apparent K′ for 21 stations in the North Atlantic were rather constant, and log K′ ranged between 7.7 and 8.3. Such high values for log K′ make it likely that copper can be complexed in favour of other metals. This assumption agrees with data published by other authors for estuarine systems (DUINKER and KRAMER, 1977; VAN DEN BERG, 1982).

A great mistake of many non-specialists is, however, to take the copper example as a rule for all kinds of complexing reactions in river and estuarine waters. Most untrue are the suggestions that organic matter is always 'very important' in this respect, when no real data are available. In general the stability constants of the other metals and dissolved humic and organic substances are some orders of magnitude lower (DUURSMA, 1970). Additionally, at low concentrations of both metal and organic matter there can be hardly any 'active' complexation. Here 'active' refers to the thermodynamically determined chelation or complexation, and not the formation of 'stable' complexes through biota.

2) Kd-distribution.

Sorption.

In the field of the partitioning of metals, radionuclides and PCBs between solution and solid matter some general opinions exist, not supported by chemical or biological evidence. The first obvious one is that adsorption occurs to only the finest particles. The second is the assumption that practically all aquatic substances are in either dissolved or particulate form, without evidence on the suspended load of the system.

These 'half-truths' are frequently used, although sufficient literature exists to give a better understanding of these phenomena (DUURSMA and EISMA, 1973). Just as for complexing, the partitioning of *e.g.* metals (M^{n+}) between solution and a solid phase can be described by :

$$Kd = [M\text{-solid}] / [M^{n+}] \qquad (3)$$

where Kd should be considered an apparent constant. Any deviation in Kd will then depend on the affinity of M^{n+} with the solid particles. A complicating factor is that in general the determined Kd is the ratio of the total metal concentration in the solid matter to that in solution, while M^{n+} may be in exchange with only M-solid. Additionally, organic complexes compete for M^{n+} (DUURSMA, 1970).

A problem for the validation and use of empirical Kds is that sorption-desorption reactions are time-consuming. This holds not only for reactions of dissolved metals with the surfaces of the solids, but also for their migration into crystal lattices. Under natural circumstances in estuaries, the mixing of freshwater and seawater is accomplished within a few days, and that of freshwater particulate matter and marine particulate matter takes months. Usually the interactions between a great number of metals and sediment particle surfaces reach equilibrium after some days. As we will see below, crystal lattice 'ab'sorption ranges over a much longer time.

Half-times of sorption.

Half-times of adsorption are indicative of sorption reaction velocities, and the type of the adsorption reaction. For 30 ocean and estuarine sediments and ca. ten metals added as radionuclide tracers, half-times ranging between 0.7 and 18 days have been determined.

The chemical receptivity of the surfaces and the type of sorption reaction are probably determining factors. Some of these reactions involve precipitation, such as for iron. Another differing reaction type as far as time and quantity are concerned is ionexchange (Cd and Cs).

For 'ab'sorption inside crystal lattices, grain size is of importance, as well as the diffusion coefficient of the particular element into the crystal lattice (Fig. 6). This figure shows

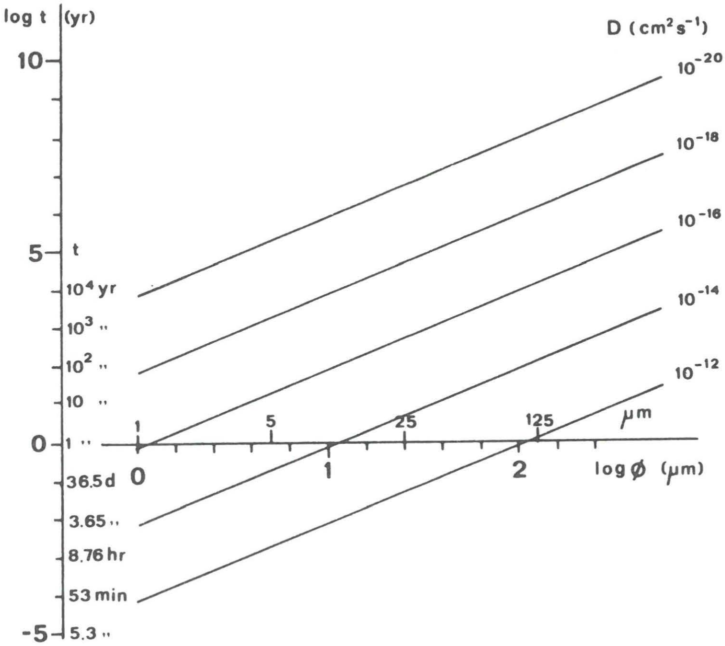


Fig. 6. Time for 90 % matrix (crystal lattice) absorption as function of grain size (ϕ) and diffusion coefficient (D). (DUURSMA and BEWERS, 1986).

that the time for 90 % absorption to small particles is much shorter than to larger particles. For example the 90 % times are, for a diffusion coefficient $D = 10^{-16}$ $cm^2.s^{-1}$, 1 and 10 years for particles of 1 and 5 μm, respectively. Hence for sediment in estuaries, usually being deposits of small particles, this type of 'ab'sorption cannot be neglected. Leaching experiments of metals from pre-contaminated sediments confirm that so-called non-loosely fixed fractions may conprise even 100 % (Table 1).

Table 1. Possible ease of release of some elements (in %) being sorbed for more than 7 months (DUURSMA *et al.*, 1975) by marine sediments excluding the original element present, as determined by leaching with acetic acid/NH_4- acetate (pH = 5.4), and acetic acid 1N (pH = 2.3).

	Elements											
Leaching Agent	Mn	Fe	Co	Cu	Zn	Sr	Ru	Ag	Cd	Cs	Ce	Pb
Ac. ac./NH_4-ac.	50	0	33	0	4-33	100	10	0	100	2	50	50
Ac. ac.	100	75	100	30	10-80	100	25	0	100	30	90	50

Redox conditions.

The Westerschelde estuary has an anoxic region at the low salinity part near Antwerp. The behaviour of metals in exchange processes may be different from that in oxic zones. For a number of metals the species composition in solution depends on the redox potential, while formation of insoluble sulphides may take place when H_2S is present.

Whether these metals also show different partition under anoxic and oxic conditions is another matter. As can be seen from Table 2, there are some differences, but these may not always be significant, due the low accuracy of the determination of the Kds.

Table 2. Distribution Coefficients (Kd) for 11 radioisotopes tracers as determined under oxygenated and anoxic conditions in Black Sea and Mediterranean sediment (DUURSMA and BEWERS, 1986).

	Kd			Kd	
Radionuclide	Oxic	Anoxic	Radionuclide	Oxic	Anoxic
Sr-90	1.2×10^2	4.7×10^3	Mn-54	4.6×10^4	4.3×10^3
Cs-137	1.4×10^3	3.1×10^3	Zr/Nb-95	1.2×10^4	9.0×10^3
Ru-106	3.4×10^4	9.0×10^3	Ce-144	9.0×10^4	1.2×10^4
Fe-59	1.2×10^4	4.0×10^3	Pu-237 (III)	1.6×10^4	1.9×10^4
Zn-65	3.0×10^3	3.2×10^4	Pu-237 (IV)	1.8×10^4	1.3×10^4
Co-60	6.0×10^3	1.4×10^4	Pu-237 (VI)	1.3×10^4	2.2×10^4
Pm-147	4.8×10^4	4.3×10^3			

Oxic : Dissolved oxygen concentration : 5 ml $O_2.l^{-1}$.
Anoxic : Hydrogen sulfide concentration : 2 - 3 mg $H_2S.l^{-1}$.

Percentual partition.

For budgeting and flux calculations of contaminants in estuarine systems, calculation of the percentual partition of metals is required. The total amount in particulate or dissolved form depends both on the Kd and the suspension load S. As given in Fig. 7 the percentual partition is log-log related with the suspension load S for a particular Kd value.

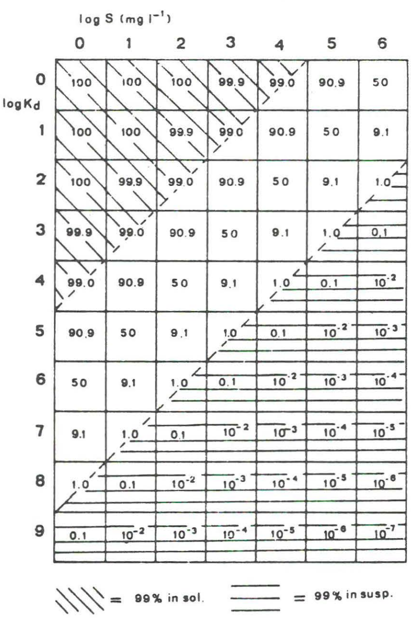


Fig. 7. Plot of percentual distribution of metals in solution as function of sediment load (S) and Kd (DUURSMA and BEWERS, 1986).

PCBs.

Essentially, the partition and exchange processes of polychlorinated biphenyls, PCBs, are not different from those of metals, and based on partitioning between water and particulate matter (seston), or water and fat in aquatic organisms (mammals excluded) (SCHNEIDER, 1980; DUURSMA *et al.*, 1986). For the Westerschelde the results (Fig. 8) clearly demonstrate that relatively constant Kds were determined for these two kinds of PCB partitioning, despite the high concentrations of PCBs in freshwater and low in seawater.

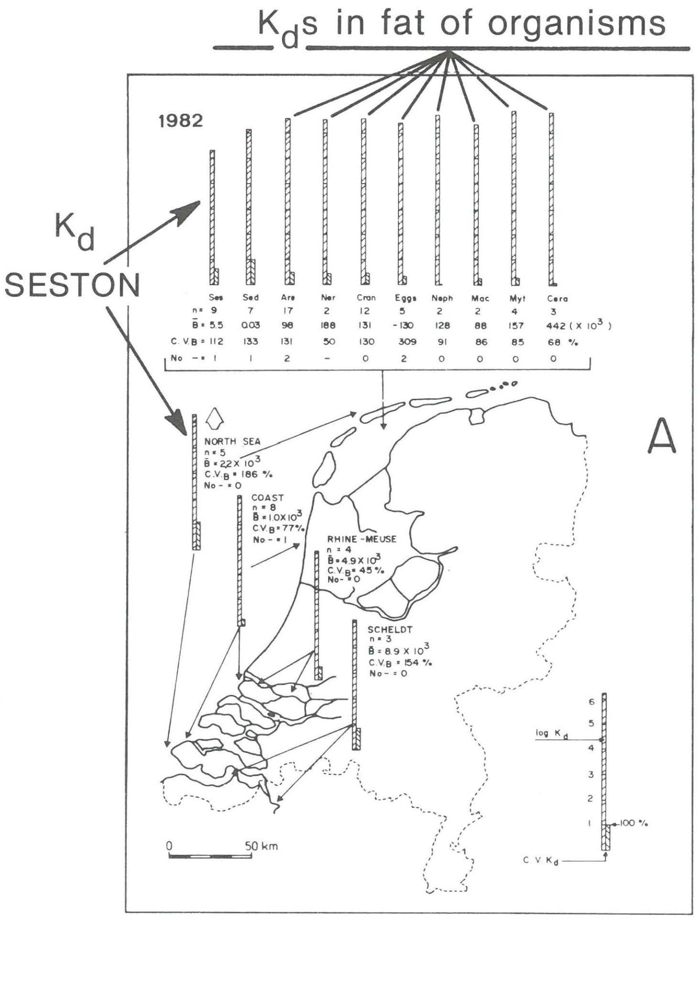
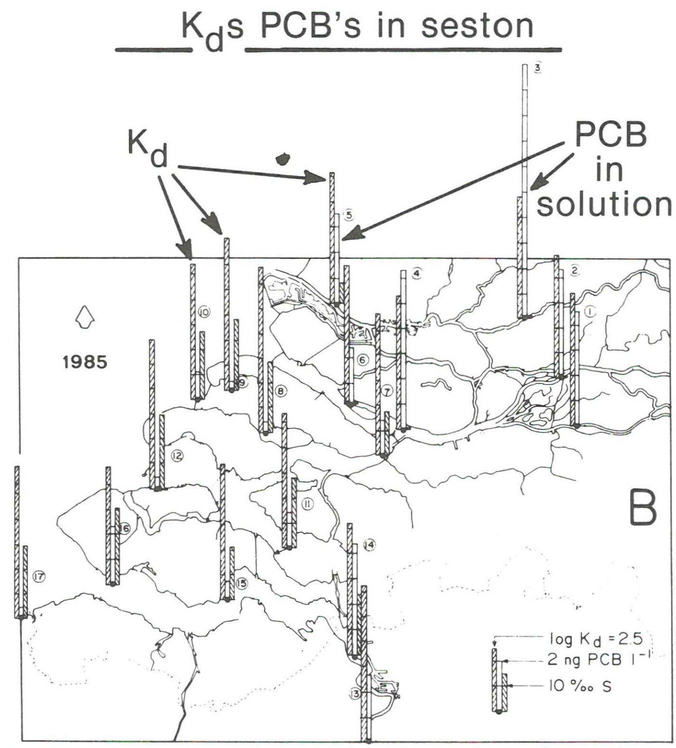


Fig. 8. A : Log Kd values of PCBs of water to substrate partitioning, coefficient variations (c.v. in %), number of measurements (n), regression coefficient (B) for the Kd-IUPAC Nr correlation of individual PCB components, c.v. of B and the number B had negative values. Kd for seston (Ses) and sediment (Sed) is based on dry weight concentrations this substrate. For the Waddensea organisms *Arenicola marina* (Ar), *Nereis virens* (Ner), *Crangon crangon* (Cran), eggs of this shrimp, *Nephtys hombergii* (Neph), *Macoma balthica* (Mac), *Mytilus edulis* (Myt) and *Cerastoderma edule* (Cer), Kd is calculated on fat weight. The bars of the North Sea, Coast, Rijn-Maas and Schelde concern water to seston partitioning.



Stations	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
B	2960	2080	150	4220	1050	6900	12300	14400	6920	1050	-1340	4650	-10600	309	494	99	-13000
P	***	***	n.s	***	**	***	*	n.s	n.s	n.s	*	*	*	*	n.s	n.s	***

Fig. 8. B : Concentrations of dissolved PCBs (of all peaks), log Kd for PCBs of water to seston partitioning, and salinities, presented as bars. Regression coefficient B and statistical probability p of the Kd-IUPAC Nr correlation of individual PCB components. p 0.001 = < ***, p<0.01 = **, p<0.05 = * and p>0.05 = n.s. (not significant). Waters studied are from south to north (with station numbers in brackets) Schelde (13), Westerschelde (14, 15, 16), Oosterschelde (11, 12), Volkerak (7), Lake Grevelingen (8, 9), Hollands Diep (4), Haringvliet (6), Maas (1), Rijn branches (2, 3, 5), Coast (10, 17) (DUURSMA *et al.*, 1986).

Interestingly, the specific partition of the different PCB congeners did not always give higher Kds for less polar congeners as given by their IUPAC numbers and their octanol to water partition coefficient, Kow. Kow is proportional to the increasing IUPAC numbers and should be linearly correlated with the Kds for the partition water to seston and water to fat in organisms, if polarity were the only determining factor. Although such significant correlations were found, they were positive (DUINKER, 1987 for the Elbe) as well as negative (NIEUWENHUIZE, pers. comm. for the Schelde river).

Both results may, however, be realistic, since the correlation curves of Kds with either Kow or IUPAC numbers of all partition results of the Dutch coastal system (DUURSMA *et al.*, 1986) were S-shaped when they were plotted as second or third degree correlations (Fig. 9).

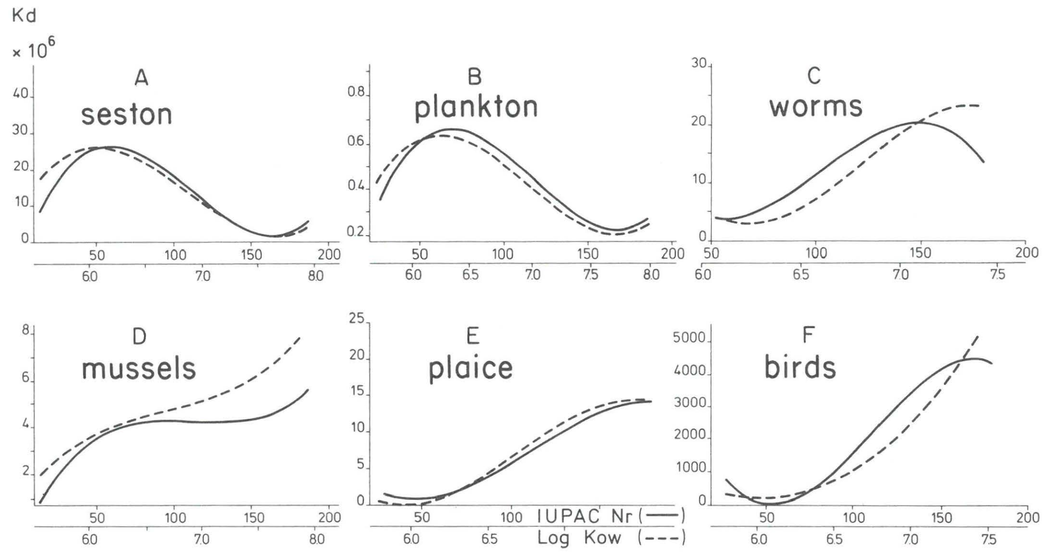


Fig. 9. Third-degree plots of Kd-IUPAC No correlations of PCB congeners for water to seston and fat in organisms (solid line) and of third-degree Kd-log Kow correlation (dashed line).

An explanation of the S-shaped correlations has not yet been found. Methodological factors, such as separation of dissolved and not-dissolved PCBs may be the cause, as well as a PCB congener dependent metabolic exchange between the different phases (BOON and EIJGENRAAM, 1988).

3) Residence times in the bottom.

Estuaries such as the Westerschelde have suffered from contamination for about 100 to 150 years. A major question is whether this uploading of metals and organics (above their natural levels) will continue for the decades to come, or whether the system has already reached an equilibrium as far as input and output are concerned.

