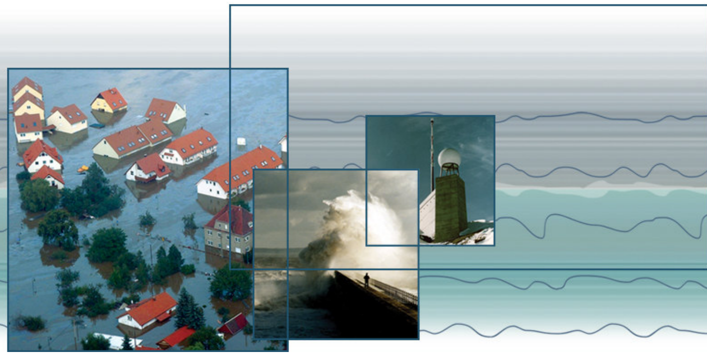
Integrated Flood Risk Analysis
and Management Methodologies



Toxic Stress:

THE DEVELOPMENT AND USE OF THE OMEGA MODELING FRAMEWORK IN A CASE STUDY – FINAL REPORT

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SUMMARY

Flooding not only induces a risk which is related to direct loss of life for humans, animals and plants. During flooding, toxicants which are normally stored or otherwise unavailable might be released. The spreading of these pollutants during the flooding or, often more important, the persistent presence of toxicants in the soil or sediment after the flooding poses an environmental risk.

Within Task 10 of FLOOD*site*, the effect of toxic stress induced by flooding was studied in a case study (Western Scheldt). For this case study, a relation between toxic concentration levels in water and sediment and acute and chronic toxic stress levels on organisms was needed. The scientific principle on which the EU Water Framework Directive (WFD) is based is that relationships between the biological state and physical and chemical properties of surface waters dictate the ecological state of the water system. Toxic stress is part of the chemical properties of the water system. Within the EU 6th framework programme REBECCA (Contract SSP1-CT-2003-502158) the relation between concentration levels of individual toxicants and their potential total toxic effect was researched. This knowledge can then be used within FLOOD*site* to calculate ecotoxicological stress during and after flooding.

The aim of this research was to develop a model framework in which the ecotoxicological stress levels in the water phase can be predicted (OMEGA). The model includes insights in how to combine the toxic risk of different toxicants. This model was used in a case study for the Western Scheldt for an area with nature reserve areas which is regularly flooded to predict the toxic risk for different groups of organisms.

Also, a case study for a simulated dike breach flooding in the Western Scheldt area (nearby the city of Middelburg) was carried out to establish the spreading of pollutants during a flood. The source and 'fingerprint' of the pollutants was diverse and based on land use (farming, industry, etc.), urban area and background concentrations in the water. By 'fingerprint' is meant that each source had a different mixture of toxic components. The results of this case study were maps with areas of high and low pollution after a spreading, which could thereafter be used for toxic risk prediction.

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1.1 Introduction

During flood events, huge amounts of sediment are transported to the inundated area. The irregular supply of thousands of tons of sediment may shape the inundated area. Due to the occasional sedimentation of massive amounts of sediments this situation is not only from the morphology point of view undesired, but also from the water quality point of view. The excess of sedimentation during flood events carries with it significantly high levels of pollutant fractions absorbed to the sediment. Moreover, the transport and distribution of pollutants from storage facilities or industries (calamity events) is another risks during flood events. High concentrations of pollutants offer high ecotoxicological risks not only for plants and animals, but also for humans.

The Water Framework Directive is focused on good ecological status. This means that habitat models to predict ecosystem health and compare the actual status with the desired reference system are an important tool. Habitat models need to be fed by cause-effect relations between pressures and effects to evaluate which pressures are important in striving for a reference system. Inundation, erosion and sedimentation are relevant physical pressures on changes in the habitat that require to be further studied.

The ecotoxicological risk prediction based on chemical concentrations in the water phase has been applied for the long term (chronic) toxic stress within the ‘normal’ (non-flooded) water system of the Western Scheldt. The choice of the Western Scheldt as water system for the case study was based on the broader scope of the FLOOD*site* project. In addition a case study near the city of Middelburg was carried out for the spreading of polluted sediments caused by a dike breach. This city is situated in the Western Scheldt area and was part of the broader FLOOD*site* case study area selection.

1.2 Objectives

Floods have impacts on ecosystems through erosion, sedimentation, inundation and through alterations in habitat conditions to flora and fauna. Next to this, floods may bring along and spread polluted sediments, and also cause the release and spread of pollutants within a flooded area, e.g. from chemical sites, oil tanks, sewage treating plants, and pesticide supplies. This work in Activity 3 of Task 10 of the FLOOD*site* project aims at improving the assessment of flood impacts and effects of flood induced pollution on ecosystems, individual species and on relevant indicators for ecological quality (e.g. for biodiversity, naturalness, ecosystem health, similarity to a reference system (standards, where appropriate related to the Water Framework Directive, rareness, etc.).

The work will proceed in the following Actions:

- State-of-the-art on environmental vulnerability
- State-of-the-art ecotoxicological models
- Definition of relevant indicators for ecological quality
- Case study based on new state of the art knowledge

1.2.1 State-of-the-art on environmental vulnerability

The objective of this Action is to make a review of the state-of-the art knowledge on environmental vulnerability to flooding. Relevant substances, processes and cascades of effects determining an area’s vulnerability from both a physico-chemical and a biological point of view will be examined.

This part of the study has been carried out in a more general way and has not been applied for the case study area. The results are therefore published in Appendix A.

1.2.2 State-of-the-art ecotoxicological models

In this Action the focus is on ecotoxicological risks in relation to flood induced changes in the system on physical pressures. The objectives of this action are to make a review of state-of-the-art ecotoxicological models and ecological models for ecotope and habitat suitability prediction, resulting in a set of requirements which ecological models should fulfil to ensure a good applicability in flooded sites.

This Action is a very detailed first approach (fundamental research) with a practical spin off for knowledge rules suitable for FLOODSITE to be implemented in more global models (like GIS or Sobek 1d/2):

- Development of a detailed (mm scale) thermodynamic speciation model for predicting the freely dissolved concentration at different locations and at different stages of the year (running water versus stagnant water or inundation versus dry top layer) in a flood plain.
- Coupling of a speciation model with an ecotoxicologal risk assessment model (OMEGA) based on the freely dissolved concentration for risk evaluation purposes.
- Translation of the detailed risk prediction model to more global knowledge rules for risks in flood plains to be incorporated in regional models like Sobek-1d/2d, leading to a GIS based risk map in time and place.

1.2.3 Definition of relevant indicators for ecological quality

Having to use a complete deterministic model framework for a first analysis of flooding risks is relatively time consuming. A more simple spread sheet like approach with a focus on identifying the main risks and bottlenecks for maintaining a good ecological status is in some cases sufficient.

The objective of this Action is to define relevant indicators (both physical and chemical) for ecological quality, as derived in Actions 1 and 2. Simplification of knowledge rules to a more spreadsheet-like approach for the major cause-effect relations between pressures (physical and chemical) and ecological effects are addressed.

1.2.4 Case study

While the development of improved models based on new state of the art knowledge can help in grasping the mechanisms and differences between (as an example) an expected good ecological quality and an observed poor one, testing the overall model framework in a ‘real world’ situation is crucial to validate the new relations.

The objective of this Action is to use the state of the art environmental vulnerability and ecological models (as developed in Actions 1, 2 and 3) in a case study for the Western Scheldt to test the added value of better predicting the ecological status and the main pressure factors that influence this status.

Based on the developed overall model framework and observed input data for the Western Scheldt a case study will be carried out to determine if the new state of the art knowledge models can be applied in a 'real world' case to yield a better understanding of the actual ecological state and the pressure factors that influence this state.

1.3 Report outline

In this report, the principle on which ecotoxicological effect prediction is based is explained, including the model framework OMEGA in which this ecotoxicological stress level can be predicted (**Chapter 2**). The model includes insights in how to translate total water or even sediment concentrations to dissolved concentration (the dissolved concentration is the base for ecotoxicological effect prediction, **Chapter 3**).

In **Chapter 4** the Western Scheldt is evaluated on the short term (acute) toxic stress. With the use of a 1D/2D water quality model the concentrations of the most important substances in the Western Scheldt are calculated and then used in the OMEGA model to calculate the ms-PAF and the contribution of each model to the ms-PAF.

During flood events, huge amounts of sediment are transported to the inundated area. The excess of sedimentation carries with it significantly high levels of pollutant fractions absorbed to the sediment. Moreover, the transport and distribution of pollutants from storage facilities or industries (calamity events) is another risks during flood events. **In Chapter 5** this is presented in a case study for Middelburg (Zeeland, the Netherlands) based on the collapse of dikes and the inundation of the land with the use of a 1D/2D water quality model.

The general conclusions of this report on flood induced pollution are given in **Chapter 6**.

Water storage has been proposed for both problems: drought and flooding damage to nature conservation areas. In **Appendix A** the paper "Water storage in nature conservations areas: does current knowledge suffice?" (Haasnoot, Klijn and van Ek) is presented. This paper addresses the Review of the state-of-the art knowledge on environmental vulnerability to flooding

2. Introduction to toxic risk prediction

2.1 Introduction to the OMEGA modeling framework

That certain chemical components cause a toxic effect in organisms or plants is a well know fact. While banning all toxic components from water bodies is desirable, it's often not possible to eliminate all sources of a specific toxicant. Therefore, to set environmental safety standards, the relation between toxic effects and chemical components concentrations has to be established.

Within the EU Water Framework Directive, the targets for priority substances for water bodies are set by risk levels (like AA-QS and MAQ-QS). These risk levels are based on observed effects in ecotoxicological risk assessments. The ecotoxicological risk assessments are carried out by establishing the concentration-effect relation for individual chemical components and individual test species (single-species toxicity tests, measuring effects to individuals) (Posthuma *et al.,* 2002; Kooijman SALM, 1987; Wagner and Løkke, 1991; Aldenberg and Slob, 1992).

However, the WFD is based on preserving or improving the ecological status of the whole water body. On the water body scale, 'simple' dose effect relations between individual toxicants and individual species have to be aggregated to an evaluation of the toxic effect of chemical components for the whole water body. To resolve this incongruity between individual-based data and the complex biological entities addressed in ecological risk assessment, a model framework (OMEGA) will be introduced to link individual species-sensitivity distributions to risk levels on the water body scale.

2.2 Risk assessment based on species sensitivity distribution

Single-species test data are combined to predict concentrations affecting only a certain percentage of species in a community. Single-species data (e.g., median lethal concentration [LC50] or no-observed-effect concentration [NOEC] values) for many species are fit to a distribution such as the lognormal or log-logistic. From this distribution of species sensitivities, a hazardous concentration (HCp) is identified at which a certain percentage (p) of all species is assumed to be affected. The most conservative form of this approach uses the lower 95% tolerance limit of the estimated percentage to ensure that the specified level of protection is achieved (Posthuma *et al.,* 2002; Kooijman SALM, 1987; Wagner and Løkke, 1991; Aldenberg and Slob, 1992).

Species-sensitivity distribution or extrapolation methods are being incorporated into assessments of ecological risk (Newman *et al*., 2000). In Figure 2.1 an example of the SSD distribution based on individual NOEC's for one chemical component is given. In Figure 2.2, the individual NOEC level for species is transferred to a Potential Affected Fraction (PAF) level for the toxicant.

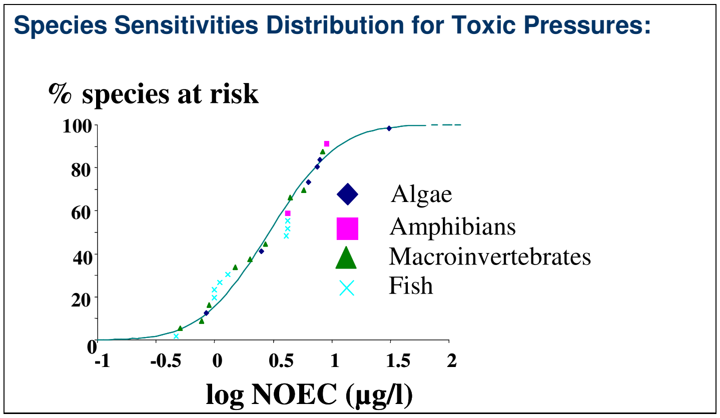


Figure 2.1 Combining individual NOEC levels

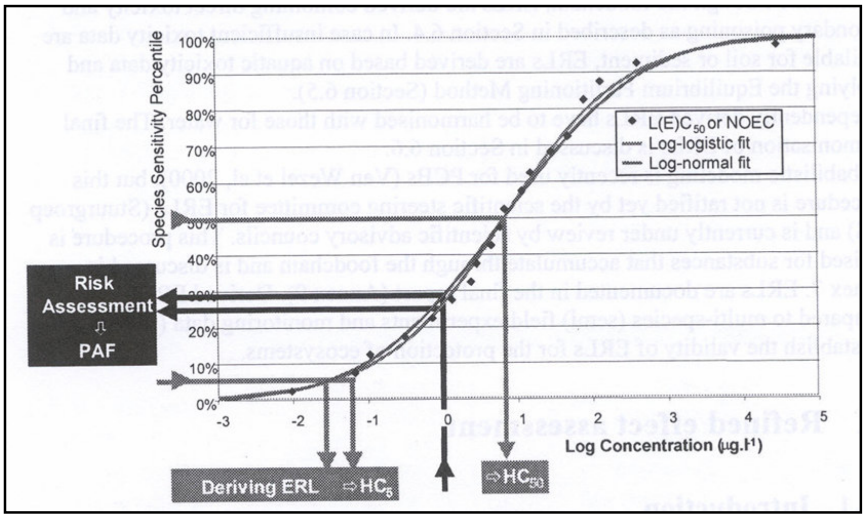


Figure 2.2 From NOEC's to Potential Affected Fraction (PAF)

2.3 Modelling framework

OMEGA stands for Optimal Modelling for Ecotoxicological Assessment, predicting effects on plants, animals and populations or ecological functions. The method is based on a stepwise approach:

1. Calculation of the potentially affected fraction of species (PAF).
2. Identification of sensitive species or species groups.
3. Calculation of accumulation in food chains.
4. Calculating effects on development of populations.

Within the current ecotoxicological risk predicament, the focus is on the first two steps within the model.

Single substance PAF
OMEGA is the aggregation of individual SSD curves to calculate a Potential Affected Fraction (Figure 2.3).

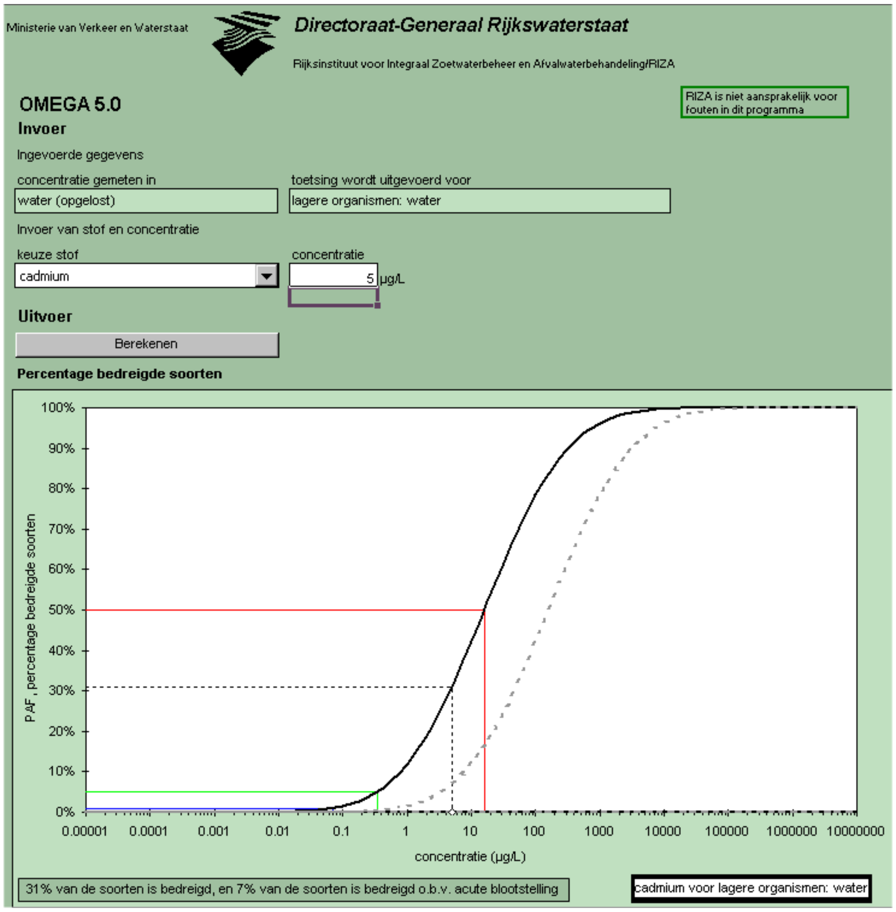


Figure 2.3 Example of PAF for Cadmium

The PAF within OMEGA is based on a wide variety of NOEC's for individual test organisms (Figure 2.4).