On the basis of the hydrodynamic model of DITORO (1984), *cf.* also VAN ECK *et al.* (1986), VAN DE VRIE and DUURSMA (1986) developed a model for the Zoommeer, applying data on dissolved and particulate metal concentrations, settling of particulate matter and diffusion into the bottom. The model was calibrated on a three-box model with existing data on the Haringvliet-Hollands Diep. The goal was to estimate the residence and 50 % adaptation time of contaminants. These times give an indication of the velocity at which material passes through the system in steady-state conditions (Fig. 10).

The times predicted for the Zoommeer, supposing a contaminated water flow of 50 - 150 $m^3.s^{-1}$, and applied for a two-compartment system, are given in Table 3. For dissolved zinc, cadmium and lead the residence times in this 80 km long system are 17 to 40 years, while those in the top 10 cm of the bottom layer are close to a thousand years. The 50 % adaptation times are not much different for water, but for the top layer of the bottom they are in the order of 25 to 70 years. These ranges partly match those of HENDRIKSMA and AL (1982), who give values of 100 to 1000 years for the bottom. However, the figures given clearly depend on the model boundary conditions taken.

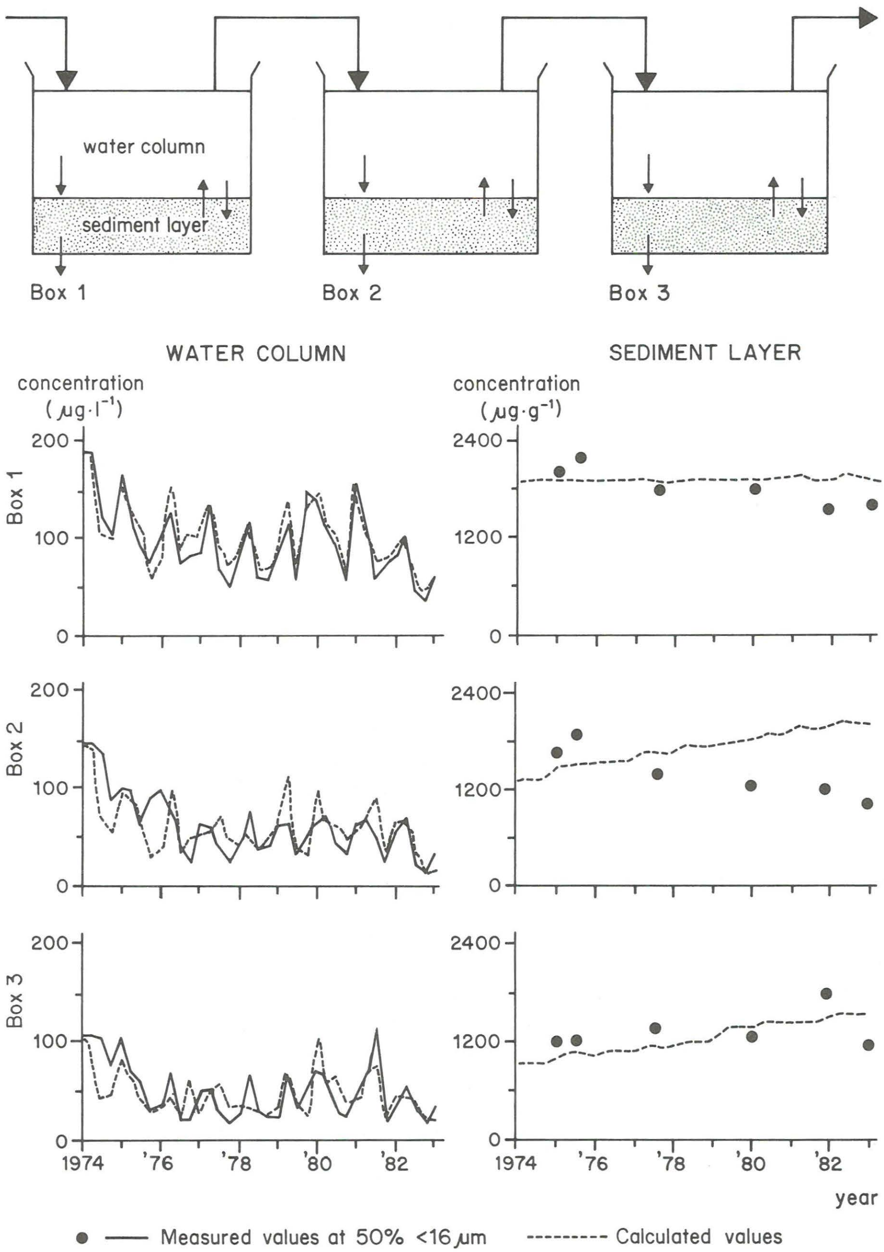


Fig. 10. A 3-box model for the Hollands Diep-Haringvliet (The Netherlands) zinc concentrations in water and sediment, based on data of the period 1974 - 1983; VAN DE VRIE, unpublished.

The Schelde river input in the Westerschelde, has an average flow of about 100 $m^3.s^{-1}$ and the order of magnitude predicted for the Zoommeer may have some value for this estuary. Anyhow, if either the Rijn or the Schelde were to become clean in the next few years, the cleansing of the underwater bottom would take decades or even centuries.

The Schelde river and Westerschelde estuary may be better understood, if the available knowledge on partitioning of contaminants and their residence or adaptation times were applied in hydrodynamical and geochemical models. This application would be of particular importance for the underwater bottom with respect to its uploading with metals and organic contaminants. Detailed modelling should be carried out, including all processes caused by the irregularly performed dredging and sediment deposition operations.

Table 3. Predictive residence (turn-over) and adaption times in the Zoommeer (VAN DE VRIE and DUURSMA, 1986).

	Turn-over times (years)				50 % Adaptation times (years)			
	Water column		Sediment layer		Water column		Sediment layer	
	Comp. 1	Comp. 2	Comp. 1	Comp. 2	Comp. 1	Comp. 2	Comp. 1	Comp. 2
Zinc	42	17	975	1230	18	37	24	45
Cadmium	42	17	975	1230	34	68	35	69
Lead	39	17	975	1230	28	59	36	70

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THE SOIL ENVIRONMENT OF THE INTERTIDAL AREA IN THE WESTERSCHELDE

OENE OENEMA, ROB STENEKER and JAAP REYNDERS

KEYWORDS : estuary; intertidal area; soil environment; biogeochemical processes; heavy metal contamination.

ABSTRACT

Hydrodynamic forces and sediment discharges determine the sedimentary environment and surface morphology of the intertidal area in the Westerschelde estuary in the S.W. Netherlands. Sandflats (clay content < 8 %) are found in the central part, mudflats (> 8 % clay) and especially salt marshes (> 15 % clay) occur in sheltered places.

The semi-terrestrial soil of the intertidal area is a complex environment where hydrosphere, biosphere, lithosphere, and atmosphere interact. Surface morphology and the semi-diurnal tide may significantly modify this interaction pattern. Organic matter and constituents of the clay-size fraction are the most reactive compounds in the soil.

The pathway and rate of organic matter decomposition determines the nature of many biogeochemical processes. Sulfur transformations are dominant processes in the sulfidic grey-black coloured horizons underneath the generally thin yellow-brown surface layer. The sulfur geochemistry also affects the sorption-behaviour of metals. Oxygen intrusion by diffusion, burrowing fauna and plant roots cause a local and sometimes temporary oxidation of reduced inorganic compounds. Thus, in the surface layers, and in and around tubes and roots at greater depth in the soil, a dynamic cycling occurs of redox active compounds, which affects the sorption mechanism of *e.g.* metals-ions.

INTRODUCTION

In the lower estuary of the Schelde river, the Westerschelde in the S.W. Netherlands (Fig. 1), extensive intertidal flats and salt marshes (11000 ha) are found. These areas have a typical landscape-morphology; they are important for biota. Below the surface, interaction occurs between the lithosphere, hydrosphere, biosphere and atmosphere; the product of these interactions is called the semi-terrestrial soil. Following PONNAPERUMA's (1972) discussion, semi-terrestrial (sub-aquatic) soil also applies to a sediment and therefore we use the term semi-terrestrial soil. A soil is defined here as the three dimensional space below the surface; it comprises all inorganic and dead-organic compounds in solid, aquaeous and gaseous phase. It houses a biological community and exhibits a clear biological and micro-biological activity. A semi-terrestrial soil is subject to semi-diurnal flooding.

The Schelde estuary has main functions for navigation and for drainage of the Schelde river, carrying the discharge of industrial and densily populated area's in its catchment. Significant changes occur as regards dissolved and particulate compounds in the water-compartment and most studies deal with this compartment. It seems, however, that the soil-environment has been overlooked.

The soil in an estuarine ecosystem has three main functions : - source and sink function for *e.g.* sand, mud, organic matter, nutrients, and contaminants; - regulation function :

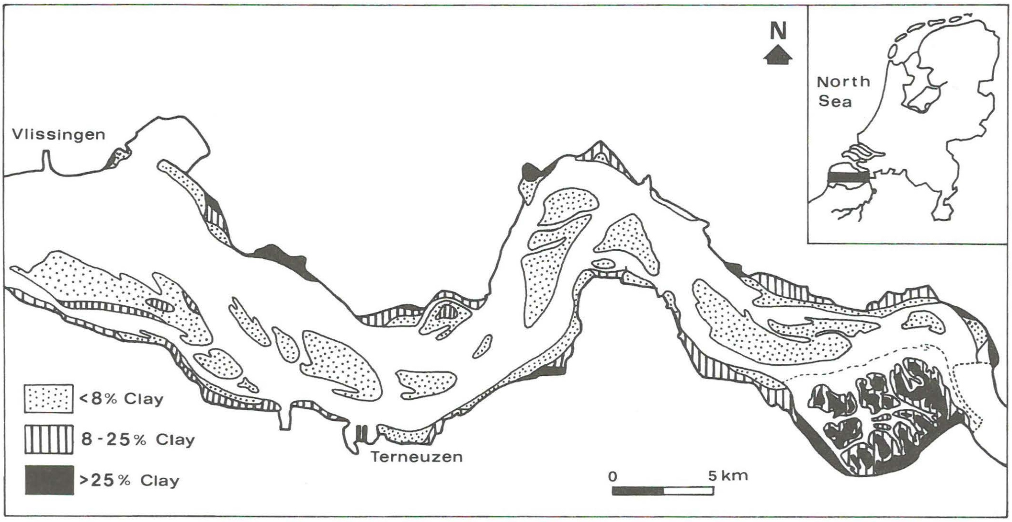

Fig. 1. Sedimentary grain-size distribution in the intertidal area of the Westerschelde.

it acts as a biological waste-treatment and several biogeochemical processes adjust and transform both the solid and dissolved constituents; - carrying function : the soil is the habitat of vegetation and zoobenthos and the fundament of several human activities.

Moreover the soil gives site-specific information on environmental processes in a historic perspective.

In the following paragraphs some characteristic features of the soils of the intertidal area in the Westerschelde are discussed. The multi-functional character of the soil and its complexity are to some degree illustrated by the discussions about the effect of sedimentation processes and surface morphology on the soil. Human influences are among other things reflected in heavy metal contamination.

This work is based on field studies in the Westerschelde since 1984 and on experiences obtained elsewhere.

SEDIMENTATION OF SOLID MATERIAL

Sedimentation of solid material on the intertidal area has two important consequences : - accretion of the surface in order to keep pace with or even exceed the relative sea-level rise; a steady supply of new, reactive materials which activate the biogeochemical processes occurring during early diagenesis in the semi-terrestrial soils.

The relative sea-level rise in the Westerschelde at Vlissingen is 0.22 cm y^{-1}, but the rise in mean high water level approaches 0.4 cm y^{-1} during the last century (DE RONDE, 1983). This faster rise results from embankments, and thus from the subsequent decrease in basin storage capacity, and from dredging in tidal channels, causing an increase in tidal volume. The sediment supply in the Westerschelde exceeds 4 - 8 times the demand of approximately 0.2 - 0.4 10^6 m^3 y^{-1} of the intertidal area required to keep pace with the relative sea-level rise. High accretion rates of 2 - 4 cm y^{-1} have been found in the marshes at Saeftinge during the last few decades (pers. comm. Frank Steyaart). Several major mud sources can be distinghuished (*e.g.* VAN MALDIGEM, 1987) : - discharges by the Schelde river (0.7 - 1 10^6 ton y^{-1}); - internal sources (primary production, dredging and erosion); - import from the North Sea (approximately 0.6 10^6 ton y^{-1}).

The fluviatile contribution to the recent deposits rapidly decreases from approximately 70 % at the Belgian-Dutch border to some 5 % at Vlissingen (ANONYMOUS, 1981). In the Land van Saeftinge a major part of the fluviatile Scheldt mud is deposited whereas the deposits in the more western part of the estuary are predominantly of marine origin.

Three major components can be distinguished in the soils of the intertidal area : detrital, biogenic and authigenic material.

Detrital material consists mainly of quartz grains in the sand fraction and of alumino-silicates in the clay fraction. Clay minerals have a large specific surface area and are ion exchangers. Therefore, they determine the physical and physico-chemical characteristics (porosity, consistency, adsorption of metals and organic molecules) of the soils to a large extent. Dominant clay minerals in the lower estuary are smectites, vermiculites and illites, with smaller amounts of kaolinites and chlorites (REYNDERS *et al.*, 1985).

Biogenic material comprises carbonates, opaline silica (mainly diatoms) and organic matter. However, a large percentage of the carbonates and the organic matter is associated with the detrital material, and thus originates from fluviatile discharges and eroding marine sediments (SALOMONS, 1975; MARIOTTI, *et al.*, 1984). Addition of biogenic carbonate by shell-fishes mainly occurs in the more marine, western part of the Westerschelde, where abundant cockles (*Cerastoderma edule* L..) have been observed (REYNDERS *et al.*, 1985) and where higher carbonate contents in the soils are found (DE LOOF, 1978, 1980).

High discharges of organic matter in the upper estuary cause a serious depletion of oxygen in the water phase. The long residence time of the water promotes the mineralization of the metabolizable fraction of organic matter in this part of the estuary. Finally, the residual organic matter entering the lower estuary is predominantly non-metabolizable (refractory). Fresh organic matter derived from in-situ primary production in the estuary has a higher reactivity, generally expressed by a lower C/N ratio. A high organic matter supply occurs in salt marsh soils (*e.g.* GROENENDIJK, 1987). The amount and reactivity of organic matter in the soil determines the rate of biogeochemical processes to a large extent.

Authigenic material results from precipitation of dissolved constituents in pore waters. Such metabolites are released to the pore waters during the microbial decomposition of organic matter and the herewith associated electron transfer (SUESS, 1979). Authigenic minerals (*e.g.* Fe and Mn sulfides, fosfates and carbonates) constitute only a small percentage (< 5 %) in the soil.

The environment of deposition is of great importance in determining the physical character of the deposit, *e.g.* its grainsize distribution, clay and organic matter content. In the lower Scheldt estuary, extending from Vlissingen to the Belgian-Dutch border the local sedimentation environment is to a large extent defined by the river discharge, current velocities, exposure to wave attack and dredging activities. Due to high current velocities, very low (< 1 %) clay contents are found in active tidal channels (DE LOOF, 1978, 1980; WARTEL, 1977). To the west a slight coarsening of the sand fraction has been observed due to import from the North Sea.

On the intertidal flats the deposits range from the predominant homogeneous well sorted fine sand (125 - 250 μm) to heterogeneous, poorly sorted silty clay (< 53 μm).

SOIL CHARACTERISTICS IN RELATION TO LANDSCAPE - MORPHOLOGY

The geomorphology of the intertidal area in the Westerschelde is determined by the hydrodynamic conditions, sediment discharge and human influences. Several morphological units can be distinguished at the intertidal flat area, each with a characteristic surface morphology, soil-type and vegetation (KOOISTRA, 1983). A transect across the lower estuary depicts the location of these units (Fig. 2). In the middle of the estuary sandflats with sandy soils (clay content < 8 %) occur. Towards the fringes mean current velocities decrease and the clay content increases to > 8 % in mudflats to > 15 % in salt marshes.

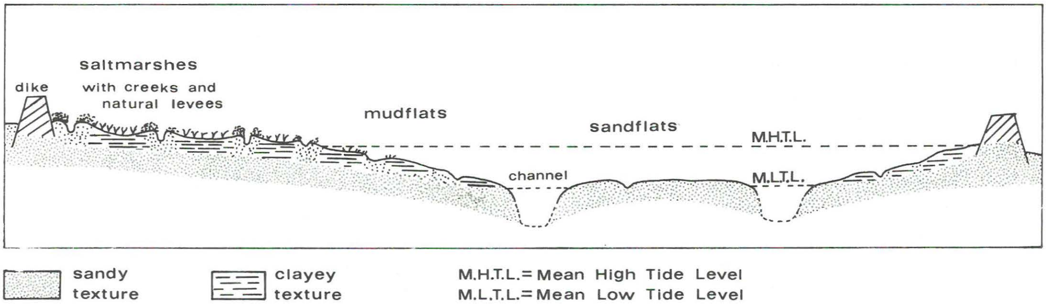

Fig. 2. Morphological units in a cross-section of the lower estuary.

The percentage intertidal area and especially the salt marsh area gradually increases from Vlissingen to the Belgian-Dutch border. Salt marshes occur in sheltered places and are only dominant in the eastern part (Land van Saeftinge). Sandflats are found throughout the whole lower estuary, in the western as well as in the eastern part, which points to dynamic sedimentation conditions in the middle of the lower estuary. The large tidal range and high current velocities contribute to these dynamics. Obviously, the variation in hydrodynamic conditions and in clay content is more pronounced in a transect across the lower estuary than along the estuarine gradient from Vlissingen to the Belgian-Dutch border (Fig. 1).