soort/proces	NL naam	groep	NOEC (µg/L)	QSAR?
Hyalella azteca	roeipootkreefje	kreeftachtigen	0.29	
Ceriodaphnia dubia	watervlo	kreeftachtigen	0.41	
Peridinium sp.		algen	0.56	
Daphnia magna	watervlo	kreeftachtigen	0.86	
Moina macrocopa		kreeftachtigen	1	
Emiliana huxleyi		algen	1.1	
Hymenomonas carterae		algen	1.1	
Prorocentrum micans	dinoflagellaat	algen	1.1	
Asterionella glacialis		kiezelalgen	1.1	
Artemia salina		kreeftachtigen	1.1	
Chara vulgaris	kranswier	waterplanten	1.1	
Gammarus fasciatus	roeipootkreeftje	kreeftachtigen	1.2	
Oncorhynchus kisutch		vissen	1.3	
Daphnia pulex	watervlo	kreeftachtigen	1.6	
Chironomus riparius	vedermug	insekten	2	
Allorchestes compressa	vlokreeft	kreeftachtigen	2.1	
Oncorhynchus mykiss	regenboogforel	vissen	2.1	
Salvinia minima		waterplanten	2.2	
Salvelinus fontinalis		vissen	2.5	
Aplexa hypnorum		weekdieren	2.5	
Xenophus laevis		amfibieën	3	
Jordanella floridae		vissen	3	
Dreissena polymorpha	driehoeksmossel	weekdieren	3	
Heterocapsa triquetra		algen	3.4	
Lithodesmium undulatum		algen	3.4	
Synechococcus bacillaris		algen	3.4	
Thoracosphaera heimii		algen	3.4	
Biddulphia mobiliensis		kiezelalgen	3.4	
Mysidopsis bahia		kreeftachtigen	3.7	
Oncorhynchus trutta		vissen	3.8	
Catostomus commersoni	zuigkarper	vissen	4.2	
Esox lucius	snoek	vissen	4.2	
Micropterus dolomieui		vissen	4.3	
Salvelinus namaycush		vissen	4.4	
Cancer anthonyi	krab	kreeftachtigen	5	
Pimephales promelas		vissen	5	
Limnanthemium cristatum		waterplanten	5	
Crassostrea virginica	oester	weekdieren	5	
Mysidopsis bigelowi	garnaal	kreeftachtigen	5.1	
Spirodela punctata		waterplanten	6.9	
Stizostedion vitreum		vissen	9	
Lemna trisculca	kroos	waterplanten	9	
Notropsis cornutis	glimvis	vissen	10	
Gymnodinium sp.	dinoflagellaat	algen	11	
Rhizosolenia setigera	diatomee	algen	11	
Streptotheca tamesis		algen	11	
Scenedesmus subspicatus		groenalgen	11	
Bacteriastrum delicatulum		kiezelalgen	11	
Bacteriastrum hyalinum		kiezelalgen	11	
Entosiphon sulcatum		oerdiertjes	11	
Scenedesmus capricornutum		groenalgen	15	
Ictalurus punctatus	meerval	vissen	15	
Lemna polyrriza	veelwortelig kroos	waterplanten	20	
Aelosoma headleyi		gelede wormen	30	
Scenedesmus quadricauda		groenalgen	31	
Chlorella vulgaris		groenalgen	33	
Eichornia crassipes		waterplanten	33	
Ditylum brightwellii		kiezelalgen	34	
Skeletonema costatum	diatomee	kiezelalgen	34	
Anabaena variabilis		blauwalgen	39	
Chlorella vulgaris		groenalgen	39	
Brachionus calyciflorus		raderdiertjes	40	
Mugil cephalus		vissen	45	
Anabaena flos-aquae		blauwalgen	50	
Chlamydomonas reinhardii		groenalgen	52	
Microcystis aeruginosa		blauwalgen	70	
Pseudomonas putida		bacteriën	80	
Chlorella fusca		groenalgen	83	
Clupea harengus	haring (baltisch)	vissen	100	
Dicrateria zhanjiangenis		algen	110	
Mytilus edulis	mossel	weekdieren	110	
Lepomis macrochirus		vissen	180	
Ophryotrocha labronica		gelede wormen	200	
Salmonella typhimurium		bacteriën	220	
Capitella capitata		gelede wormen	320	
Callianassa australiensis	garnaal	kreeftachtigen	320	
Chlorella pyrenoïdosa		groenalgen	330	
Monhystera microphthalma		aaltjes	500	
Ophryotrocha diadema		gelede wormen	500	
Nereis arenaceodentata		gelede wormen	560	
Tetrahymena pyriformis		oerdiertjes	670	
Scenedesmus pannonicus		groenalgen	900	
Ctenodrilus serratus		gelede wormen	1000	
Pleuronectes flesus		vissen	1000	
Dunaliella sp.		groenalgen	1100	
Monhystera disjuncta		aaltjes	5000	
Pellioditis marina		aaltjes	25000	

Figure 2.4 NOEC data within OMEGA for cadmium

2.4 New developments

The OMEGA framework (Figure 2.5) has been expanded to include:

- More ecotoxicological data.
- The use of QSAR's to fill data gaps.
- The selection of subsets of organisms.
- Combination of toxic effects of different toxicants.
- The use of measured or calculated freely dissolved concentrations (derived from surface water concentrations, whole water concentrations or pore water concentrations in sediments).

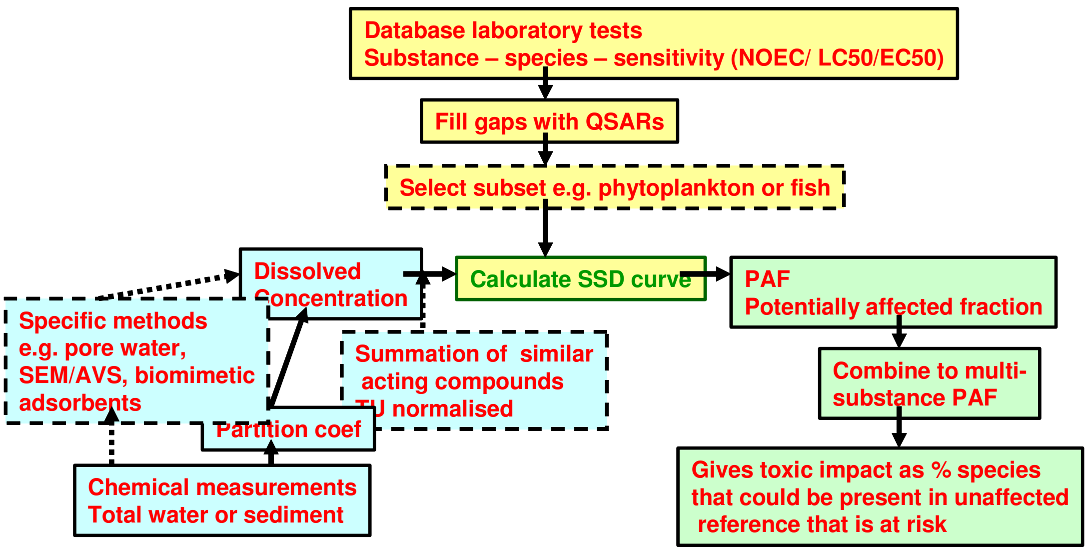


Figure 2.5 Modeling framework for determining critical concentrations (bioaccumulation) in aquatic biota (OMEGA)

While the data in the OMEGA basis has been in use for quite some time (the current Dutch legislation for surface water and sediment quality (Fourth National Policy Document, 2000) and part of the current Water Framework Directive draft fact sheets (Joint Research Centre, 2004) is based on this data), the increase in data availability and new developments like QSAR relations make it possible to improve on the relations between toxic exposure and predicted effects.

2.5 From effect on the total population to effect on specific groups

The OMEGA framework is currently based on setting a PAF risk level based on all available data. This approach is partly based on the lack of ecological cause-effect data, not using a part of the available data would yield insufficient data. Progress with regard to the availability of more ecotoxicological data and the filling of knowledge gaps with QSAR relations yield the possibility to be more restrictive in the data used to establish the NOEC level for a certain group of organisms.

At the moment, OMEGA is adopted to be able to calculate a NOEC for the following groups of organisms:

- Phytoplankton;
- Macrophytes;
- Phytobenthos;
- Benthic Invertebrates;
- Fish.