Some general soil characteristics are shown in Tabel 1. The clay content is a key variable; several soil characteristics are related to it. For example, with an increase in clay

Table 1. Some properties of the topsoil (0 - 10 cm) at the different morphological units.

	sandflats	mudflats	salt marshes
surface area (ha)	5668	2572	2775
clay fraction (%)	< 8	> 8	15 - 50
organic-C (%)	< 1	< 5	2 - 7
$CaCO_3$ (%)	5 - 25	5 - 20	10 - 25
copper (ppm)	< 10	10 - 20	10 - 100
lead (ppm)	< 30	25 - 60	50 - 275
cobalt (ppm)	< 10	5 - 20	15 - 40
cadmium (ppm)	< 5	< 10	2 - 15
chromium (ppm)	< 100	80 - 200	150 - 600
nickel (ppm)	< 25	20 - 75	50 - 250
manganese (ppm)	< 200	150 - 500	200 - 2500
zinc (ppm)	< 400	200 - 600	200 - 900

content porosity, organic matter content, carbonate and metal content increase. To facilitate comparison between different soils with respect to *e.g.* organic matter or metal content, it is a common practise to express these data on a standard clay content. Using such a correction, it has been shown (*e.g.* REYNDERS *et al.*, 1985) that the organic matter content of salt marsh soils exceeds the organic matter content in mudflats. This points to organic matter addition in the salt marsh soils.

Soils are generally not homogeneous with depth. Soil characteristics may change with depth not only because of variations in clay content : biogeochemical processes and the effect of bioturbation, plantroots and a varying groundwater table also contribute to these variations. At the surface of sand- and mudflats a yellow-brown coloured horizon depicts the layer with a dynamic cycling of nitrogen, manganese, iron and sulfur (see below). This layer should not be confused with the depth of oxygen penetration. Generally, it is an accumulation zone of solid phosphorus, manganese and iron. Underneath, a grey-black coloured sulfidic horizon is found where sulfur transformations are dominant biogeochemical processes. In between a gray-black coloured horizon with yellow-brown iron oxi-hydroxide mottles may occur, due to partial re-oxidation of reduced inorganic compounds caused by oxygen intrusion. Such a transition zone has been observed in some strongly bioturbated sand- and mudflats. Stable polychaete tubes permit rapid air intrusion into the soil. A similar effect occurs in salt marshes where oxygen release by some plant roots may oxidize the reduced inorganic compounds. Consequently, in and around these tubes, roots and mottles comparable cycling processes may occur as in the yellow-brown surface horizon. A good correlation has been observed between the redox potential (Eh value) and the colour of the soil horizon.

The groundwater table in the semi-diurnal flooded sand- and mudflats does not sink more than 0 - 2 cm below the surface during low tide, except at the fringes and along gullies intersecting the flats. Water table variations depend on the hydraulic head and the saturated pore water flow. In sand the saturated pore water flow is high (some metres per day) due to large pore sizes, permitting rapid drainage of the pores during low tide at the fringes of the sandflat. At greater distance from the fringe the hydraulic head is low (Fig. 3) and the groundwater table does not penetrate more than a few millimeter. This variation is characterized by thick (>10 cm) yellow-brown surface layers, generally with a convolute structure (DE BOER, 1979) at the fringe and thin (<2 cm) surface layers more ahead. Areas with a thick (>10 cm) yellow-brown surface layer are very poor in burrowing macrozoobenthos. In contrast, at some distance (5 - 100 m) from the fringes, occasionally high densities of *e.g. Cerastoderma edule, Arenicola marina* and *Nephtys hombergii* were found, in combination with a slightly higher clay content in the soil. In mudflats the saturated pore water flow is very low (some mm per day) due to the presence of very small interstices. Consequently, the drop in the groundwater table does not exceed a few millimeter during low tide, regardless the hydraulic head. Mudflats generally contained relatively high densities of macrozoobenthos.

In the salt marshes, the flooding frequency depends on the height of the marsh surface relative to NAP (Ordnance Level). Low marshes are flooded several hundred times a year and the depth of the lowest groundwater table does not exceed 10 cm in the backmarsh and approximately 40 cm in the natural levees bordering the creeks. In the natural levees both the sandy texture of the soil and the large hydraulic head permits better drainage than in the backmarsh. High marshes (*e.g.* in the Land van Saeftinghe) have only a few inundations per year, and during neap tides the groundwater table is found at $\geq$ 50 cm in the backmarsh, up to 1 metre in the natural levees. After a subsequent flooding the groundwater table may rise to the surface again. This causes a periodic shift in the reduction and oxidation of the redox active compounds. The marsh vegetation promotes the lowering of the groundwater by water-uptake; it stimulates the reduction-oxidation processes by organic matter addition and oxygen release via the root system. These effects are much smaller in stands of the pionier *Salicornia maritima* then in a dense stand of deeply rooted *Spartina anglica.*

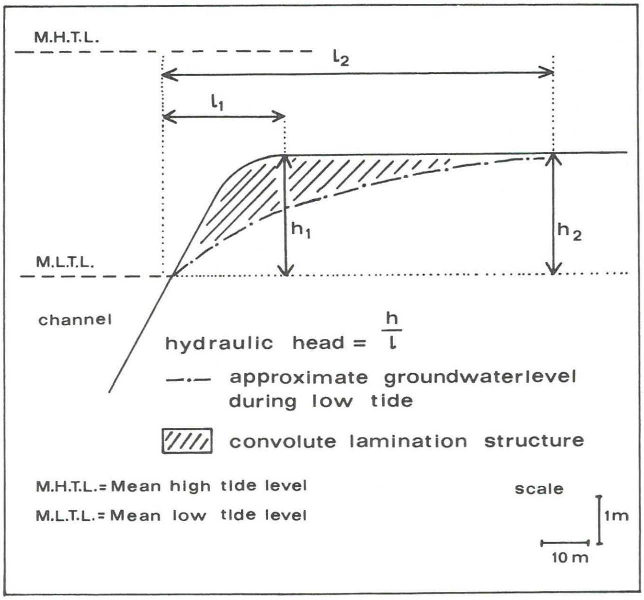

Fig. 3. Groundwater variations at the fringe of a sandflat.

BIOGEOCHEMICAL PROCESSES

The rate and pathway of organic matter decomposition in semi-terrestrial soils determines the nature and rate of related biogeochemical processes. High rates of organic matter oxidation, usually associated with a rapid sedimentation, causes oxygen depletion at shallow depth and thus anoxic conditions below. Since the introduction and subsequent more widespread use of micro-electrodes (*e.g.* REVSBECH *et al.*, 1980) it has been shown that the penetration of oxygen into estuarine and marine sediments usually does not exceed a few millimeter, except in the vicinity of animal burrows and living roots. In the anoxic zone underneath, subsequently NO_3^-, Mn^{4+}, Fe^{3+}, SO_4^{2-}, and finally CO_2 may function as oxidants. Denitrification occurs at the boundary of the oxic and anoxic zones, usually in a zone of less than 1 cm. Manganese and iron reduction is evidenced by typical subsurface maxima in dissolved manganese and ferrous-iron concentration. Due to its abundance sulfate is the dominant electron acceptor in marine and estuarine sediments. In many nearshore marine and estuarine sediments sulfate reduction accounts for more then 50 % of the total organic matter decomposition. Sulfate reduction and the subsequent reactions involving sulfide precipitation and partial oxidation are therefore dominant biogeochemical processes.

The sulfur cycle in sediments is complex and not yet completely understood. Reduced sulfur compounds store energy from the oxidized organic matter until the reduced sulfur species are oxidized again. Several studies (HOWARTH, 1984) have shown that over 90 % of the total sulfide production is reoxidized at the redox-discontinuity. The exact pathway of the sulfide oxidation is not clear however, but it has been speculated that chemolithoautotrophic bacteria may play an important role, utilizing the energy released by the sulfide oxidation. Such a microbial colony of chemolithoautotrophic bacteria at the transition of the oxygen and sulfidic zones may serve as food for *e.g.* deposit feeders at and in the sediment surface layers. Thus, energy stored in low quality (refractory) organic matter in the subsurface layers is transferred into high quality (easy metabolizable) organic matter in the surface layers via the sulfur cycle. Clearly, more research is needed to verify such mechanisms.

Under anoxic conditions metal sulfides (*i.e.* mainly iron, but also 'heavy' metals) have a low solubility, thereby adjusting the dissolved sulfide and metal concentrations to relatively low levels. This reduces the risk for vegetation and zoobenthos, because both dissolved sulfide and certain metal-ions are toxic at high levels.

Pyrite (FeS_2) is the major reduced sulfur compound. The pyrite contents in the semi-terrestrial soils are strongly related to the organic carbon contents, being highest in the salt marshes. In the sand- and mudflats pyrite contents vary between 0 - 0.5 %, in salt marshes between 0.1 - 2 %. A typical pyrite profile of a middle high salt marsh is shown in Fig. 4a. In the surface layer (0 - 2 cm) rather high contents (0.3 - 0.5 %) are found, because

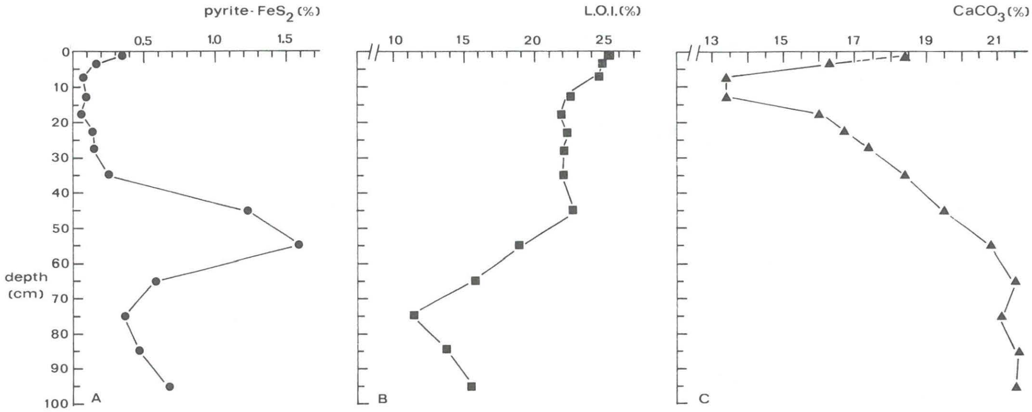

Fig. 4. Distribution of pyrite (a), loss on ignition (b) and carbonate in a salt marsh soil profile. (Note the break in the concentration scales in b. and c.).

of sedimentation of marine sediments on this marsh. In the upper rooting zone (0 - 30 cm), pyrite oxidation caused by oxygen release from the *Spartina anglica* roots, is dominating over pyrite formation and pyrite contents are low. Below a depth of 30 - 40 cm, in the sulfidic horizon, pyrite formation is dominant and pyrite contents increase to 1.5 %. At depth, another decrease is observed because of a decrease in both clay and organic matter content, illustrated by the decrease in loss on ignition (L.O.I.) at 950 °C (Fig. 4b). Such a depth distribution of pyrite has also been observed in salt marshes in the Oosterschelde (OENEMA, 1986).

The intensity of sulfide oxidation is related to the growth dynamics of *Spartina anglica* and the flooding frequency and may vary seasonally, being highest in the summer (HOWARTH and TEAL, 1979). This periodicity in oxidation affects the sorption of heavy metals, but also causes dissolution of carbonates. Reduction of sulfate generates an equivalent amount of bicarbonate according to the following simplified reaction :

$$2CH_2O + SO_4^{2-} = H_2S + 2HCO_3^-$$

These bicarbonate ions are consumed again during the subsequent oxidation (VAN BREEMEN *et al.*, 1983). However, in the reduction stage a downward transport of pore water (mass flow) causes a partial leaching of bicarbonate ions. Consequently, the protons generated during the oxidation stage cannot be neutralized completely by bicarbonate. The excess protons now causes a partial dissolution of the carbonates (Fig. 4c).

HEAVY METAL CONTAMINATION

Industrial and domestic discharges in the Schelde basin cause a substantial contamination with heavy metals. The main input of metals occurs in the upper estuary. During transport, complexation and (de-)sorption processes determine the distribution of metals between the liquid- and solid phase. Heavy metals tend to associate strongly with constituents of the clay fraction and a strong correlation between the total amount of metals and the clay content has been found. Sedimentation of fluviatile suspended matter is only dominant in the eastern part of the lower estuary and the fluviatile compound in the semi-terrestrial soil decreases rapidly downstream. In this direction the regression constants between metal- and clay content decreases, indicating a gradient in contamination. Highest concentrations, corrected for clay content have been found in the Land van Saeftinge. Locally (*e.g.* Sloe-area) higher concentrations of certain metals were found due to local sources. Mean ranges of some heavy metal contents in the topsoil of the distinguished morphological units are shown in Table 1.

Uptake of metal-ions from the soil solution and from the overlying water during flooding may cause an accumulation of metals in the salt marsh vegetation. Consequently, significant higher concentrations have been found in the leaf tissues of the marshes at Saeftinge than in the leaf tissues of the less contaminated marshes in the Oosterschelde (BEEFTINK and NIEUWENHUIZE, 1986).

A historic record of metal-contamination may be obtained by determining carefully the concentration of heavy metals in the vertical soil profile. However, diffusion, initiated by spatial differences in oxidation - reduction potentials, and mass transport by organisms and dissolved organic matter may disturb such profiles.

The sorption of heavy metals in the soils is highly influenced by soil characteristics like textural composition, organic matter content, redox potential, pH and salinity. These characteristics also determine the concentration of dissolved metals in the pore water and therefore the bio-availability. To estimate the relative bonding strength of metals sequential extraction procedures have been applied (TESSIER, 1979).

Since the selectivity of the various extractants for a specific phase varies for different metals but also for different soiltypes, the results of sequential extraction procedures cannot be interpreted as distinct geochemically defined phases but rather as method and soil defined fractions. The results of a sequential extraction, according to the method of CALMANO and FÖRSTNER (1983), of zinc in a salt marsh soil profile are shown in Fig. 5. Both, the high metal content and the relatively small residual fraction indicate metal contamination in the upper part of the soil profile. The relative bonding strength of zinc differs significantly between the brownish upper horizon and the sulfidic dark-grey horizons below. In the reducing, sulfidic horizons (20 - 60 cm) zinc is predominantly bound by sulfides. In the suboxic brown coloured upper horizon more mobile (exchangeable) forms occur. This change is not caused by differences in grain-size distribution because the clay content is relatively constant in the upper 50 cm. Thus, the change in the depth distribution represents true differences in bonding strength probably caused by difference in redox potential. Recent experiments, using leaching procedures support this conclusion (STENEKER *et al.*, 1987).

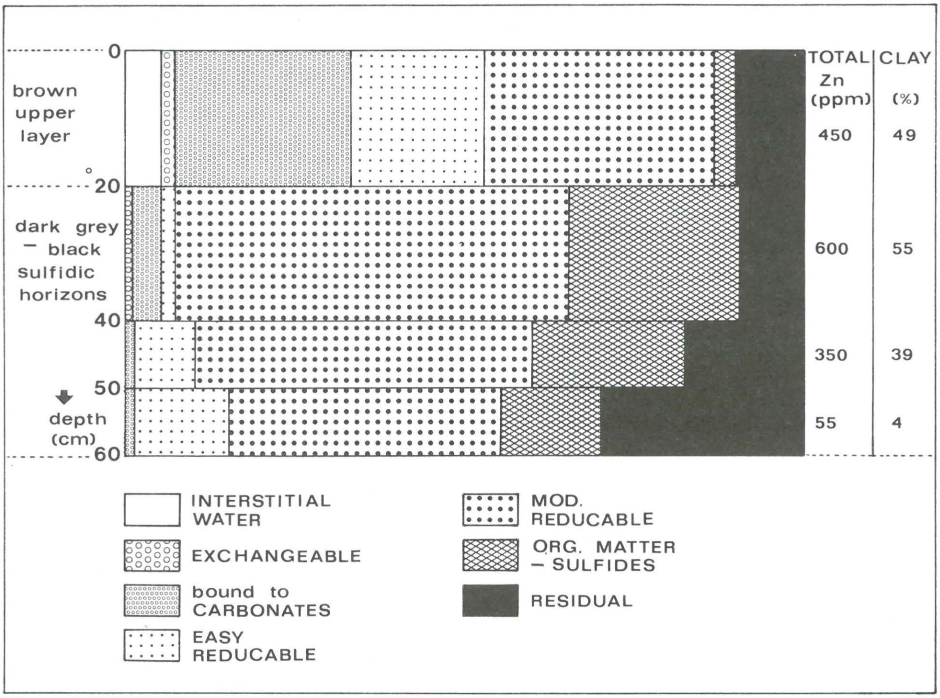

Fig. 5. Sequential extraction of Zn in a salt marsh soil profile at Saeftinge.

CONCLUSIONS

In the preceding chapters a generalized picture has been presented of the soil environment of the intertidal area in the Westerschelde. The soil is an essential compartment of the estuarine ecosystem and has some clearly defined functions. Surface morphology, and the spatial variation of soil-types and grain-size distribution all contribute to the diversity of the estuarine ecosystem.

Until recently, the soil in the Westerschelde has not been explored in great detail. The spatial variation in soil-types and grain-size distribution is rather well known, but the elemental composition, including the depth distribution is not well documented. Notably lacking are data on rates (sedimentation and organic matter decomposition) and on the fate and behaviour of several mobilized compounds in the soil. Therefore the importance of the regulation function of the soil is largely unknown.

Until now, little can also be said about the prediction of eco-toxicological effects from soil analysis. Thus far the data indicate a different sorption of certain metal-ions in highly reducing sulfidic horizons and in (sub-)oxic horizons. Especially in salt marshes seasonal variations will be expected in the behaviour and bio-availability of these metals in the soil.

The source-sink function of the soil is clearly expressed by the large amounts of mud that reside in marshes and mudflats. Moreover, huge amounts of sand have been withdrawn from the lower estuary by dredging in recent years. The data also show storage of excess heavy metals.

A rough estimation of the soil quality with respect to its carrying function can be obtained from the interpretation of the surface morphology and the grain-size distribution, including its depth distribution. Especially physical characteristics of the soil determine this habitat function.

Clearly, more effort is needed to understand the soil environment of the intertidal area in the Westerschelde in full detail.

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BIOTA AND ABIOTIC ENVIRONMENT IN THE WESTERSCHELDE ESTUARY *

CARLO HEIP

KEYWORDS : estuaries; tidal mixing; suspended matter; sedimentation; nitrogen; plankton; benthos.

ABSTRACT

An estuary such as the Westerschelde is a highly dynamic environment, both on an ecological time scale where climatic and hydrodynamic forces, mainly the tides, shape a very variable environment and on a geological, evolutionary time scale, since estuaries are young and very unstable habitats.

Low species diversity and high adaptability of the resident animal and plant populations are characteristic of estuarine habitats where large fluctuations in submersion, salinity, temperature etc. occur. The existing biota are therefore resilient to environmental stress and effects of the important influx of anorganic and organic pollutants from the river Schelde and its tributaries on the biota in the estuary are not easy to detect.

Although water movement and sedimentation patterns in the Westerschelde are relatively well known, there exists little information on important ecological processes such as primary production and heterotrophic metabolism in the estuary.

INTRODUCTION

Estuaries are zones where the sea and the rivers meet and they are characterized or defined by this mixing of water types. A well-known definition is that by PRITCHARD (1967) : 'An estuary is a semi-enclosed coastal body of water, which has a free connection with the open sea, and within which sea water is measurably diluted with fresh water derived from land drainage'.

Water movement is the all important key factor that governs estuarine processes. In the Westerschelde, the impact of the tides is dominant. It is one of the longest tidal rivers in Europe and contains vast intertidal areas where large amounts of mud and organic matter are deposited.

The tidal forces are also at the origin of the estuary. About 2000 years ago the coast north of Cap Griz Nez was a wadden sea bordered with many islands separated by channels through which the tides could enter, creating gullies in the wadden sea. When such a tidal channel caught the basin of a major river, the tides could penetrate much deeper inland and the estuary was created and maintained. This process, together with the rising sea level over the past few thousand years, has created a very dynamic environment that has only very recently been controlled by man.

THE WESTERSCHELDE ESTUARY

PETERS and STIRLING (1976) divide the Schelde in three zones. In the maritime zone, about 70 km long including the part submerged in the North Sea, complex sediment transport occurs through a system of flood and ebb channels in constant evolution. Because formerly important sea arms have been closed naturally or artificially, deep and large channels have developed separated by large sand banks. Mixing of the water is intense and in most circumstances there is no stratification of salinity or current. The central zone of the estuary is about 50 km long. In this zone mixing between fresh and sea water is only partial and more saline waters occur near the bottom. A turbidity maximum and flocculation zone exist close to Antwerp. The fluvial zone of the estuary is still characterized by a rapid and large penetration of the tides.

Despite this important water movement in and out of the estuary, the actual net displacement of the water is relatively small. The total volume of the estuary between the mouth and the Dutch-Belgian border is 2.5 billion cubic meters. From the Schelde there is a water flow of 105 m3/sec on average, about 9 million m3 per day. This indicates that the residence time of the water in the estuarine zone is about 75 days or 150 tides, a long time compared to the daily displacement of water with the tides.

SEDIMENTATION

Fine sedimentary deposits are a highly characteristic feature of estuaries and they are derived both from the sea and from land. Their deposition is controlled by current speed and the smaller particles will only settle at slack water at high tide over the intertidal areas. These particles are unable to settle within one tidal cycle, but the speed of settlement may be increased by flocculation of clay particles in salt water forming larger particles that sink faster. The middle reaches of an estuary are therefore characterized by a turbidity maximum (POSTMA, 1967) where the inward flow of sea water along the bottom stops and the sediments rise to mix with the surface fresh water. WOLLAST (1976) has estimated that from a total of 1.52 million tons of material of continental origin 1.2 million tons are accumulating in the landward zone; in the seaward zone a further 200.000 tons are deposited so that only 120.000 tons reach the sea. However in this zone a further 800.000 tons of marine sediments are deposited.

To the suspended matter that sedimentates heavy metals and organic matter is absorbed. Sedimentation inside the estuary thus limits the output to the sea but has important consequences for the estuary itself since these substances may be incorporated into the food web; resuspension in periods of high freshwater flow may occur and dredging of this material presents another possible hazard.

THE CHEMICAL ENVIRONMENT

Physical and chemical characteristics of the water are modified by the mixing of fresh and sea water. These modifications entrain a series of transformations of the chemical species present in the water which affects their distribution. These chemical changes are enhanced because of the long residence times in the estuary. The heavy load of organic matter in the Schelde catalyses an intense heterotrophic bacterial activity which rapidly exhausts the oxygen in the river. Other oxydants are used for anaerobic respiration : manganese, nitrate, iron and sulphate. When all electron acceptors are exhausted, sulphate reduction thus may become the dominant metabolic process in the river. When the organic matter content of the water decreases due to flocculation a strong decrease in bacterial activity occurs. The process is equivalent (BILLEN *et al.*, 1976) to two successive titrations of Schelde waters : the first of the oxydants by organic matter catalyzed by the heterotrophic activity in the water and the second of the reductants by oxygen catalyzed by reaeration and autotrophic reactions with an increase in the redox potential.

THE BIOTA

We have only rough estimates of primary production in the water column in the Westerschelde. On the basis of monthly samples through 1983, BILLEN *et al.* (1985b) estimate a total annual production of 225 g $C.m^{-2}$ in the river upstream Antwerpen and only 85 g $C.m^{-2}$ in the mixing area around the Dutch-Belgian border. Two other, potentially important, sources of organic matter have not been studied yet : the production by higher plants in the salt marshes along the river banks and the production by benthic autotrophs, mainly diatoms, in the intertidal areas.

Nitrogen in the Schelde is probably never limiting. BILLEN *et al.* (1985a) estimate that about 82.000 tons of nitrogen enter the river each year but due to denitrification about 55.000 tons disappear as nitrogen gas to the atmosphere. The total output of the Westerschelde to the North Sea would be around 27.000 tons of nitrogen per year.

Most of the organic matter produced *in situ* or imported from upstream is deposited in the sediments and fuels benthic metabolism. Estuarine benthos is generally well known and predictable, poor in species but with a high biomass. It is often used for purposes of ecological monitoring but there has been no recent thorough survey of the benthos in the river Schelde, though some occasional observations indicate that benthic fauna is absent or extremely scarce even in intertidal areas (MOERLAND, 1987; HUMMEL *et al.*, this volume). On the other hand, the fauna in the estuary downstream the Dutch-Belgian border is similar to the fauna in other estuaries, with some possible exceptions such as the scarcity of Harpacticoid copepods in sediments (VAN DAMME *et al.*, 1984) and the recent disappearance of the polychaete *Scoloplos armiger* (HEIP *et al.*, 1986).

Because of the long residence time and rapid mixing the dilution of seawater is gradual and stable. Salinity zones in the estuary are relatively stable and are maintained in more or less the same position throughout a tidal cycle. Over the year there may be larger changes according to the amount of the freshwater flow. This is one of the important factors why an autochtonous plankton can persist in the estuary, *e.g.* the dominant diatom *Coscinodiscus commutatus* and the copepod *Eurytemora affinis* (DE PAUW, 1975). Nevertheless, one of the evident characteristics of the estuarine flora and fauna is its low diversity. Only a few species are found and they are characteristic for this environment. It has long been thought that this is due to the physiological problems an organism has to face in an environment with a changing salinity. However, in stable and long existing environments, a diverse and rich brackish water fauna has developed and it appears that the low diversity of the fauna in tidal estuaries is mainly due to their unstable geological history. There are some considerable scientific advantages in the study of this system since many of the dominant species have been well studied and a lot of information is available on the physiology and ecology of many estuarine animals and plants. Since they are not numerous and since their distribution is quite predictable, physical forces may be important (BAKKER and DE PAUW, 1974; 1975) and biological interactions are relatively unimportant or easily studied.

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THE CONCOMITANT EXISTENCE OF A TYPICAL COASTAL AND A DETRITUS FOOD CHAIN IN THE WESTERSCHELDE ESTUARY *

HERMAN HUMMEL, GERARD MOERLAND and CEES BAKKER

KEYWORDS : Westerschelde; estuarine food chains; detritus; brackish tidal zone.

ABSTRACT

Until now studies of the aquatic ecology of the Westerschelde estuary have been erratic. Yet, from the results of a literature research some structural aspects of the ecosystems in the Westerschelde could be derived.

Clear gradients could be found from the river to the sea for the abiotic as well as for the biotic features of the estuary. On the basis of the available quantitative data 2 tentative food chains were distinguished; a mainly detritus-based food chain in the upstream brackish part and a coastal food chain in the downstream seaward part. The centre of the area of the detritus food chain coincided with the zone of a turbidity maximum at the interface of salt and fresh water. This food chain is characterized by a low primary production but a year-round high concentration of suspended organic matter. The suspended organic matter consists of aggregates of detritus and bacteria. The zooplankton as well as the zoobenthos can reach high biomasses. The coastal food chain is mainly based on a seasonally changing high primary production. The diversity of flora and fauna in this system is much higher than in the detritus food chain, although the biomass can be lower.

INTRODUCTION

The Westerschelde estuary (Fig. 1) is the last estuary remaining in the Delta area in the south-western part of The Netherlands. As a result of extensive projects to protect the mainland from storm-surges several estuaries at the mouths of the rivers Rhine, Meuse and Schelde, the so-called Delta area, were enclosed.

In the estuary the abiotic and biological features are largely determined by the mixing of fresh water from France and Belgium at the one side and sea water from the North Sea at the other. Moreover, a substantial influence can be expected from the discharge of industrial and domestic wastes. However, a comprehensive view of these influences, as well as of the functioning of the Westerschelde ecosystem is not available. Most studies until now were scattered and not integrated in a model. Yet, from the scanty literature some main trends for the ecosystem could be abstracted.

The aim of this study is to review some relevant chemical and biological features of the Westerschelde estuary. Moreover, a preliminary model for the Westerschelde ecosystem, abstracted from quantitative data, is presented.

MATERIALS

The area studied (Fig. 1) is the tidal part of the Schelde river and the Westerschelde estuary running from Gent to Vlissingen. With regard to the ecosystem model the study had to be focussed on the trajectory from Antwerpen to Vlissingen, because most essential

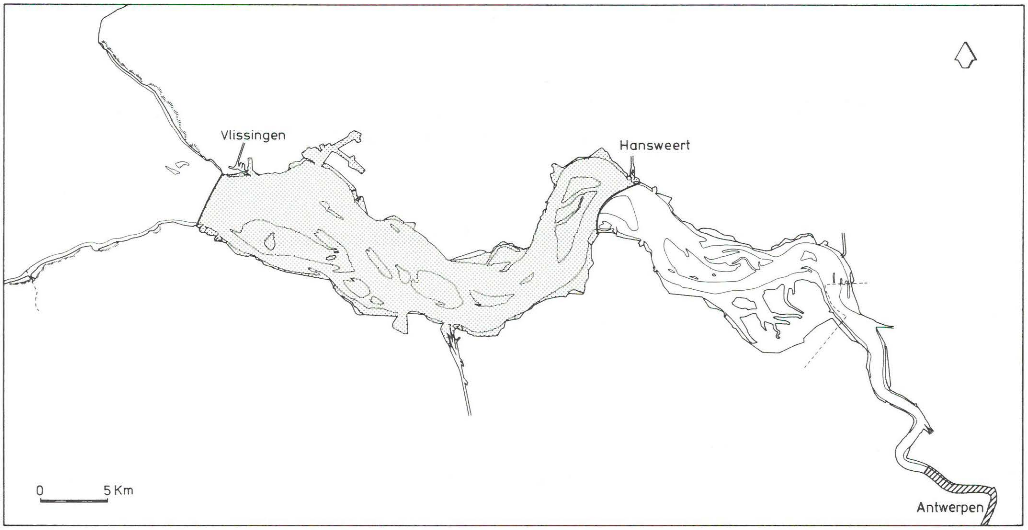

Fig. 1. The Westerschelde estuary, with a subdivision in 3 zones : a freshwater (hatched), a brackish (blank) and marine (dotted) tidal zone.

quantitative data from the fresh water tidal part are lacking. Most data are from the end of the sixties and later. The literature was screened for quantitative data on chlorinity, oxygen, suspended matter, transparency, nutrients (N, P, Si) and for qualitative and quantitative data on organisms.

Data on abiotic parameters were obtained from DE PAUW (1975); those on planktonic organisms from BAKKER and DE PAUW (1974, 1975), DE PAUW (1975), BAKKER *et al.* (1977) and JACKSON and DE VISSCHER (1985). Data on organic matter were abstracted from WOLLAST (1976) and VAN DAMME *et al.* (1984); on bacteria from PEELEN (pers. comm.). Data on benthic organisms were derived from VAN DAMME *et al.* (1980), VERMEULEN (1980), VAN IMPE (1985) and HEIP *et al.* (1986).

The average depth was taken as 7 meters for the trajectory from Antwerpen to Hansweert and 12 meters for Hansweert to Vlissingen (according to Rijkswaterstaat, Service Tidal Waters). Values on biomass, expressed in g.C.m^{-2}, were calculated taking a value of 0.4 to convert ash-free dry weight to organic carbon. Based on the chlorophyll-*a* measurements using a fluorometer (JACKSON and DE VISSCHER, 1985) a value of 30 was chosen to convert chlorophyll to organic carbon.

RESULTS AND DISCUSSION

Abiotic and biotic gradients

1. Abiotic gradients.

The chlorinity of the water in the Westerschelde estuary increased downstream, from Gent to Vlissingen (Fig. 2a), as a consequence of the increasing mixing of sea water with fresh water. Similarly, the oxygen concentration increased from around 20 % between Gent and Antwerpen to saturation levels near Vlissingen (Fig. 2a). The low oxygen concentrations upstream are caused by the heavy load of suspended matter (Fig. 2b) of which the organic part is mineralized (DE PAUW, 1975). As a result of the mineralization high concentrations of nutrients (N, P, Si; Fig. 2c) were found upstream

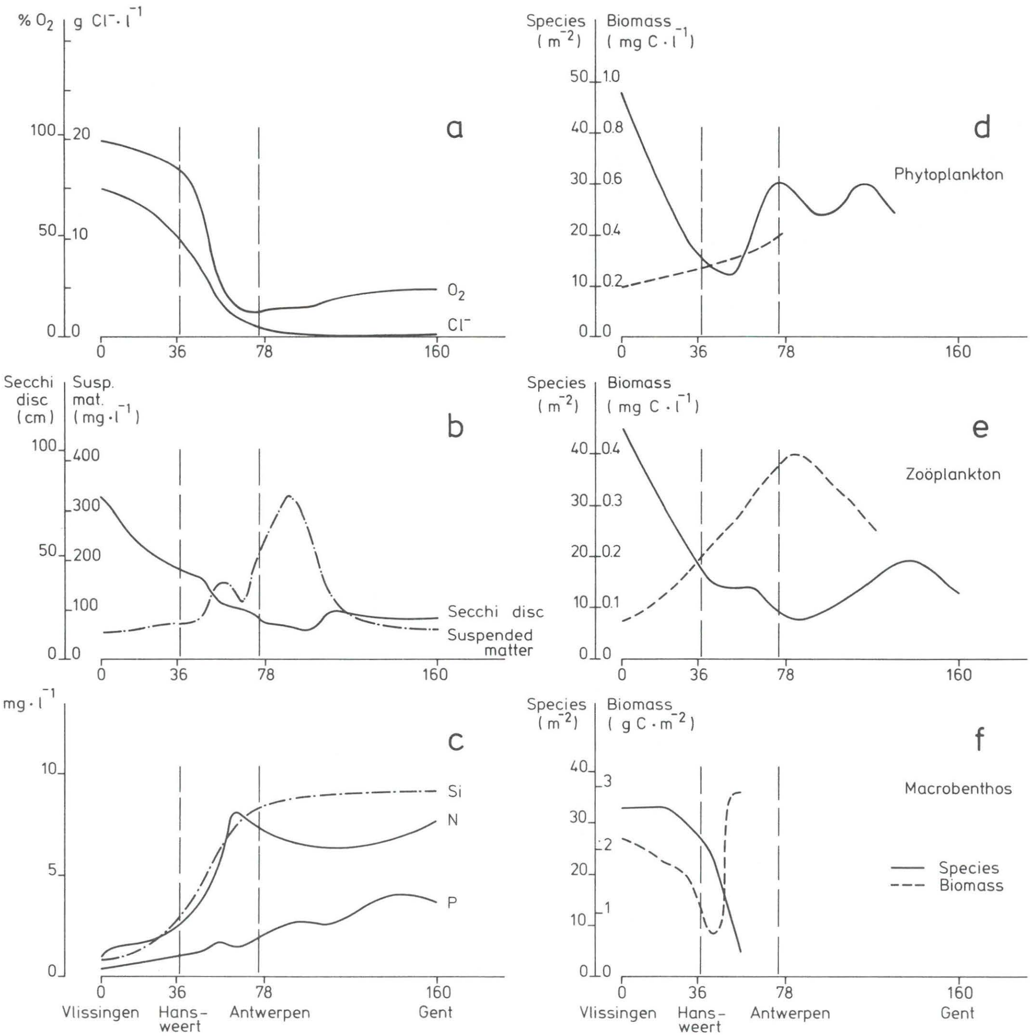

Fig. 2. Gradients for a) chlorinity, oxygen, b) suspended matter, secchi disc visibility, c) nutrients, and for number of species and biomass of d) phytoplankton, e) zooplankton and f) macrobenthos in the Schelde and Westerschelde estuary from Gent to Vlissingen.

Antwerpen. Moreover, the high load of suspended matter resulted in a low transparency (measured by secchi disc) of the water around and upstream Antwerpen (Fig. 2b). The small turbidity maximum downstream Antwerpen is caused by the flocculation of organic matter at the transition zone between fresh water and sea water (WOLLAST, 1976). The high load of suspended matter upstream Antwerpen is of domestic and industrial origin.

2. Biotic gradients.

Gradients on bacteria, birds and fish could not be derived from the literature because data were too scanty. The data on phyto- and zooplankton are spatially and temporarily erratic, but still some trends could be recognized.