2.6 Combination of toxic effects of different toxicants / Mode of Action

OMEGA is currently mainly based on the effects of one toxicant. Combination of the toxic effects of different toxicants to calculate a multi-substance PAF (ms-PAF) is possible. This can be done according to two principles:

1. Response Addition (RA).
2. Concentration Addition (CA).

Response Addition is based on the absence of interaction between toxicants on the target site of toxic action. The mixture toxicity can be described by calculating the combined effect, assuming that there is *no correlation* between the up-take of compounds.

Developments of insight in the Mode of Action (also part of the QSAR programme, see Chapter 5.3, by Erik/IVL) makes it possible to develop a Concentration Addition model based on calculation and summation of Hazard Units for each toxicant.

Both pathways can be incorporated within OMEGA and can help to understand toxic stress in water bodies with more then one significant toxicant – see Figure 2.6.

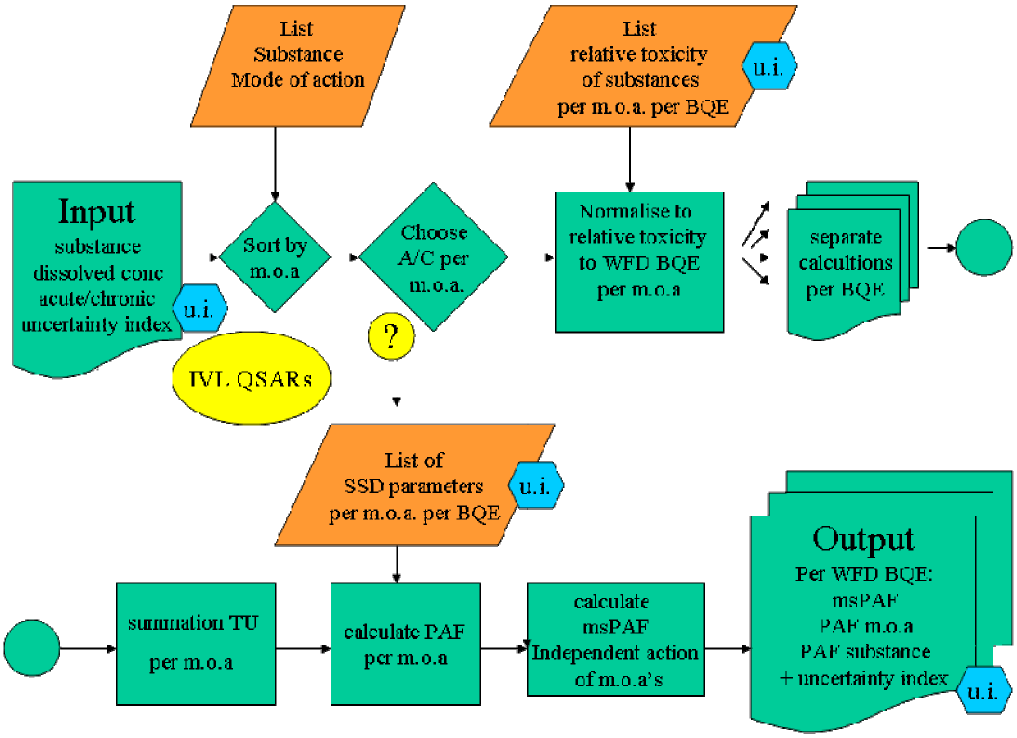


Figure 2.6 Flow schema of OMEGA input and output, including QSAR relations and decision rules

If all toxic stress relations are incorporated in OMEGA and if sufficient toxic data for the water system is present, the PAF for all toxic components can be calculated. This PAF can be split in the PAF for different groups of organisms (see Section 2.5 above).

By quantifying toxic component concentrations to a PAF, the toxic component has become a water system stress parameter. This can be done as part of a multi-stress analyses for the water system.

3. Algorithms for translating all media pollutant concentrations to freely dissolved concentrations

3.1 Algorithms for Heavy metals

3.1.1 Heavy metals, simple screening from whole water concentrations to dissolved concentrations

Uptake in water organisms is mainly through the water phase. Not all member countries measure the dissolved water phase of their water systems. Because of load criteria, the whole water approach is also very popular.

Therefore, a simple screening tool is necessary to calculate the dissolved concentration at different suspended sediment concentrations (see *Figure 3.1*).

surface water:

1) First screening on normalization based on basic partition between <u>suspended sediment</u> and <u>surface water</u>

NW4 (1998)

metals	required quality sediment (mg/kg)	required quality groundwater (ug/l)	Kp (standard sediment) (l/kg)
Cd	0.8	0.08	10000
Hg	0.3	0.01	30000
Cu	36	0.5	72000
Ni	35	3.3	10606
Pb	85	0.3	283333
Zn	140	2.9	48276
Cr	100	0.3	333333
As	29	1	29000

fraction bounded to suspended sediment

10% OM
25% lutum

metal	**5 mg/l**	**10 mg/l**	**25 mg/l**	**50 mg/l**
Cd	5%	9%	20%	33%
Hg	13%	23%	43%	60%
Cu	26%	42%	64%	78%
Ni	5%	10%	21%	35%
Pb	59%	74%	88%	93%
Zn	19%	33%	55%	71%
Cr	63%	77%	89%	94%
As	13%	22%	42%	59%

(standard Dutch practice, so only valid within a limited scope)

Figure 3.1 Example of screening for recalculating whole water concentrations to dissolved concentration

3.1.2 Heavy metals, correction on AA-QS based on macro chemical conditions in the water body

Ecological tests are carried out in standardized water. Even in this controlled test medium the biological uptake of the available dissolved fraction is somewhat influenced by complexity. But in general, there is a clear dose-effect relation based on the dissolved concentration. In natural water systems, the macro-chemical conditions vary greatly, influencing the dissolved concentration in a less linear way. High loads of a toxicant are not automatically linked to high dissolved concentrations (Cornelissen, 1999; van Steenwijk and Cornelissen, 2004; Hulscher, 2005; Schroder, 2005). But not only the total dissolved concentration is influenced by natural variation, also the complexity and therefore uptake of the dissolved concentration varies (Di Toro *et al.* 2000, 2001; Meyer, 1999).

The effect of macro-chemical variation on the water quality standards (which are related to the ecotoxicological NOEC data) has been calculated. For this, it has been assumed that ecotoxicological tests have been carried out under standardized conditions (Dutch Standard Water), Table 3.1, so that deviation from these conditions in natural conditions can lead to different risk standards (AA-QS), Table 3.2. The following macro-chemical conditions of the water have been taken into account:

- salinity;
- pH;
- alkalinity;
- Dissolved Organic Carbon (DOC) concentration.

Table 3.1 Composition of Dutch Standard Water

DWS										
	DSW	Na2SO4	MgSO4.7H2C	KCl	CaCl2.6H2O	CaCO3	MgCl2 .6H2C	NaCl	sum (gram)	mol
MG	12.2		**12.2**	0.0	0.0	0.0		0	12	0.50
CA	80.2	0.0	0.0	0.0	36.1	**44.1**		0	80	2.00
NA	69.0	23.0	0.0	0.0				**46**	69	3.00
K	7.8	0.0	0.0	**7.8**	0.0	0.0		0	8	0.20
CL	141.8	0.0	0.0	7.1	**63.8**	0.0		71	142	4.00
SO4	96.1	**48.0**	48.0	0.0	0.0	0.0		0	96	1.00
CO3	66.0	0.0	0.0	0.0	0.0	66.0		0	66	1.10
weight	**(gram/m3)**	**71**	**60**	**15**	**100**	**110**		**117**	**999.65**	**kg/m3**
	molar weight	142	120	74	111	100	95	59		
	(mol/m3)	**0.50**	**0.50**	**0.20**	**0.90**	**1.10**	**0.00**	**2.00**	**0.47**	**promille**
PH =	**8.4**									
HCO3- =	**2.24E-03**	**mol/m3**	**alk =**	**2.2**	**meq/l**		**sum alk:**	**2.27**	**meq/l**	
CO3- =	**2.64E-05**	**mol/m3**	**alk =**	**0.0**	**meq/l**					