The number of phytoplankton-species decreased from Vlissingen to Hansweert (Fig. 2d), rising again in the brackish waters near Antwerpen through the contribution of freshwater species. The increase of the number of zooplankton species (Fig. 2e) was shifted in the upstream direction. These trends are in agreement with the curve of Remane (REMANE and SCHLIEPER, 1971) indicating that a minimum in the number of species is found around 3 $^{o}/_{oo}$ chlorinity. According to this model the number of species should rise again to a maximum in the fresh water area. However, in contrast to Remane's curve the number of species in the freshwater tidal zone of the Westerschelde estuary remained relatively low (Fig. 2d, e, f).

Surprisingly, the decrease in numbers of species are not reflected in the biomass. On the contrary, from Vlissingen to Antwerpen the biomass increased and reached maxima in the brackish area near Antwerpen (Fig. 2d, e, f). This seemingly contradiction will be explained in section 3.3.1.

A freshwater, brackish and marine tidal zone

On the basis of the abiotic and biotic gradients the estuary could be partitioned in 3 clearly different areas : a freshwater (Gent to Antwerpen), a brackish (Antwerpen to Hansweert) and a marine tidal zone (Hansweert to Vlissingen) (Fig. 1, 2).

In the freshwater zone low values for chlorinity, oxygen and transparency, and high values for suspended matter and nutrients are found. An impoverished flora and fauna is found, as a consequence of the high load of suspended matter and subsequent low oxygen concentration (DE PAUW, 1975).

In the very dynamic brackish tidal zone a drastic downstream increase of chlorinity, oxygen and transparency and a strong decline of suspended matter and nutrients is found. Although the species diversity of the flora and fauna may reach minima, the biomass reaches maxima in this zone.

The marine tidal zone is characterized by high chlorinity, oxygen and transparency values, still high suspended matter and low nutrient concentrations. A well-developed flora and fauna is found with a high diversity and intermediate (benthos) or rather low (plankton) biomass.

Because of strong tidal and seasonal changes in the estuary there is a gradual transition of the 3 zones and the boundaries of Fig. 2 indicate only an average position.

A tentative ecosystem model

For the 2 zones best described, the marine and brackish, a tentative model for their ecosystem could be derived from quantitative data (Fig. 3).

1. Detritus food chain.

In the brackish tidal zone the influence of suspended (organic) matter is overwhelming. The tremendous input of organic matter from the freshwater tidal zone and the flocculation of organics in the transition zone of fresh and sea water give rise to high concentrations of organic matter (DE PAUW, 1975; WOLLAST, 1976). About 50 % of the organic matter sinks to the bottom after flocculation, 40 % is remineralized and 10 % exported to the marine tidal zone (WOLLAST, 1976). Thus, the bacterial activity and biomass will be high too in the brackish zone, as is the concentration of nutrients. However, due to the low transparency the primary production remains low, irrespective of the high nutrient concentration (JACKSON and DE VISSCHER, 1985).

The zooplankton consists mainly of the copepod *Eurytemora affinis*

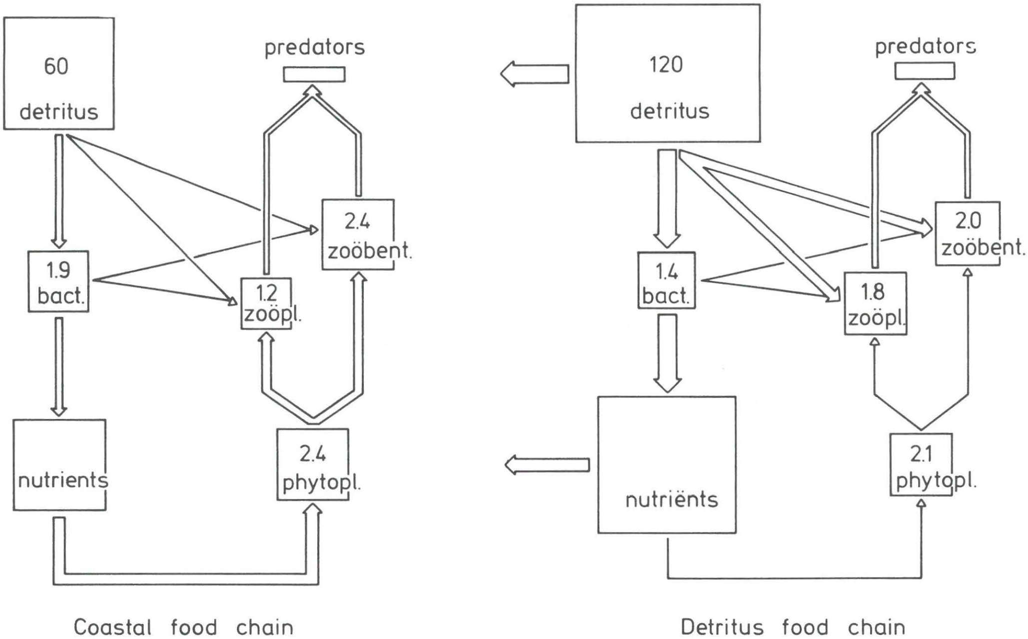

Fig. 3. Tentative ecosystem-models for the brackish and marine zone of the Westerschelde according to resp. a detritus and a coastal food chain. Biomass figures in g.C.m^{-2}.

and ciliates, indicating that the flocculated aggregates of detritus and bacteria may be a main food source in this zone, instead of phytoplankton (BAKKER and DE PAUW, 1974, 1975; DE PAUW, 1975; BAKKER *et al.*, 1977). High concentrations of organic matter are found year-round, and, therefore, a high biomass of zooplankton can be found throughout the year. Thus, the few zooplankton as well as zoobenthos species that can resist the harsh conditions of the brackish environment probably experience a high year-round supply of potential food and, thereby, can reach a high biomass. HEINLE and FLEMER (1975) reported similar phenomena in the Patuxent River Estuary (USA). In the upper part of this estuary the water is highly turbid, as in the Western Scheldt, and no pronounced spring phytoplankton maximum could be demonstrated here as well. Heinle and Flemer showed that gross algal production was 5 - 75 times smaller than the carbon requirements of the local *Eurytemora* population. Detrital carbon on the other hand was always present in large amounts. Thus, this organic detritus probably constitutes an important link between primary and secondary production in the estuarine ecosystem, bacteria being essential for the transfer of energy through this food chain. The constantly large amount of detritus present will have a stabilizing effect by leveling out the results of changing rates of primary production due to seasonal or other variations in external factors (FENCHEL, 1972).

It is concluded that the ecosystem in the brackish tidal zone is primarily based on the presence of allochthonous organic matter and, therefore, the food web is termed a detritus food chain.

2. Coastal food chain.

The influence of suspended (organic) matter is less pronounced in the marine tidal zone. The more important primary production of phytoplankton reaches higher levels because of the higher transparency of the water. As in the temperate zone the primary production is seasonal, the supply of potential food for zooplankton and zoobenthos consequently is seasonally restricted too. Although the diversity of flora and fauna is higher here than in the brackish zone, the biomass is not.

The food web based on the seasonal production of phytoplankton resembles that of coastal temperate areas (RYTHER, 1969; PARSONS *et al.*, 1977), and, therefore, is called a coastal food chain.

EPILOGUE

The impact of detrital organic matter in the food chain of the brackish zone is obvious. Yet, it is not clear to what extent this potential food supply depends on natural flocculation processes or on the import of effluents from the freshwater tidal zone. Studies on the nature and bio-availability of this organic matter were not at our disposal.

More generally, no information was present on the carbon flow through the food chain, *i.e.* no figures were available on production and consumption of organic matter by organisms in the Westerschelde. Moreover, the biomass values given in Fig. 3 were often estimated indirectly. For example, the biomass of phytoplankton is based on chlorophyll-carbon ratio's, including well-known large incertainties (BANSE, 1977).

From this it is clear that future research in the Westerschelde has to pay more attention to the integration of the different aspects of the estuarine ecosystems.

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MODELLING MICROBIAL PROCESSES (PHYTO- AND BACTERIOPLANKTON) IN THE SCHELDE ESTUARY

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KEYWORDS : Schelde; estuary; phytoplankton; bacterioplankton; ecological models.

ABSTRACT

Since about 10 years, studies have been conducted at the University of Brussels in modelling the microbiological processes affecting biogenic substances in the Schelde estuary and its watershed.

The first model, a one dimensional redox model, simulated the longitudinal distribution of oxygen, nitrate, iron and manganese in relation to the observed bacterial heterotrophic activity. A model allowing calculation of bacterial activity from data on organic discharge was coupled to this model. It was completed by a model of phytoplanktonic development within the estuary. Finally, an idealized model of the hydrographical network, based on Horton analysis, is being established in order to calculate the quality of the water at the downward boundary of the estuary.

Together, all these sub-models form a general model of the ecological working of the Schelde estuary, able to predict at least the general trends of the redox state, the organic carbon, mineral nitrogen, and chlorophyll-*a* concentrations as a function of distance to the sea, from the knowledge of geomorphological and meteorological data, along with informations concerning the distribution of anthropogenic discharges. This model therefore provides a powerfull tool for the rational management of the Schelde estuary.

INTRODUCTION

Estuaries are the site of intense chemical and biological processes which strongly affect the transfer to the sea of pollutants from land based sources. This is particularly true for the Schelde, due to the long residence time (about 1 month) of the water masses within this estuary. Budget calculations showed that about 70 - 90 % of the organic carbon load (BILLEN *et al.*, 1985) and 40 % of the nitrogen load (BILLEN *et al.*, 1986) entering the estuarine zone is eliminated by microbiological processes in the water before reaching the North Sea. These processes also deeply affect the water quality of the estuary itself. A rational management of both the North Sea coastal waters and the Schelde estuary therefore requires water quality models taking into account microbial activity.

The present paper is an attempt to summarize the studies conducted since about 10 years by our research group at the University of Brussels, in modelling the microbial processes affecting biogenic substances in the Schelde estuary. We shall try to show how the various contributions published separately since 1976 now constitute *pieces* of *puzzle* which, put together, will soon produce a complete, simplified but realistic, ecological model of the estuary.

THE HYDRODYNAMICAL MODEL

Although the production of the hydrodynamical model is not an original contribution of our group, it is easier to briefly describe at the beginning this model to which all subsequent chemical and microbiological models have been coupled.

Transport and dispersion processes in the estuary result from a residual downward circulation, linked to the freshwater discharge, and from flood and ebb currents linked to the tides. The resultant effect of these complex hydrodynamical processes can be described by a simplified one dimensional stationary model (WOLLAST, 1973). The balance of a conservation substance S at a given place in the estuary is written :

$$\frac{dS(x)}{dt} = - \frac{Q}{A(x)} \frac{\delta S(x)}{\delta x} + \frac{1}{A(x)} \frac{\delta}{\delta x} \left(A(x)\, D_s(x) \frac{\delta S(x)}{\delta x} \right)$$

where x is the distance to the sea
Q is the freshwater discharge
A(x) is the wetted section which obeys the relationship :

$$A(x)\ (m^2) = 151350.10^{-0.02\ x\ (km)}$$

$D_s(x)$ is an apparent mixing coefficient.

By adjustment on experimentally observed salinity profiles, D_s has been shown to obey the following empirical relationship with discharge :

$$D_s(m^2/h) = 10^5 + 4500.Q(m^3/sec) \qquad \text{for } x < 75 \text{ km}$$

$$10^5 + 22500.Q(m^3/sec) \qquad \text{for } x > 80 \text{ km}$$

This simplified representation of the complex hydrodynamics of the Schelde estuary is very crude. It is, however, quite satisfactory for the needs of most microbiological models for which more complex, two or three dimensional hydrodynamical models are not required.

THE REDOX MODEL

Our first attempt in modelling the effect of microbiological activity on water quality of Schelde estuary consisted in a purely thermodynamical model. It relates the redox potential of the water to heterotrophic activity (BILLEN and SMITZ, 1976a, b). The model was intended to predict the longitudinal distribution of redox species (oxygen, nitrate/ammonium dissolved and particulate manganese and iron, sulfate/sulfides) from the knowledge of the hydrodynamics of the estuary, of the boundary conditions and of experimentally determined longitudinal profiles of heterotrophic bacterial activity (Fig. 1). The latter was estimated from dark $^{14}C\text{-}HCO_3$, incorporation measurements according to the method (admittedly now obsolete) of ROMANENKO (1964). It was viewed as an electron flux imposed to the water column and affecting its whole redox balance. The redox state of the water is characterized by a variable F defined as the weighted sum of the concentration of all oxidants involved in microbial respiration :

$$F(meq/l) = 4(O_2) + 2(MnO_2) + 8(NO_3^-) + 1(Fe(OH)_3) + 8(SO_4^{2-})$$

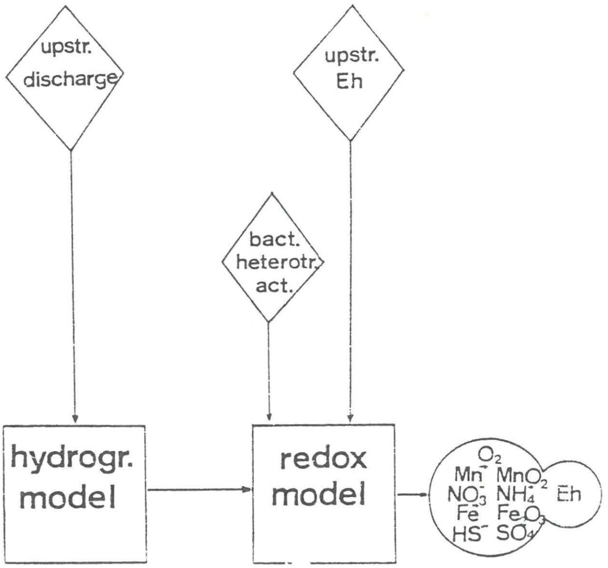

Fig. 1. Diagrammatic representation of the redox model of the Schelde estuary (BILLEN and SMITZ, 1976). State variables are represented by circles, input variables by rhombus. The model itself (*i.e.* the whole list of hypothesis translated into mathematical equations relating the state variables and the input variables) is represented as a square.

This variable is subject to hydrodynamical transport processes, to consumption by organotrophic activity and to supply by reaeration. Once F has been determined, the concentration of each individual redox species (Red_i/Ox_i) is calculated by assuming an internal thermodynamical equilibrium, represented by the relationship

$$Eh = E^o_i + \frac{RT}{nF} \ln \frac{Ox_i}{Red_i} \quad \text{for all (i)}$$

The link between F value and redox potential (Eh) is represented by a titration curve similar to that shown in Fig. 2 for the case of Rupelmonde. It depends on the total

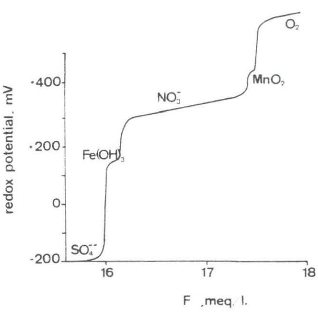

Fig. 2. Theoretical relationship between F function (weighted sum of oxidants) and redox potential for the composition of Schelde water at Rupelmonde (Redox titration curve).

concentration of each redox couple involved, which must be calculated at each point of the longitudinal profile. For MnO_2/Mn^{++}, $Fe(OH)_3/Fe^{++}$ and SO_4^{--}/HS^-, the total concentration is assumed to be conservative (possible losses by sedimentation of solid species are neglected). For mineral nitrogen, a complete balance is established at each point, taking into account : input of ammonium nitrogen by water discharges, mineralization of organic nitrogen linked to organotrophic activity, denitrification representing the total of organotrophic activity when nitrates are the best oxidants present in the water. Owing to the slow development of nitrifying bacteria, the hypothesis of internal equilibrium can lead to overestimation of the rate of nitrification. An empirical limit of the rate of nitrification is therefore introduced within the model. This limit is set to 1 $\mu M.h^{-1}$ on basis of experimental observations (SOMVILLE, 1978).

Fig. 3 shows two examples of the simulations obtained by this model and their comparison with observed data.

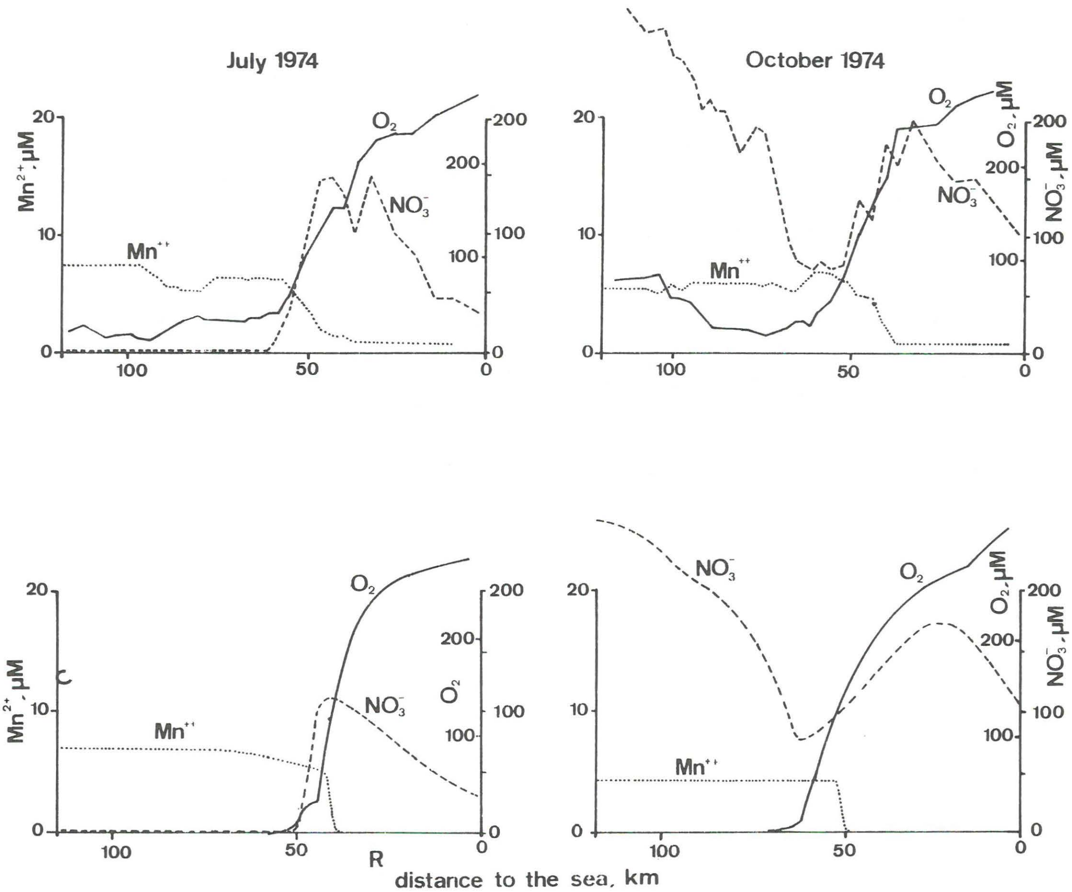

Fig. 3. Observed and calculated longitudinal profiles of redox species in the Schelde estuary in July and October 1974. The calculated profiles are the result of the BILLEN and SMITZ (1986) redox model with observed bacterial activity measurements as input data.

Given an empirical relationship between temperature and organotrophic activity, the same model has been used for simulating the effect of the thermal discharges of the nuclear power plant at Doel on the water quality of the Schelde estuary (BILLEN and SMITZ, 1976b).

ORGANIC MATTER DEGRADATION MODEL

One serious limitation to the predictive capacity of the redox model described above was that it required experimentally determined data on bacterial activity as an input to the simulation. This, of course, did not allow to use the model for simulating the effect of a modification of the organic load discharged into the estuary. This prompted us to develop a model of organic matter degradation, linking organic load to bacterial organotrophic activity (BILLEN *et al.*, 1983).

Most existing models of organic matter degradation basically derive from the STREETER and PHELPS (1925) model, in which the rate of organic matter degradation is assumed to be first order with respect to the organic load. This simplified approach is unsuitable for simulating the complexity of the microbiological processes occurring in the Schelde estuary, and the model we developed was based on a more realistic view of the process of organic matter degradation by bacteria.

Organic matter in natural water is mostly supplied under the form of macromolecular biopolymers which have to be hydrolyzed before they can be taken up by microorganisms. This exoenzymatic hydrolysis is thought to represent the limiting step in the microbial utilization of organic matter. It obeys a Michaelis-Menten kinetics (SOMVILLE and BILLEN, 1983). The concentration of exoenzymes is directly proportional to bacterial biomass (FONTIGNY *et al.*, 1987). It is customary to consider that organic matter in natural waters is present under at least two classes with different biodegradabilities (WESTRICH, 1983; GARBER, 1984) *i.e.* with different susceptibilities to exoenzymatic hydrolysis.

The process of microbial organic matter degradation is thus represented by the following system of equations :

$$\frac{dH_1}{dt} = p_1 - e_1max \frac{H_1}{H_1 + K_1H} . B$$

$$\frac{dH_2}{dt} = p_2 - e_2max \frac{H_2}{H_2 + K_2H} . B$$

$$\frac{dB}{dt} = Y \left(e_1max \frac{H_1}{H_1 + K_1H} + e_2max \frac{H_2}{H_2 + K_2H}\right) B - kdB$$

where for each class i of organic matter :

H_i	is the concentration ($mgC.l^{-1}$)
p_i	is the rate of supply ($mgC.l^{-1}.h^{-1}$)
e_imax	is the maximum rate of exoenzymatic hydrolysis (h^{-1})
K_iH	is the half-saturation constant of hydrolysis ($mgC.l^{-1}$)
B	is the bacterial biomass ($mgC.l^{-1}$)
Y	is the growth yield constant, *i.e.* the ratio of bacterial biomass formed to the total organic matter metabolized and is generally close to 0.3 in nitrogen-non-limited environments (LANCELOT and BILLEN, 1985; SERVAIS, 1986).
kd	is the first order mortality constant of bacteria, which is close to 0.01 h^{-1} (SERVAIS *et al.*, 1985).

The kinetic parameters of biopolymers hydrolysis were determined by adjustment on the results of a batch experiment, in which the bacterial utilization of the organic matter from Schelde water was followed for 30 days (BILLEN *et al.*, 1983). The following values of the parameters at 20 °C were obtained :

$$e_1max = 0.4\ h^{-1}$$
$$K_1H = 1\ mgC/l$$
$$e_2max = 0.06\ h^{-1}$$
$$K_2H = 13\ mgC/l.$$

The dependence of e_1max on temperature was supposed to be characterized by a Q_{10} of 2 as for most biological processes. Water temperature varies seasonally according to :

$$T = 15 - 9 \cos \frac{2\pi}{365} (t - 30)$$

The results obtained by this model are shown in Fig. 4 : the model is able to predict the major trends of the longitudinal distribution of the rate of bacterial organotrophic activity as

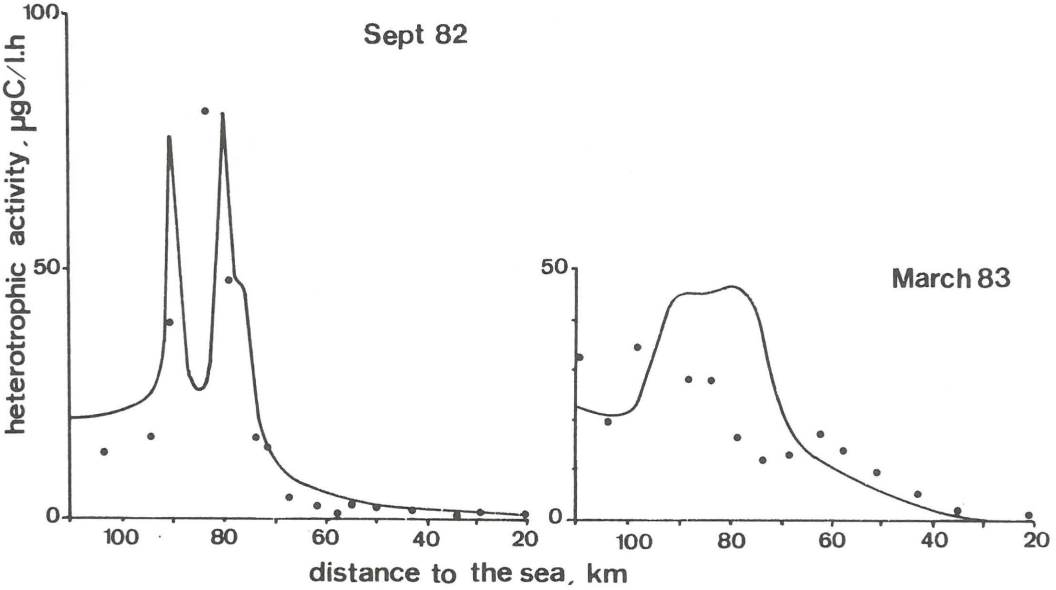

Fig. 4. Observed and calculated longitudinal profiles of bacterial organotrophic activity in the Schelde estuary in September 1982 and March 1983. The calculated profiles are the result of the organic matter degradation model of BILLEN *et al.*, 1983, with discharge of organic carbon as input data.

well as of organic matter concentration. Coupled with the redox model described above, it allows to predict the quality of the water for given inputs of organic matter into the estuary (Fig. 5).

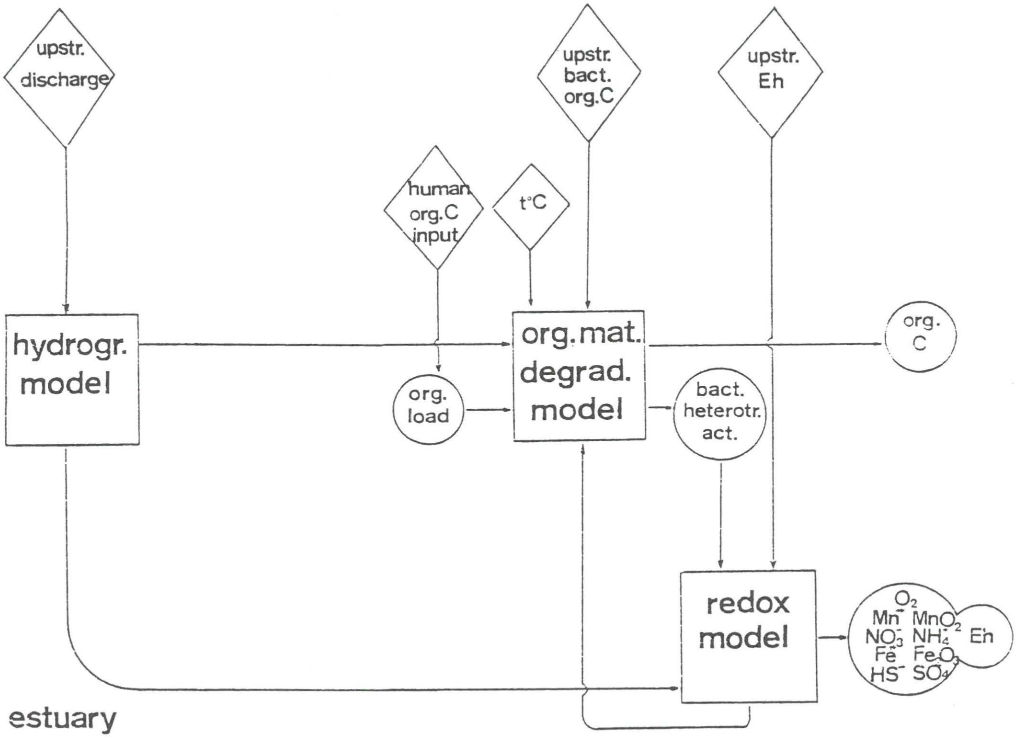

Fig. 5. Diagrammatic representation of the coupling between the redox model (Fig. 1) and the organic matter degradation model of BILLEN *et al.*, 1983.

PHYTOPLANKTON MODEL

When running the previously described model of organic matter degradation, a difficulty lays in the unaccuracy of data concerning the organic load to the Schelde estuary. This is caused by the lack of good inventories of domestic and industrial discharges (together about 60 TC/day) and by the fact that phytoplankton contributes for a significant part of this organic load in summer (primary production up to 100 TC/day). A model of phytoplankton development in the Schelde estuary should therefore be a valuable improvement to the organic degradation model. Lancelot developed such a model (BILLEN *et al.*, 1985), according to the following principles : two distinct phytoplanktonic populations, adapted respectively to fresh- and seawater, and present alone one at the upstream and the other at the downstream limit of the model, are considered to mix, develop and die within the estuary. The dependance of their growth and mortality on light, temperature, nutrients and salinity was empirically determined. The form of these relationships and the values of the corresponding parameters are summarized in Table 1. The control by nutrient concentrations must be included for the sake of completeness even though no nutrient limitation occurs in the present state of the Schelde estuary, where high turbidity is the major limiting factor of phytoplanktonic growth. Drastic reduction of the suspended sediment load and of the nutrient load could possibly result in a shift from light to nutrient limitation.

Incident available light intensities and photoperiod were calculated from integrated measurements of total incident solar radiation collected every 30 min. at Uccle by the *Institut Royal Météorologique de Belgique.*

For setting the value of the extinction coefficient η, a complete model of the suspended matter in the estuary should be necessary. Instead, empirical relationships between suspended matter (SM) deduced from the observation of a large number of longitudinal profiles, were used :

Table 1. Environmental control of phytoplankton growth and mortality in the Schelde estuary (from BILLEN *et al.*, 1985).

Freshwater population	Sea water population

GROWTH RATE

$$\mu = \mu_{max} \frac{1}{24} \int_o^{\lambda} dt \, \frac{1}{h} \int_o^{h} dz \, \frac{(I/I_k)}{(1 + (I/I_k)^2)^{1/2}}$$ (VOLLENWEIDER, 1965)

with $\mu_{max} = \mu'_{max} \, N / (N+K_N)$

and $I = I(o) \, e^{-\eta . z}$

where
- $I(o)$ is the incident photosynthetically available light intensity (range 400 - 700 nm)
- λ is the photoperiod
- h is the depth
- η is the light attenuation coefficient
- μ_{max} and I_k are physiological parameters which characterize the photosynthesis-light relationship :
- N is the concentration of a possibly limiting nutrient
- K_N is the half saturation constant for the uptake of the limiting nutrient (about 5 μM for mineral nitrogen).

$$\mu_{max}\,(\text{mgC/mgChla.h}) = 8\,(2^{\frac{T-20}{10}})\,.\,(e^{-(\frac{S}{10})^2}) \qquad = 17.6\,(2^{\frac{T-20}{10}})\,.\,(e^{-(\frac{S-17}{5})^2})$$

$I_k(J/cm^2.h) = 0.2\ Phar/\eta$ for $Phar/\eta < 100\ J/cm^2.day.m^{-1}$

20 for $Phar/\eta > 100$

where Phar is the daily integrated photosynthetic active radiance.

MORTALITY RATE

k_d $0.005\ h^{-1}$ for $S < 5$ gCl$^-$/l $= 0.005$ for $S > 13$ gCl/l

0.015 $>$ $= 0.015$ $<$

SM (mg/l) = 4000. $1/Q(m^3/sec)$ in the Antwerp zone.
SM (mg/l) = 60 for $Q \leq 50\ m^3/sec$ in the mixing zone.
= $0.6\ Q(m^3/sec) + 30$ for $Q > 50\ m^3/sec$
S_m (mg/l) = 60 downstreams.

Fig. 6 and 7 show the result of phytoplankton biomass and production calculation by

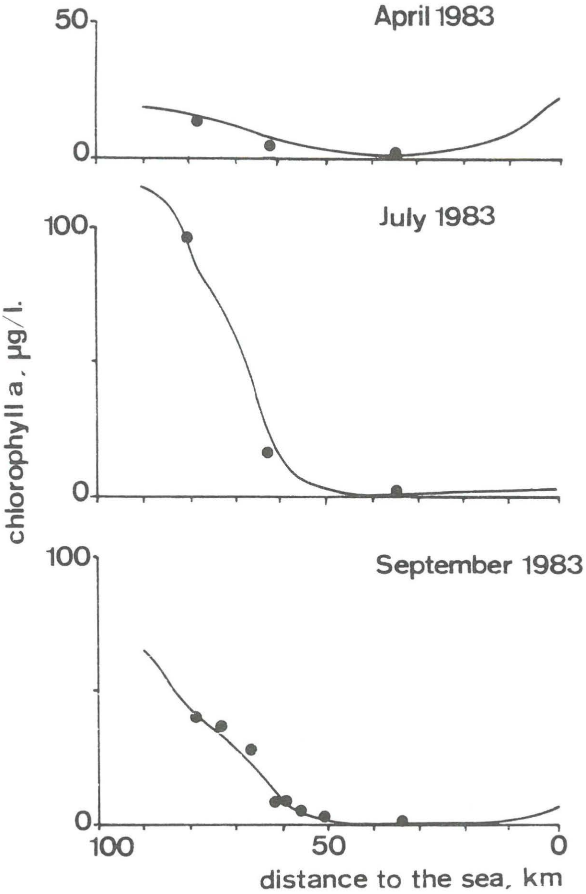

Fig. 6. Observed calculated longitudinal profiles of phytoplankton biomass in the Schelde estuary in 1983. Calculated profiles are the result of Lancelot's phytoplankton model using meteorological data as input variable.

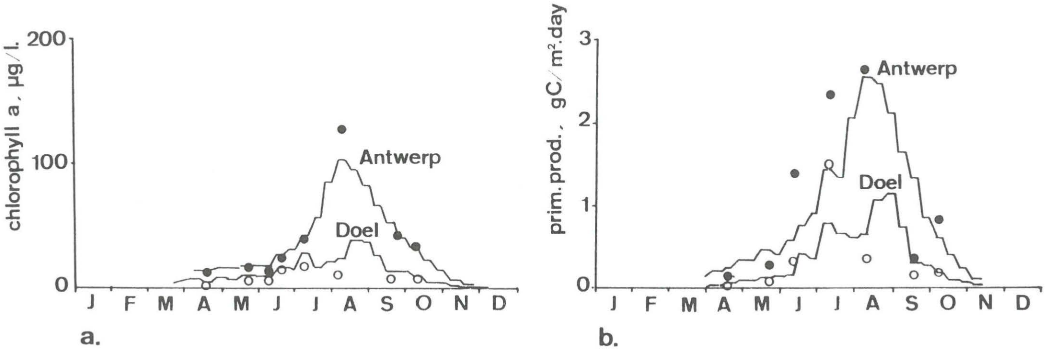

Fig. 7. Observed and calculated seasonal variations of phytoplankton biomass (a) and primary production (b) at Antwerp (o) and Doel (o), in 1983.

this model. Comparison with experimental observations shows that the simulation reproduces the major trends and levels of both the spatial and temporal phytoplankton variations within the estuary. As represented in Fig. 8, this model can be easily coupled with the organic matter degradation/redox model, thus providing a complete coverage of the major microbiological processes occurring in the Schelde estuary.