Table 3.2 Relation between the macro-chemical conditions in the water (salinity, pH, alkalinity and DOC) on the AA-QS for heavy metals (expressed as a correction factor on Dutch Standard Water)

correction factor for	CU				
normalised on DSW					
multiplier					
factor < 1: norm has to be lowered					
factor > 1: norm can be raised					
	salinity	pH	alkalinity	DOC	free
Dutch standard water	0.5	8.5	2.3	3.5	1.0
Scenario's					
1.1	0.3	8.5	2.3	3.5	1.00
1.2	1.0	8.5	2.3	3.5	1.02
1.3	3.0	8.5	2.3	3.5	1.06
1.4	10.0	8.5	2.3	3.5	1.18
1.5	30.0	8.5	2.3	3.5	1.36
2.1	0.5	4	2.3	3.5	0.04
2.2	0.5	5	2.3	3.5	0.10
2.3	0.5	6	2.3	3.5	0.14
2.4	0.5	7	2.3	3.5	0.16
2.5	0.5	8.5	2.3	3.5	1.00
3.1	0.5	8.5	0.1	3.5	0.96
3.2	0.5	8.5	0.3	3.5	0.97
3.3	0.5	8.5	1.0	3.5	0.97
3.4	0.5	8.5	1.5	3.5	0.98
3.5	0.5	8.5	2.3	3.5	1.00
4.1	0.5	8.5	2.3	1	0.54
4.2	0.5	8.5	2.3	3	1.00
4.3	0.5	8.5	2.3	10	6.2
4.4	0.5	8.5	2.3	15	13
4.5	0.5	8.5	2.3	25	37
Note: pH scenario at pH = 8.5 yields correction factor of 1.0					
There is a very strong pH dependant change around pH = 7 - 9					

correction factor for	CD				
normalised on DSW					
multiplier					
factor < 1: norm has to be lowered					
factor > 1: norm can be raised					
	salinity	pH	alkalinity	DOC	free
Dutch standard water	0.5	8.5	2.3	3.5	1.0
Scenario's					
1.1	0.3	8.5	2.3	3.5	1.0
1.2	1.0	8.5	2.3	3.5	2.8
1.3	3.0	8.5	2.3	3.5	8.2
1.4	10.0	8.5	2.3	3.5	36
1.5	30.0	8.5	2.3	3.5	173
2.1	0.5	4	2.3	3.5	0.75
2.2	0.5	5	2.3	3.5	0.76
2.3	0.5	6	2.3	3.5	0.78
2.4	0.5	7	2.3	3.5	0.78
2.5	0.5	8.5	2.3	3.5	1.00
3.1	0.5	8.5	0.1	3.5	0.99
3.2	0.5	8.5	0.3	3.5	0.99
3.3	0.5	8.5	1.0	3.5	0.99
3.4	0.5	8.5	1.5	3.5	1.00
3.5	0.5	8.5	2.3	3.5	1.00
4.1	0.5	8.5	2.3	1	0.92
4.2	0.5	8.5	2.3	3	1.00
4.3	0.5	8.5	2.3	10	1.40
4.4	0.5	8.5	2.3	15	1.64
4.5	0.5	8.5	2.3	25	2.20

correction factor for normalised on DSW	**NI**				
multiplier					
factor < 1: norm has to be lowered					
factor > 1: norm can be raised					
	salinity	**pH**	**alkalinity**	**DOC**	**free**
Dutch standard water	0.5	8.5	2.3	3.5	**1.0**
Scenario's					
1.1	**0.3**	**8.5**	**2.3**	**3.5**	**1.00**
1.2	1.0	8.5	2.3	3.5	**1.04**
1.3	3.0	8.5	2.3	3.5	**1.14**
1.4	10.0	8.5	2.3	3.5	**1.44**
1.5	30.0	8.5	2.3	3.5	**2.56**
2.1	0.5	4	2.3	3.5	**0.98**
2.2	0.5	5	2.3	3.5	**0.99**
2.3	0.5	6	2.3	3.5	**0.99**
2.4	0.5	7	2.3	3.5	**0.99**
2.5	**0.5**	**8.5**	**2.3**	**3.5**	**1.00**
3.1	0.5	8.5	0.1	3.5	**0.96**
3.2	0.5	8.5	0.3	3.5	**0.96**
3.3	0.5	8.5	1.0	3.5	**0.97**
3.4	0.5	8.5	1.5	3.5	**0.98**
3.5	**0.5**	**8.5**	**2.3**	**3.5**	**1.00**
4.1	0.5	8.5	2.3	1	**0.99**
4.2	**0.5**	**8.5**	**2.3**	**3**	**1.00**
4.3	0.5	8.5	2.3	10	**1.08**
4.4	0.5	8.5	2.3	15	**1.19**
4.5	0.5	8.5	2.3	25	**1.58**

correction factor for normalised on DSW	**PB**				
multiplier					
factor < 1: norm has to be lowered					
factor > 1: norm can be raised					
	salinity	**pH**	**alkalinity**	**DOC**	**free**
Dutch standard water	0.5	8.5	2.3	3.5	**1.0**
Scenario's					
1.1	**0.3**	**8.5**	**2.3**	**3.5**	**1.00**
1.2	1.0	8.5	2.3	3.5	**1.03**
1.3	3.0	8.5	2.3	3.5	**1.12**
1.4	10.0	8.5	2.3	3.5	**1.46**
1.5	30.0	8.5	2.3	3.5	**2.76**
2.1	0.5	4	2.3	3.5	**0.58**
2.2	0.5	5	2.3	3.5	**0.59**
2.3	0.5	6	2.3	3.5	**0.64**
2.4	0.5	7	2.3	3.5	**0.68**
2.5	**0.5**	**8.5**	**2.3**	**3.5**	**1.00**
3.1	0.5	8.5	0.1	3.5	**0.93**
3.2	0.5	8.5	0.3	3.5	**0.93**
3.3	0.5	8.5	1.0	3.5	**0.94**
3.4	0.5	8.5	1.5	3.5	**0.96**
3.5	**0.5**	**8.5**	**2.3**	**3.5**	**1.00**
4.1	0.5	8.5	2.3	1	**0.96**
4.2	**0.5**	**8.5**	**2.3**	**3**	**1.00**
4.3	0.5	8.5	2.3	10	**1.19**
4.4	0.5	8.5	2.3	15	**1.30**
4.5	0.5	8.5	2.3	25	**1.50**

Note: pH scenario at pH = 8.5 yields correction factor of 1.0
There is a very strong pH dependant change around pH = 7 - 9

Incorporating a model to calculate the available fraction of metals into OMEGA looks promising since the model can explain most of the observed different risk levels for metals in different water systems (like coastal water, Rhine river water or acid lakes).

3.1.3 Heavy metals, Sediment toxicity

In sediments, uptake of toxic components can be largely correlated to the pore water concentration Di Toro *et al.*, 2000) by direct contact with the skin of organisms. While this approach can be refined by taking into account the oral and dermal uptake route for food digestion (Vijver, 2005), the mechanism of pore water exposure as an important process for bioaccumulation is widely accepted.

When the relation between sediment concentrations, sediment conditions and the sediment pore-water concentrations have been derived, OMEGA can calculate the PAF based on the sediment pore water concentration. This is normally done by:

1. Direct pore water measurement.
2. Equilibrium partitioning based on organic matter for organic compounds.
3. Equilibrium partitioning based on organic matter and clay content of the sediment for heavy metals.

3.2 Algorithms for organic pollutants

OMEGA produces a result of potentially effective fraction (PAF) of species in a given ecosystem as a function of chemical concentrations. This response is based on species sensitivity distributions (SSDs) created through ecotoxicological tests with chemicals in aquatic solutions. Thus, the resulting PAF relates to dissolved concentrations of chemicals. However, existing monitoring data as well as future monitoring data under the water framework directive (WFD) may consist not only in the form of dissolved concentrations of chemicals but also in the form of total water concentrations, suspended particulate concentrations or even data from sediment and biota monitoring. Nevertheless, there is evidence that mainly the dissolved part of organic micro-pollutants matter for biological toxicity. Moreover, the SSDs used as the basis in OMEGA are calibrated to dissolved concentrations and the input data for OMEGA should consequently be always dissolved concentrations.

Therefore, algorithms are needed for transforming all media monitoring data into a corresponding dissolved concentration of organic chemicals. Minimum input needs for accompanying monitoring data for such calculations are defined and some default values suggested.

3.2.1 Transferring all media water monitoring data into dissolved concentrations

Organic pollutants with polar or even ionic character are generally measured in the liquid phase with or without separation of the particulate phase. Due to those substances' low sorption to the particulate matter these pollutant data can be usually considered as dissolved concentrations for the purpose of OMEGA and data input performed directly. Thus, the following considerations and assumptions mainly regard hydrophobic organic pollutants, which are frequently measured in suspended particulate matter (e.g., separated by filtration, centrifugation etc.) or whole water (e.g. liquid-liquid extraction or combination of separate liquid and solid extracts for single measurement). Considerable advances have taken place in hydrophobic organic pollutants' multimedia model development in conjunction with improvements in computer architectures over the last two decades.