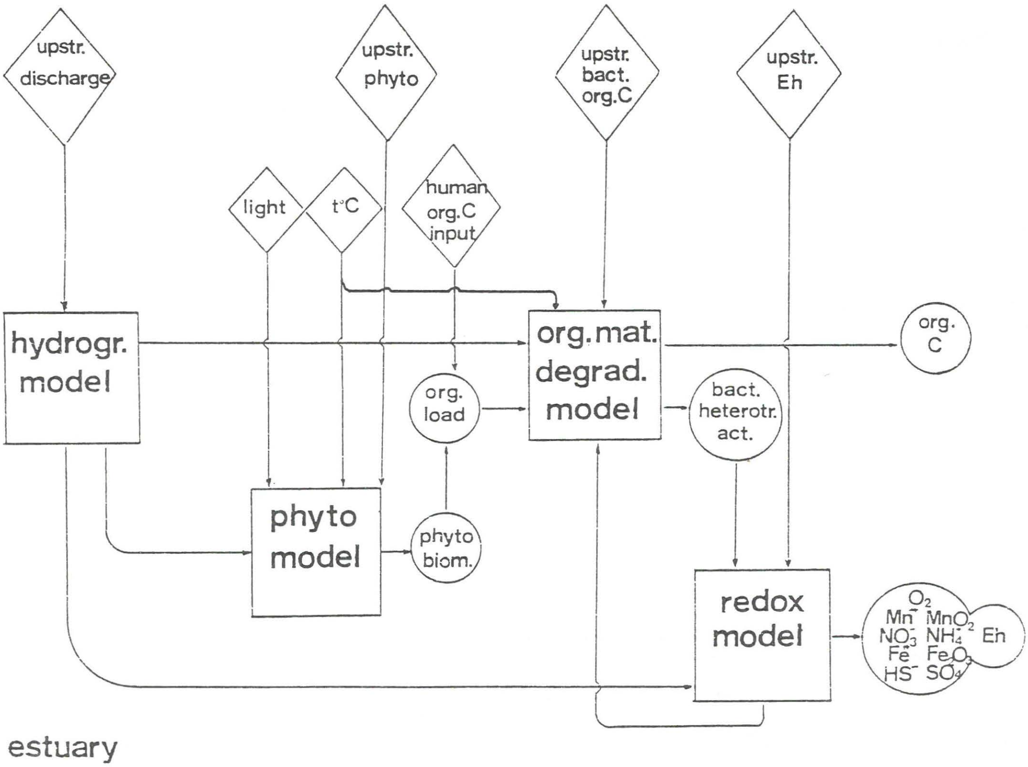

Fig. 8. Diagrammatic representation of the coupling between the organic matter degradation/ redox model (Fig. 5) and Lancelot's phytoplankton model.

THE UPSTREAM HYDROGRAPHICAL NETWORK MODEL

The model represented in Fig. 8 still suffers from one major handicap : it requires, as boundary condition, the values at the upstream boundary of the main variables (redox state, organic load, bacterial biomass, phytoplankton concentration) and their seasonal variations. This implies that observed data at Rupelmonde have to be injected in the model for calculation of the scenarios. As the answer of the model is very sensitive to this boundary condition, the predictive capabilities of the model are severely limited by this requirement. Moreover, the model is unable to evaluate the impact on the estuarine zone of new perturbations or sanitation measures occurring in the upstream river system. For this reason, we are now developing a simplified model of the whole Schelde hydrographical network, intended to calculate phytoplankton and bacterial biomass, organic carbon concentration and redox state of the water at the entry of the estuarine zone.

The basic principle of this model (BILLEN *et al.*, 1987) is to couple microbiological models similar to those discussed above for the case of the estuarine zone, to a general hydrological model based on Horton's geomorphological analysis of the watershed. This analysis (LEOPOLD *et al.*, 1964) consists in attributing an order to the various tributaries and in deducing general relationships between these orders on the one hand, and the number of

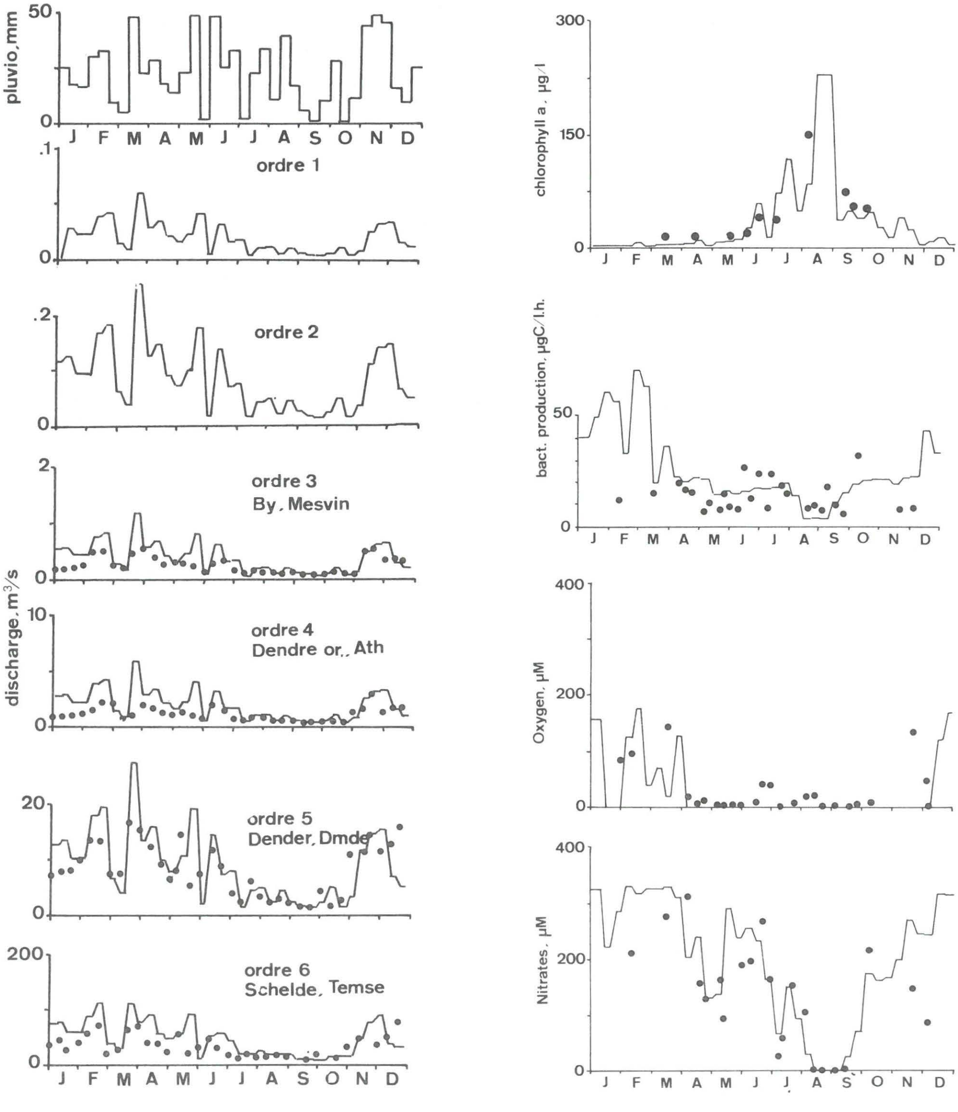

Fig. 9. Some examples of results of the idealized model of the Schelde hydrographical network, based on Horton analysis (BILLEN *et al.*, 1988). a. Observed and calculated seasonal variations of discharge for tributaries of the different orders in the Schelde watershed in 1977.
b. Observed and calculated seasonal variations of chlorophyll-*a* concentration in the river Schelde at the entry of the estuarine zone in 1983. c. Observed and calculated seasonal variations of bacterial activity (production) in the river Rupel at the entry of the estuarine zone of the Schelde in 1987. d. Observed and calculated seasonal variations of oxygen and nitrate concentrations in the river Rupel at the entry of the estuarine zone of the Schelde in 1987.

tributaries and their geometry (length, width, watershed area, slope) on the other hand. From these characteristics and from pluviosity data, it is possible to calculate the seasonal variations of discharge, rate of flow and mean depth in each order of tributaries. An example is shown in Fig. 9a. These hydrological data are then used to calculate, for increasing order of tributaries, activity and biomass of phytoplankton, organotrophic activity, bacterial biomass, organic load and concentrations of nutrients and oxidants. A few examples are shown in Fig. 9b, c, d. The values of these variables for the Schelde at Rupelmonde can then be used as the upstream boundary condition for the phytoplankton/organic matter degradation/redox model described above. (Fig. 10).

The coupling of these different sub-models thus results in a general model of the ecological working of the Schelde system. The only information required as input data are geomorphological and meteorological data, along with informations concerning the distribution of anthropogenic organic discharges. The model will therefore be quite convenient for answering questions put by the concern of rational management of this important ecosystem : what would be the impact of a waste water purification policy in the Schelde watershed (with or without tertiary treatment) on the water quality in the Schelde estuary and the coastal North Sea ? What should be the optimal target in this purification policy for reaching a given objective in term of water quality ?

Questions of this nature are not at all trivial to answer, due to the complexity of the interactions involved at a very large scale. The model we are elaborating might be a prototype of ecosystem management models at the scale of entire industrial states.

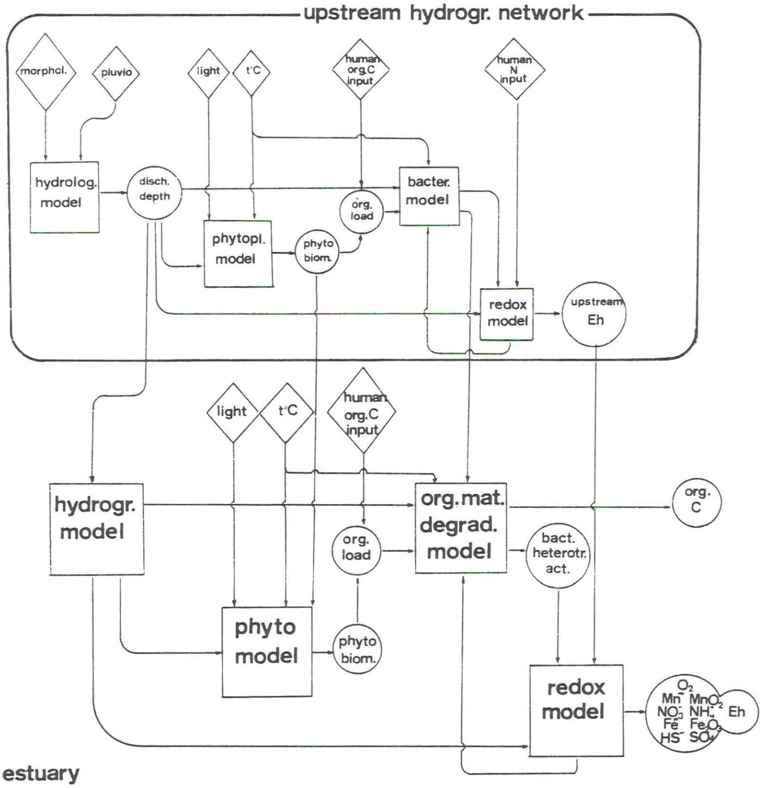

Fig. 10. Diagrammatic representation of the coupling of the watershed model with the estuarine model (Fig. 8).

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THE SALT MARSHES OF THE WESTERSCHELDE AND THEIR ROLE IN THE ESTUARINE ECOSYSTEM *

AD H.L. HUISKES

KEYWORDS : Westerschelde; salt marshes; carbon fluxes; nutrient fluxes.

ABSTRACT

The salt marshes along the Westerschelde estuary have been influenced by various human activities of which reclamation has been a major cause for the loss of salt marsh area. The salinity gradient in the aquatic system is also mirrored in the vegetation of the salt marshes.

The role of the salt marshes for the estuary as a whole is manifold but a major importance is their function as a sink for anthropogenic substances.

The possible role as a carbon and mineral source for the estuary is discussed in this paper. It is estimated that the total area of salt marsh adds about 8 % to the organic matter input in the estuary while the nutrient input may be as high as 25 %.

INTRODUCTION

Estuaries such as the Westerschelde have a number of functions of which especially the economic and ecological functions frequently conflict. The salt marshes as part of the estuary are no exception to this rule, especially because of the space they occupy. Space that is commonly expensive since estuaries are situated in the most populous and most heavily industrialized areas of the world. The Westerschelde estuary is no exception to this rule.

SAEIJS (1977) gives a survey of the salt-marsh areas in the Westerschelde estuary in 1958 (Fig. 1). Since that year a substantial area of marsh has been lost, due to engineering works (Fig. 2).

Momentarily the Westerschelde estuary harbours about 3300 ha of salt marsh from the Dutch-Belgian border to the North Sea. From the border upstream up to Antwerpen a few fragments of marshland remain (Fig. 2). Salt marshes along estuaries are influenced by the salinity gradient in the estuarine water, which results in a characteristic array in especially the flora of these marshes. A nice descriptive review of the changes in marsh vegetation along the Schelde and the Westerschelde due to salinity changes are found in SLOFF (1936). This paper shows at the same time the losses of salt-marsh area since then when compared with figure 2 and more recent papers (BEEFTINK, 1957; WESTHOFF and VAN DER MAAREL, 1973) which deal in a more quantitative way with the changes in the vegetation along the salt gradient (Table 1). Although the changes in vegetation are the most obvious, changes in composition of the fauna along the gradient occur as well (*e.g.* VAN SOELEN and MARKUSSE, 1985; WITTEVEEN, 1986).

HUMMEL *et al.* (this volume) divides the Westerschelde in three compartments with divisions near Hansweert, this is about the 10 $^{o}/_{oo}$ Cl^{-} - isohaline, and near Kallo, the 1 $^{o}/_{oo}$ Cl^{-} - isohaline. Hummel's division is a simplification of PEELEN's (unpubl.) (Fig. 3). The isohalines mark the distribution limit of some plant species. West of the 10 $^{o}/_{oo}$ Cl^{-} - isohaline we find plant species like *Limonium vulgare* and *Halimione portulacoides.* These species occur East of the line but rarely. The line marks also the western limit of *Cochlearia officinalis* a

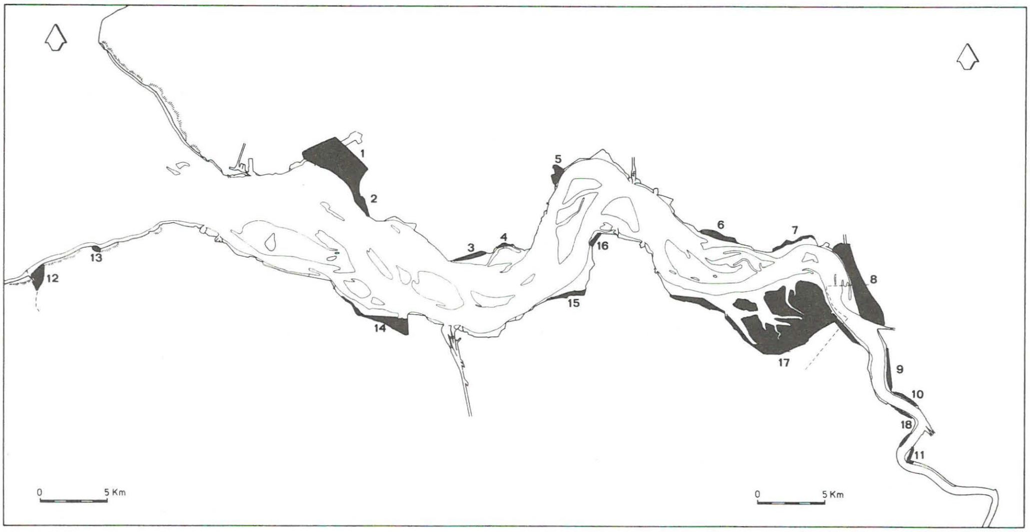

Fig. 1. Situation of salt marshes along the Westerschelde estuary in 1958 (after BEEFTINK, 1957). 1 = Zuid-Sloe; 2 = Kaloot; 3 = Zuidgors near Ellewoutsdijk; 4 = Baarland marsh; 5 = Biezelingse Ham; 6 = Waarde marsh; 7 = Marsh West of Bath; 8 = Marshes near Ossendrecht and Zandvliet; 9 = Galgeschoor; 10 = Marsh near Oude-Kermisschans; 11 = Marsh near St. Philippe fortress; 12 = Zwin; 13 = Verdronken Zwarte Polder; 14 = Marshes of the Braakman; 15 = Marsh of the Hellegatpolder; 16 = Marsh near Ossenisse; 17 = Verdronken Land van Saeftinge; 18 = Marshes between fortresses Liefkenshoek and Ste. Marie.

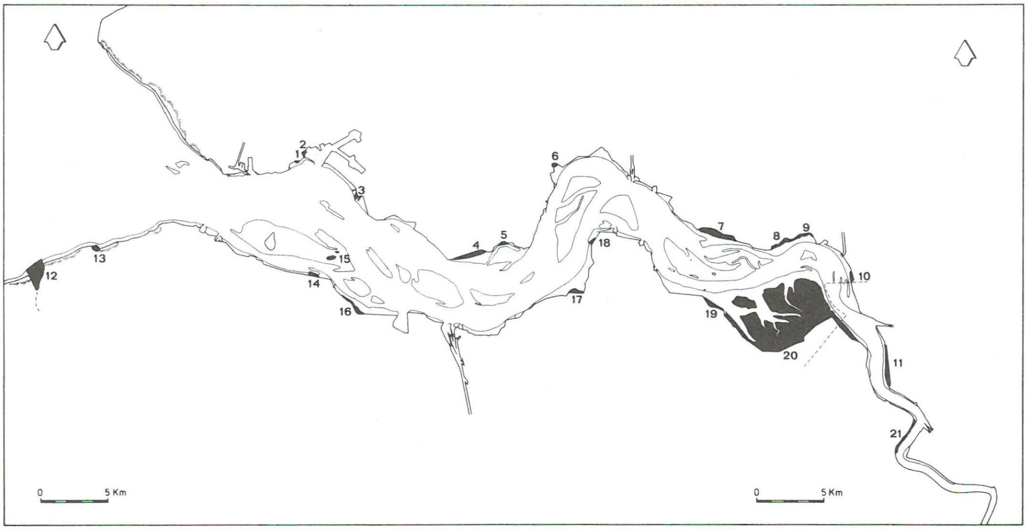

Fig. 2. Situation of salt marshes along the Westerschelde estuary in 1987. 1 = Sand spit near Rammekens Fortress; 2 = Rammekenshoek; 3 = Kaloot; 4 = Zuidgors near Ellewoutsdijk; 5 = Baarland marsh; 6 = Biezelingse Ham; 7 = Waarde marsh; 8 = Zimmermanpolder marsh;

9 = Marsh West of Bath; 10 = Marsh near Ossendrecht; 11 = Galgeschoor; 12 = Zwin; 13 = Verdronken Zwarte Polder; 14 = Marsh near Hoofdplaat; 15 = Hoge Springer; 16 = Marsh of the Paulinapolder; 17 = Marsh of the Hellegatpolder; 18 = Marsh near Ossenisse; 19 = Marsh near Baalhoek; 20 = Verdronken Land van Saeftinge; 21 = Marshes between fortresses Liefkenshoek and Ste. Marie.