Nevertheless, the increased complexity of the model and the improved fidelity of the calculations increase the requirements for input data and reduce the possible application to well-defined and well-studied areas. In contrast, accompanying information together with available monitoring data for use in the generic OMEGA model may be limited if not absent. Therefore, simple equilibrium partition theory with the assumptions of steady state and an infinite number of equal energy sorption sites in a closed system requiring only few additional parameters from the system was selected here to propose algorithms for obtaining dissolved concentration (CW) from water monitoring data delivering whole water (liquid + particulate phases) concentrations (CT) , SPM concentration (CS), or concentration in the liquid phase (i.e., without SPM) (CW), respectively, and define minimum input needs. Finally, among the sorbents present in the aquatic environment, organic matter (and of organic matter the organic carbon, which makes up approximately 50% and is usually easily determined by combustion) plays the most significant role for sorption of hydrophobic organic chemicals. Hence, the partitioning of hydrophobic pollutants between the liquid phase and the organic carbon of the solid matter presents the basis for the following calculations.

Minimum input needs:

Octanol-water partition coefficient (K_{OW}) or organic carbon normalized sorption coefficient (K_{OC}) of the chemical substance

- Total suspended matter (TSM) concentration in the water body and organic carbon fraction (f_{OC}) of the suspended matter
- C_T or C_S (or C_{SPM}) or C_W

Algorithms:

Non planar compounds: $\log K_{OC} = 0.74 \log K_{OW} + 0.15$
Planar compounds: $\log K_{OC} = 0.98 \log K_{OW} -0.32$
$K_d = C_S/C_W$ and $K_{OC} = K_d/f_{OC} = C_S/(C_W*f_{OC})$
$C_T = C_S + C_W$ and also $C_W = f_W*C_T = C_S/(K_{OC}*f_{OC})$ as well as $C_S = f_S*C_T = C_W*K_{OC}*f_{OC}$
Cs [µg /L] = C_{SPM} [µg/kg] * TSM [kg/L]
$f_W = 1/(1 + TSM*K_d)$ and $f_S = TSM*K_d/(1 + TSM*K_d)$; of course: $f_W + f_S = 1$

Examples:

1.) Monitoring data:
C_T of Chemical X (K_{OW} = 10 000 000) is 10 µg/L; TSM = 10 mg/L (= 0.00001 kg/L) and $f_{OC} = 0.1$
$\log K_{OC} = 0.74 \log (10\,000\,000) + 0.15 = 5.33 \rightarrow K_{OC} = 213796$
Therefore: $f_W = 1/(1 + TSM*f_{OC}*K_{OC}) \rightarrow f_W = 1/(1 + 0.00001*0.1*213796) = 0.824$
$\Rightarrow C_W = f_W*C_T \rightarrow C_W$ = 0.824*10 µg/L = **8.24 µg/L (input for OMEGA)**

2.) Monitoring data:
C_{SPM} of Chemical Y (K_{OW} = 3 581 410) is 200 mg/kg and $f_{OC} = 0.15$
$\log K_{OC} = 0.74 \log (3\,581\,410) + 0.15 \approx 5 \rightarrow K_{OC} = 100\,000$
With $C_W = C_S/(K_{OC}*f_{OC})$
$\rightarrow C_W$ = 200/(100000*0.15) = 0.0133 mg/L = **13.33 µg/L (input for OMEGA)**

3.) Monitoring data: C_W of Chemical Z is **3 µg/L ⇒ direct input for OMEGA**

Default values if not obtained from monitoring data:

In contrast to K_{OW} or even directly K_{OC} of chemical substances that are usually available from tables or public references, TSM and f_{OC} have to be determined analytically together with the chemical concentrations. The measurement of total suspended matter (TSM) concentration according to the European Standard EN 872:2005 is a simple method by filtration through glass fibre filters followed by gravimetry. Organic carbon can be determined by dry combustion following ISO 10694:1995. An attempt is made here to propose some kind of “default values” that may be used in absence of TSM and f_{OC} so that calculations can be conducted. Nevertheless, the uncertainty of the estimated dissolved concentration applying above proposed algorithms without these parameters measured accurately increases significantly, which should be made aware to the user of OMEGA or of these algorithms, respectively.

TSM:

Lakes:	Oligotrophic	→	0.5 to 2 mg/L
	Mesotrophic	→	1 to 5 mg/L
	Eutrophic	→	3 to 10 mg/L

Rivers: Low flow → 10 to 50 mg/L
High flow → 50 to 500 mg/L

Estuaries 10 to 500 mg/L

Lagoons 10 to 100 mg/L

Coastal waters 0.5 to 10 mg/L

f_{OC}: 0.1

3.2.2 Transferring sediment monitoring data into dissolved (pore water) concentrations

The same considerations as for water monitoring data principally apply also to sediment monitoring data. However, pore water concentrations, which matter principally for benthic organisms as bioavailable fraction and are needed as OMEGA input is only rarely measured directly even for less hydrophobic to polar chemicals. In most cases and for most chemicals sediment monitoring data is available in concentrations of pollutants in dry sediment. Thus, simple equilibrium partition theory with the assumptions described above is used here as well for obtaining dissolved concentration in pore water ($C_{pore\ water}$) from sediment monitoring data delivering sediment concentrations (C_{Sed}). However, in contrast to suspended particulate matter in surface water (TSM = 1 to 100mg/L), where hydrophobic adsorption sites (e.g., black carbon) for planar hydrophobic pollutants may be more easily saturated by the chemical mixture present and therefore be ignored, neglecting the adsorption of planar molecules to black carbon in sediments (TSM = r_{SW} = 3 to 10 kg/L) is likely to yield a significant overestimation of pore water concentrations derived from sediment concentrations (Schwarzenbach *et al*., 2003). Consequently, additional algorithms are proposed.

Minimum input needs:

- K_{OW} or K_{OC} of the chemical substance
- Organic carbon fraction f_{OC} and eventually black carbon fraction f_{bC} of the sediment
- C_{Sed}

Algorithms:

Non planar compounds: $\log K_{OC} = 0.74 \log K_{OW} + 0.15$
Planar compounds: $\log K_{OC} = 0.98 \log K_{OW} -0.32$
Planar compounds: $\log K_{bC} = 1.6 \log K_{OW} -1.4$
$K_d = C_S/C_W$ and $K_{OC} = K_d/f_{OC} = C_S/(C_W*f_{OC})$ as well as $K_{bC} = K_d/f_{bC} = C_S/(C_W*f_{bC})$
From $C_S = K_{OC}*f_{OC}*C_W$ for non-planar compounds and $C_S = K_{OC}*f_{OC}*C_W + K_{bC}*f_{cC}*(C_W)^{0.7}$ for planar compounds considering also adsorption to hydrophobic surfaces like black carbon, pore water concentrations can be derived as follows:
$C_{pore\ water} = C_S/(K_{OC}*f_{OC})$ for non-planar compounds and $C_{pore\ water} = C_S/(K_{OC}*f_{OC} + K_{bC}*f_{cC}*(C_{pore\ water})^{-0.3})$ for planar compounds (the latter equation can only be solved iteratively by "trial and error"!)

Examples:

1.) Monitoring data:
C_{Sed} of **<u>non-planar</u>** Chemical X (K_{OW} = 1 000 000) is 50 mg/kg (50000 µg/kg) and f_{OC} = 0.03
$\log K_{OC} = 0.74 \log (1\ 000\ 000) + 0.15 = 4.59 \rightarrow K_{OC} = 38904.5$
With $C_{pore\ water} = C_S/(K_{OC}*f_{OC})$
→ $C_{pore\ water}$ = 50000/(38904.5*0.03) = **42.8 µg/L (input for OMEGA)**

2.) Monitoring data:

C_{Sed} of **planar** (e.g. PAHs) Chemical Y (K_{OW} = 1 000 000) is 50 mg/kg (50000 µg/kg), f_{OC} = 0.03 and f_{bC} = 0.05*f_{OC} = 0.0015
log K_{OC} = 0.98 log (1 000 000) -0.32 = 5.56 → K_{OC} = 363 078
log K_{bC} = 1.6 log (1 000 000) -1.4 = 8.2 → K_{bC} = 158 489 319
With $C_{pore\ water} = C_S/(K_{OC}*f_{OC} + K_{bC}*f_{cC}*(C_{pore\ water})^{-0.3})$
→ $C_{pore\ water}$ = 50000/(363 078*0.03 + 158 489 319*0.0015*$(C_{pore\ water})^{-0.3}$)
→ $C_{pore\ water}$ derived by trial and error: ≈ **0.1 µg/L (input for OMEGA)**

Default values if not obtained from monitoring data:

f_{OC}: 0.03 (0.01 to 0.05)

f_{bC}:= 0.05* f_{OC} (0.01 to 0.1* f_{OC}) ⇒ **f_{bC} = 0.0015**

4. The use of OMEGA in a Western Scheldt case study

4.1 Introduction

The Western Scheldt is a Dutch/Belgium Estuary, Figure 4.1.

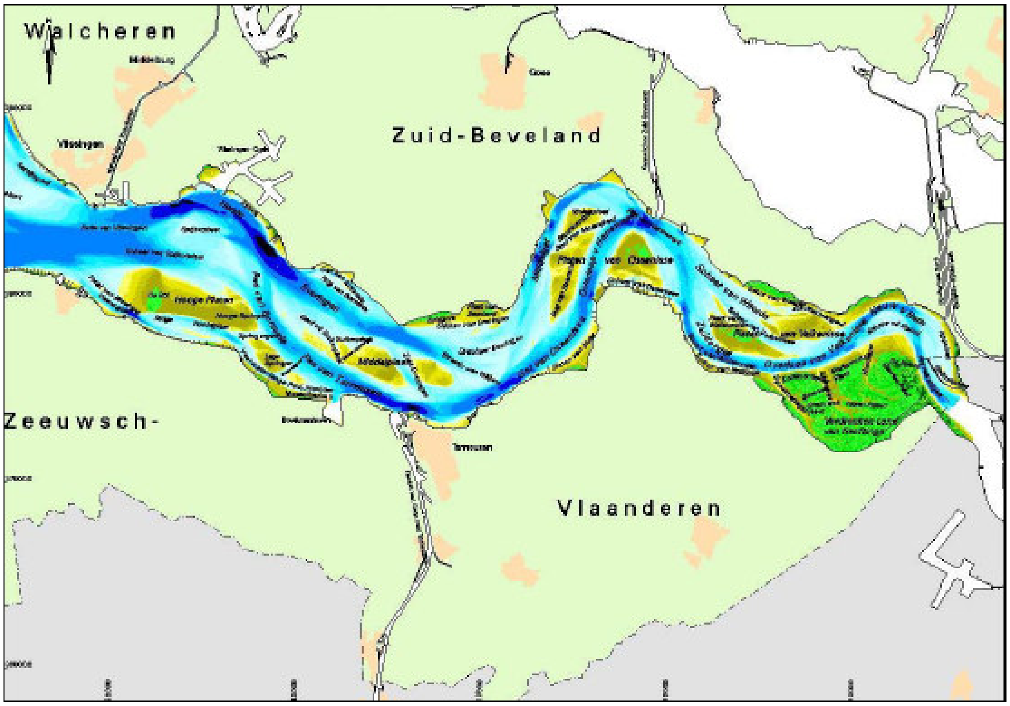


Figure 4.1 Western Scheldt Estuary

The Western Scheldt is an important ecological system and the estuary has a special protected status within the EU Habitat Conservation and Protection Guidelines, Bird directive. This places an extra focus on the chemical quality status of the estuary. The estuary has also a high economic value since it functions as the entrance to the harbour of Antwerp (Figure 4.2). Especially since the large scale flooding in Zealand in 1953 (Figure 4.3), flood risk management of the Western Scheldt is also an important factor in the management of the estuary.

Figure 4.2 Sea ship in the Western Scheldt on its route to Antwerp

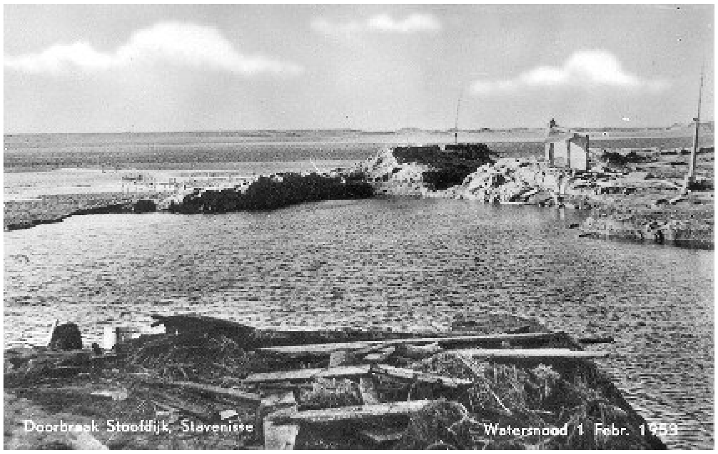


Figure 4.3 Flooding of 1953

The combination of these factors and the dedicate balance in the morphological conditions of the estuary (which have an impact on the channel system) make it very complex to change any of the specific functions (like channel deepening for the entrance of bigger sea ships) without impacting the other functions.

From a water quality point of view, the estuary has seen a reduction in the load of pollutants over the last decades. However, historical polluted sites within the estuary (like small harbours) and the discharge of pollutants from the Belgium river Scheldt / Rupel (by the end of 2006 a new waste water treatment plant for Brussels will impact the pollutant load from the river Scheldt) still have an influence on the chemical status of the estuary. The question which has been addressed in this case study is if the chemical status can be translated to a toxic risk.

4.2 Materials and methods

4.2.1 Chemical status

The Western Scheldt has an intensive monitoring network for water quantity parameters, Figure 4.4.

Unfortunately, only for Schaar van Ouden Doel (in the East near Antwerp) a complete record on a two week measurement frequency is present for the dissolved concentrations of organic and heavy metal pollutants (Figures 4.5 and 4.6). For this location the heavy metals Zinc and Copper are the dominant pollutants, but other pollutants have also been taken into account:

- Cd
- Cu
- Zn
- Naphtalene
- HCB
- α,β,γ, HCH

Data from the year 2000 is used.