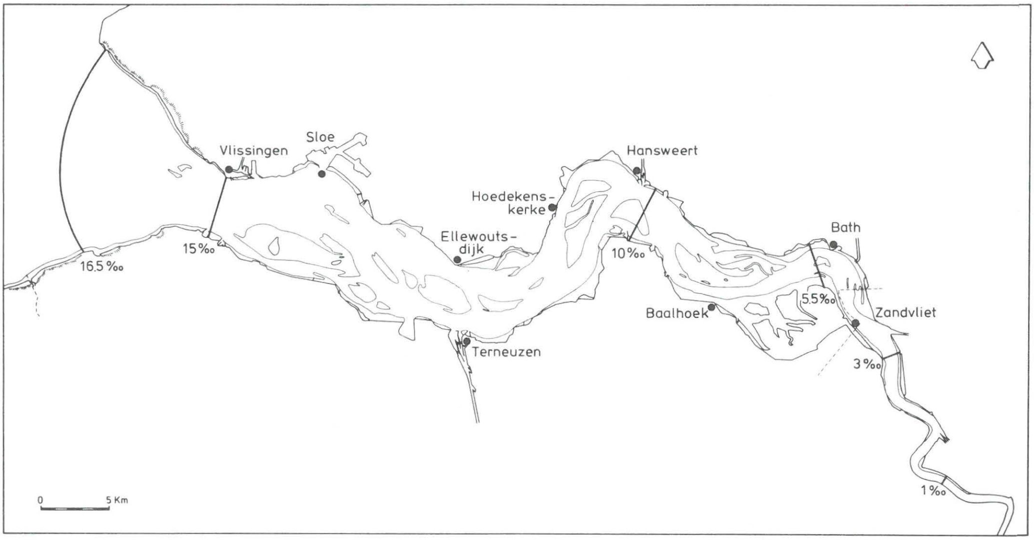

Fig. 3. Isohalines at midtide and average river discharge (105 $m^3.sec.^{-1}$). (PEELEN, unpubl.).

plant species of fresh and brackish marshes and *Scirpus maritimus* and *Phragmites australis*. The 1 $^{0}/_{00}$ Cl^- - isohaline marks the occurrence of salt marshes along the Schelde. It is also the seaward limit of *Caltha palustris* (BEEFTINK, pers. comm.).

The role of the salt marshes in the estuarine ecosystem is manifold. Their importance for ecological research is unquestionable. Their hydrological role as a flood reservoir can be large. In the Westerschelde this is only the case in the eastern part. Their role as sink for anthropogenic substances is important (see *e.g.* OENEMA *et al.*, this volume).

The salt marshes along the Westerschelde are also important for nesting and migrating birds. The 'Verdronken Land van Saeftinge' harbours the largest colony of the Black-headed gull (*Larus ridibundus*) and of the Herring gull (*Larus argentatus*) outside the dunes (M. JACOBUSSE, pers. comm.). It is one of the few areas in the country where the Corn bunting (*Emberiza calandra*) breeds. SAEIJS and BAPTIST (1977) conclude from their bird studies that the wetlands along the Westerschelde may be ranked in the 'A' category of the MAR-list (criteria by Szijj (CARP, 1971)).

The role of the Westerschelde salt marshes as nursery grounds for fish and shellfish is limited. Only for the brown shrimp (*Crangon crangon*) the shallow eastern parts of the estuary play a role, especially the tidal creeks in salt marshes (DE VEEN *et al.*, 1979).

Another role of the salt marshes however, is their source- and sink function, especially with respect to carbon and minerals. Salt marshes are highly productive ecosystems. They are wetlands and all wetlands together account for 2.3 % of the world's net productivity although they occupy only 0.4 % of the world's area (LIETH and WHITTAKER, 1975). They equal in productivity the tropical rain forests. This yearly production dies off and is eventually washed out of the salt marsh into the estuary, deposited against the dikes or decomposed in the marsh itself. Litter within the marsh comes from various sources but about

80 % of it originates from higher plants (PERKINS, 1974). Few data are available for Dutch salt marshes. KETNER (1972) studied a marsh in the Wadden Sea area and estimates for a middle marsh vegetation (*Puccinellia maritima*) a maximum live biomass of 463 g.m^{-2} (4.63 tonnes per hectare) and a maximum dead biomass of 277 g.m^{-2}. Results of WOLFF *et al.* (1980) for the same vegetation in an Oosterschelde marsh are somewhat higher respectively 576 and 384 g.m^{-2}.

Standing-crop however is not synonymous to production, the latter being dynamic over a complete growing season. Moreover the above-mentioned figures are based on the above-ground plant material, the below-ground biomass is not accounted for. GROENENDIJK (1984) and GROENENDIJK and VINK-LIEVAART (1987) give production estimates instead of standing crop figures and they observed below-ground production as well (Table II). The figures in this table were obtained by using the so-called paired-plot method. This is a time-consuming method but the estimate of the yearly production is more accurate. This method accounts for the disappearance of dead tissue during the growing season (WIEGERT and EVANS, 1964). Another commonly used method does not account for the loss of dead tissue (SMALLEY, 1959). GROENENDIJK (1984) compared both methods and discussed their advantages and disadvantages (Table III). From this study it appeared that 70 % of the yearly production of *e.g.* a *Spartina anglica* and an *Elymus pycnanthus* vegetation is below ground. For a *Halimione portulacoides*-vegetation this was an even higher percentage (about 80 %). GROENENDIJK (1984) proved that below-ground production is very important indeed and conclusions based on above-ground production only, may lead to a completely different outcome. For instance WOLFF *et al.* (1980), working in the same marsh area as GROENENDIJK (1984) and GROENENDIJK and VINK-LIEVAART (1987), concluded that the marsh imported particulate organic matter while the latter authors came to a net export of material.

GROENENDIJK and VINK-LIEVAART (unpubl.) summarize their findings in Figure 4. Although there is still a number of question marks in the scheme it is clear that a substantial amount of biomass produced is exported to the estuarine waters, 600 g.m^{-2}.y^{-1}, which is the same figure HAINES (1979) found in Georgia marshes. And this figure is derived from the above-ground production only. It is sensible to assume that this figure will only increase when the fluxes of the below-ground production will be clarified.

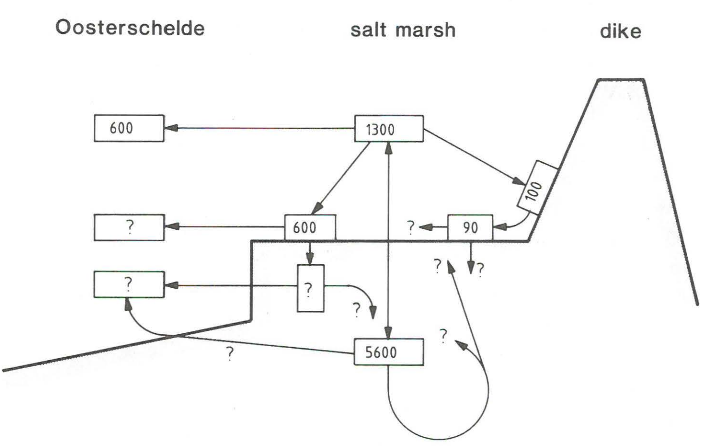

Fig. 4. Production and transport of plant biomass in the salt marshes along the Oosterschelde. The figures are in grams per m^2. (After GROENENDIJK and VINK-LIEVAART, unpubl.).

The production figures of GROENENDIJK (1984) were used for the Westerschelde marshes in the following calculation. Taking the total marsh area of the Westerschelde, which is about 3300 hectares, the total above-ground production of biomass will be $4{,}4 \cdot 10^4$ tonnes yr^{-1} and the below-ground biomass $1{,}9 \cdot 10^5$ tonnes yr^{-1}. According to GROENENDIJK and VINK-LIEVAART (unpubl.) 46 % of the above-ground biomass will be exported which is $20 \cdot 10^3$ tonnes yr^{-1}. It is difficult to say anything about the fate of the below-ground biomass. Some authors (TEAL, 1962; ODUM and DE LA CRUZ, 1967; VALIELA *et al.*, 1976) assume that only the above ground production is exported. The recirculation of plant litter *in situ* would however almost certainly increase the organic matter content in the soil, which is not found in salt marshes in tidal situations. An unknown, but substantial portion of the below-ground biomass therefore must be exported, probably as minerals. What happens with this amount of organic matter is virtually unknown. Most of it comes from the marshes in the eastern, brackish part of the Westerschelde as 85 % of the marsh area is situated here. It was assumed that the production of organic matter is the same as in saline marshes, but is probably higher.

HEMMINGA and VAN SOELEN (unpubl.) showed that a substantial amount of plant

Table 1. Presence of the principal species in the salt marshes along the estuary of the river Schelde (the numbers of the salt marshes are the same as in Fig. 1). (After BEEFTINK, 1957). +++ = abundant, ++ = common, and + = scarce.

	Species	Salt marsh									
		2	3	4	5	6	7	8	9	10	18
1.	*Salicornia europaea*	+++	+++	++	++	+					
2.	*Suaeda maritima*	+++	++	++		+					
3.	*Armeria maritima*	+	+			+					
4.	*Limonium vulgare*	+++	+++	+++	++	++	++				
5.	*Halimione portulacoides*	+++	+++	+++	+++	+	+				
6.	*Artemisia maritima*	+++	+++	+++	+++	++	+				
7.	*Spartina townsendii*	++	++	++	++	++	+	+			
8.	*Spergularia marginata*	++	++	++	++	++	++	+			+
9.	*Plantago maritima*	+++	++	+++	+++	+++	+++	+	++	+	+
10.	*Triglochin maritimum*	+++	+++	+++	+++	+++	+++	++	+++	++	+
11.	*Aster tripolium*	+++	+++	+++	+++	+++	+++	++	+	+++	++
12.	*Festuca rubra* subvar. *littoralis*	+++	+++	+++	+++	+++	+++	+++	+++	+++	++
13.	*Agropyron littorale*	+++	+++	+++	+++	+++	+++	+++	+++	+++	++
14.	*Puccinellia maritima*	+++	+++	+++	+++	+++	+++	+++	+++	+++	+++
15.	*Atriplex hastata*	+	++			++	++	+++	+	++	+
16.	*Glaux maritima*	+	+	++	+++	+++	+++	+++	+++	+++	+++
17.	*Juncus gerardi*		+			+	+		+++	+++	+
18.	*Agrostis stolonifera*					+	++	+++	+++	+++	+++
19.	*Scirpus maritimus* var. *compactus*							+++	+++	+++	+
20.	*Phragmitis communis*							++	++	+++	
21.	*Spergularia salina*								+		+
22.	*Alopecucurs geniculatus* var. *tuberosus*								++		++
23.	*Hordeum secalinum* var. *marinum*								++		++
24.	*Plantago major*								+		++
25.	*Trifolium fragiferum*								+		
26.	*Lolium perenne*									++	+++
27.	*Festuca arundinacea*									+	+
28.	*Agropyron repens*										+

Table 2. Annual primary production of above- and below-ground parts of plant species in an Oosterschelde salt marsh (after GROENENDIJK, 1984 and GROENENDIJK and VINK-LIEVAART, 1987).

Species	Production $g.m^{-2}.yr^{-1}$	Above/below-ground
Spartina anglica	2250	above
	6044	below
Elymus pycnanthus	1787	above
	4421	below

Table 3. Annual above-ground primary production of some plant species in an Oosterschelde salt marsh.

Species	Production $g.m^{-2}.yr^{-1}$	Method	Author
Spartina anglica	723	Smalley	WOLFF *et al.*, 1980
	1649	Smalley	GROENENDIJK, 1984
	2250	Wiegert-Evans	GROENENDIJK, 1984
Elymus pycnanthus	1008	Smalley	WOLFF *et al.*, 1980
	850	Smalley	GROENENDIJK, 1984
	1787	Wiegert-Evans	GROENENDIJK, 1984

debris from brackish marshes containing dead parts of *Scirpus maritimus* was found in the strandline in the western part of the Westerschelde. This is predominantly above-ground biomass however. Assuming that below-ground biomass is exported in the same proportion but in a mineralized form, the figure by HUMMEL *et al.* (this volume) has to be increased. The total export of above-ground biomass from the salt marshes increases the detritus box with 8 %. Assuming that 41 % (HUMMEL *et al.*'s figure) of this detritus is mineralized in the water ($0{,}41 \,.\, 20 \,.\, 10^3$ tonnes yr^{-1}) and that of the below-ground production the whole fraction is exported as minerals (being $8{,}5 \,.\, 10^4$ tonnes yr^{-1}) the nutrients-box of this figure has to be increased with approximately 25 %!

These figures are as yet hypothetical, but not unrealistic. They show that salt-marshes may contribute in a minor way to the detritus pool in the estuary, but their role in the nutrient pool might be significant.

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A SYNTHESIS OF THE RESEARCH ON THE ECOLOGY AND CHEMISTRY OF THE SCHELDT ESTUARY

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During the workshop on the ecology of the Scheldt estuary, Terneuzen, The Netherlands, 27 May 1987, representatives of the biology group discussed a number of questions concerning the following topics :

THE RELATIONSHIP BETWEEN PRIMARY PRODUCTION AND THE DETRITIAL FOOD CHAIN

In connection with the paper presented by HUMMEL *et al.* (this volume) the discussion focused on how much is known about the processes involved in this relationship. Hummel's model afforded some insight into the major pathways of the carbon flow in coastal (salt water) and detrital (brackish water) food chains. However, the model was based partly on estimated biomass values : no information was available on production and consumption. Moreover, primary production in the fresh- and brackish water part of the estuary was assumed to be low. BILLEN *et al.* (this volume) however, were able to demonstrate that primary production could be high in these areas.

LONG-TERM AND SPATIAL BIOMASS FLUCTUATIONS OF MACROBENTHOS

The biomass of the benthos was assumed to be very low in the Belgian part of the estuary, because of pollution and a lack of oxygen in the waters. However, VAN IMPE (1985) showed that a small number of species could nevertheless attain a very high biomass in that part of the estuary. During the workshop, measurements such as those taken by Van Impe were the subject of lively debate, and further research on this subject was recommended.

A BUDGET FOR THE SALT-MARSHES

According to a budget-model presented by HUISKES (this volume) there would be a net export of the carbon produced in the Saeftinghe salt-marsh. It was argued that the model had too many 'open ends', and that any conclusions as to the import or export of material would be premature. The model promped a wide-ranging discussion and presented scientists with an opportunity to cooperate in this matter in the near future.

PATHWAYS OF POLLUTANTS AND THEIR EFFECTS ON ORGANISMS

One of the main questions was : 'Does selection of different species or races take place due to the pollution in the Scheldt estuary ? ' It was suggested that a comparison should be made of the various organisms present in the Scheldt estuary and in the less heavily polluted Eems-Dollard. It was thought by some that the exchange of genes is almost unlimited in the aquatic ecosystems of the tidal areas, and that the influence of pollutants on the genetic constitution of organisms could therefore best be studied in the semiterrestic ecosystems, *i.e.* the salt marshes.

HUMAN IMPACT ON THE ECOSYSTEM

The effects of dredging, the role of tidal flats and marshes and the extensive discharge of labile organic matter and nutrients in the Scheldt ecosystem were discussed. It appeared that there is insufficient basic knowledge on the Scheldt ecosystem for a proper answer to be given to these complex questions (HEIP, this volume).

The questions discussed by the Chemistry working group concerned pollution, chemical speciation and models. The tentative theoretical review presented by DUURSMA *et al.* (this volume) on various physico-chemical processes in estuaries was clarified with the aid of data on the behaviour of PCB's and plutonium in the Scheldt estuary.

After a useful discussion on the gradients and speciation of different compounds in the estuary and the bioavailability of these compounds, it was concluded that there was still an enormous lack of data and knowledge on the processes and fate of different pollutants in estuaries, especially in the Scheldt estuary. It appeared that the speciation models are insufficient as a basis for describing the behaviour of different fractions in the water phase, sediments and organisms. It was suggested that, for the time being, all the different physico-chemical processes should be put into a black box and that the bioaccumulation of pollutants should be used as an indicator of pollution.

One important item of mutual interest was how to standardize the concentration of micropollutants in sediments and organisms. During the discussion it became clear that different institutes are already working on the problem of micropollutants in sediments.

During the symposium and in the biology and chemistry workshops, different types of model were presented in the form of papers and diagrams. During the plenary closing session the models together with other possible models were summarized :

- Hydrodynamic models
- Advective-dispersive models
- Suspended matter models
- Carbon (energy) models, ecomodels
- Oxygen models
- Nutrient models
- Spatial models
- Food-uptake models
- Bioaccumulation models for toxic compounds
- Toxic effect models

Due to the lack of data, the discussion on models was rather academic. The discussion focused mainly on which model was most appropriate for describing the Scheldt estuary. In general, from a scientific point of view, ecological models were preferred. One group wanted to construct more specific models, known as dedicated models, which refer to well-defined problems. It was not the aim of this workhop to choose one particular model. The discussion on how to use the different models made it clear that several models should be used simultaneously.

Although the discussion in the biology group was more concerned with simulation models, the general consensus on the topics discussed in both groups was that hardly any data were available for the Scheldt estuary and it was necessary to obtain more information as quickly as possible.

During the symposium, different scientists from different institutes and different countries gave suggestions on cooperation in the near future (see HUMMEL's Introduction, Table 1b).

A special data bank and a proposed 'Scheldt Year' were the positive results of a two-day symposium where scientists from different countries developed a common approach to the problems of the Scheldt estuary.

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