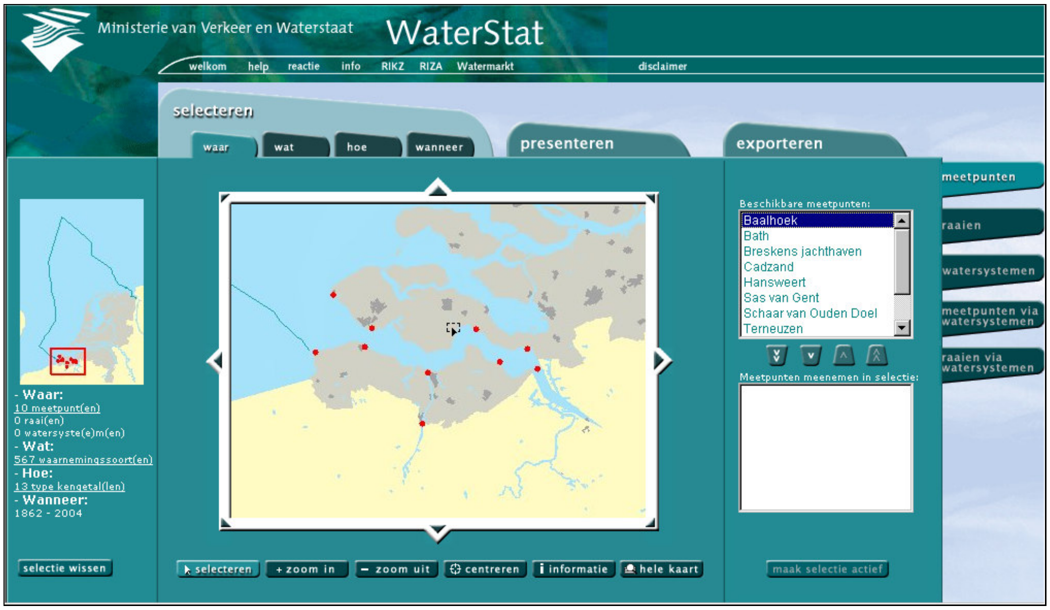


Figure 4.4 Monitoring points water quantity and quality Western Scheldt

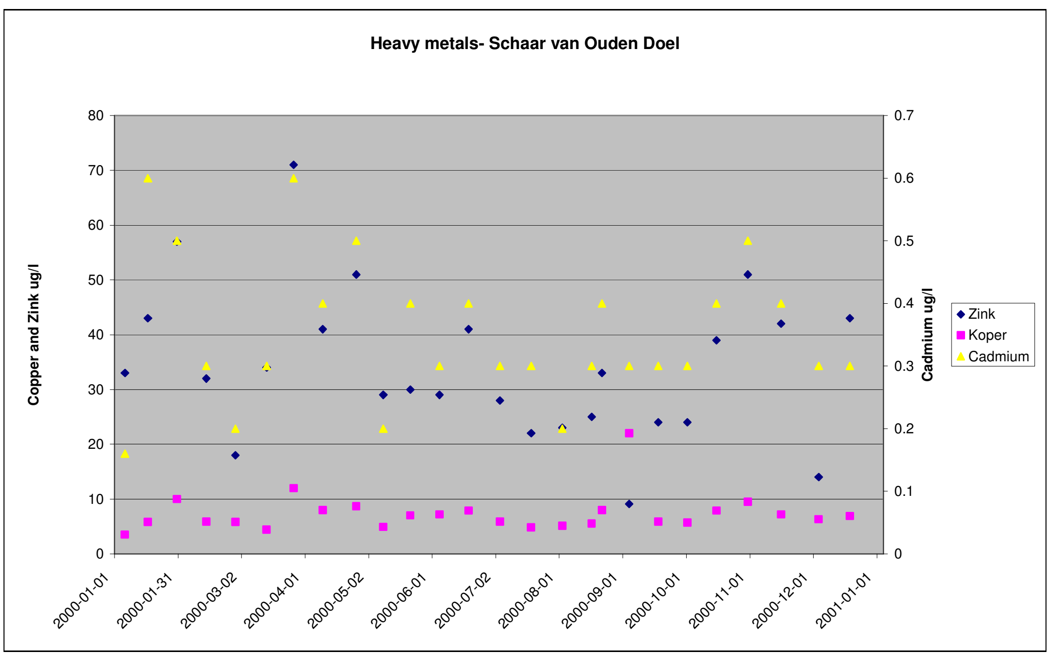


Figure 4.5 Trends in heavy metal concentrations at Schaar van Ouden Doel (2000)

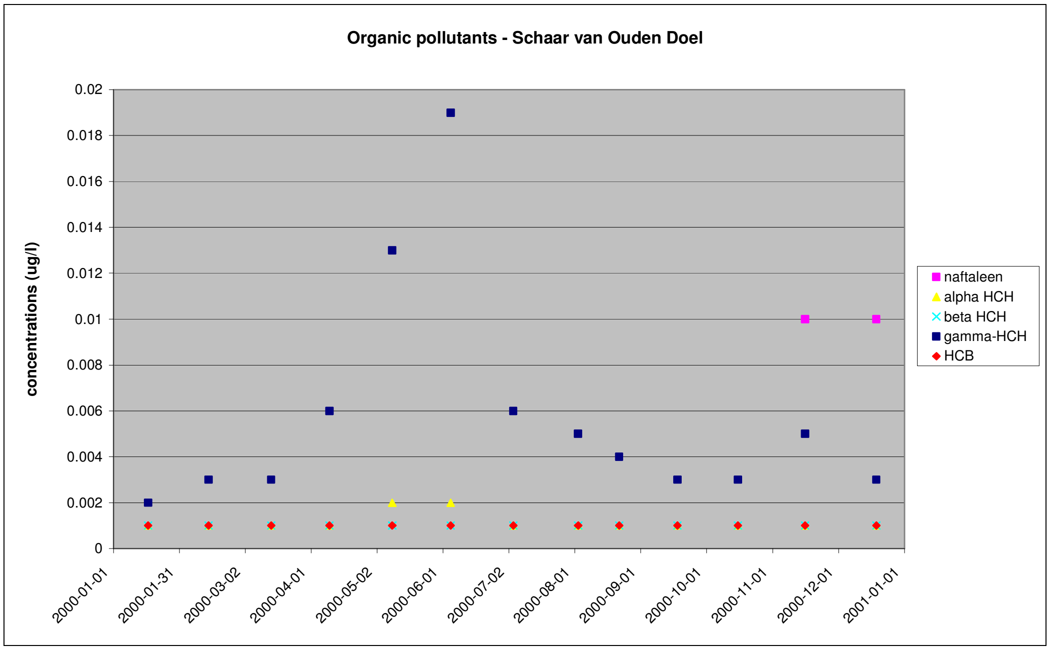


Figure 4.6 Trends in organic pollutants concentrations at Schaar van Ouden Doel (2000)

4.2.2 Sobek 1D/2D

The Sobek 1D/2D model was based on the national grid for the Dutch River systems, taking into account the bathometry of the Western Scheldt, the discharge on the river Scheldt and the tidal influence of the North Sea Boundary, Figure 4.7.

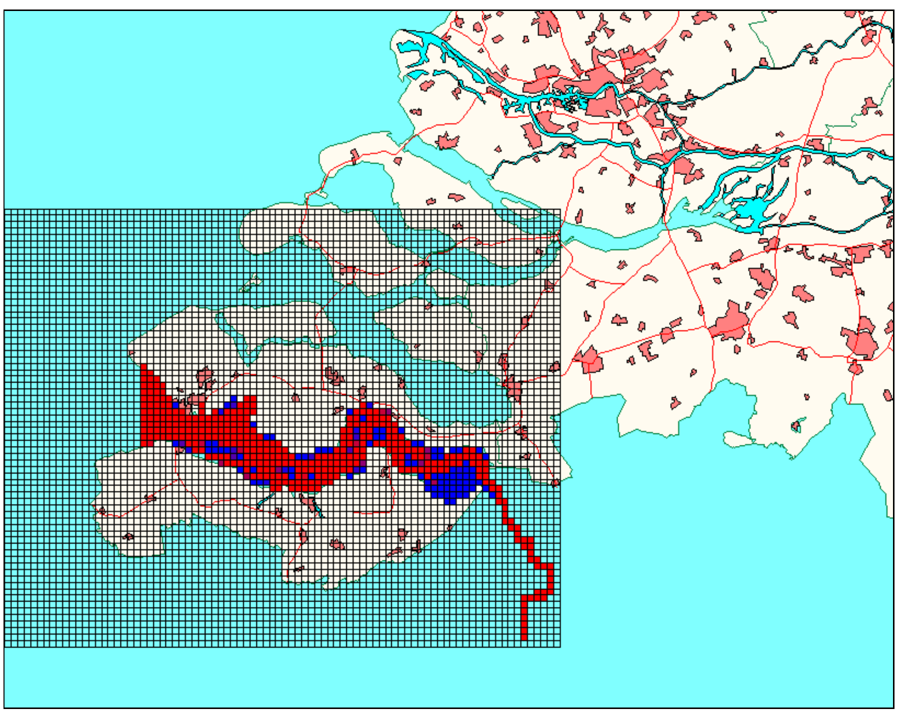


Figure 4.7 Modelled Western Scheldt area

The model is a 1D channel model, in which the second dimension is added to include the flow outside the main channels, Figure 4.8.

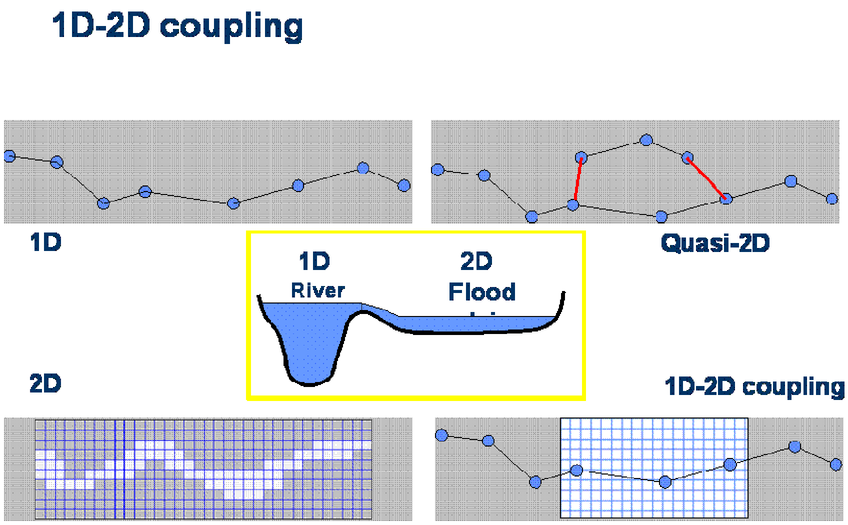


Figure 4.8 Principle of Sobek 1D/2D coupling

After the calculation of the hydrological conditions, the water quality is calculated with the water quality model Delwaq. By taking the North Sea concentration as the Western boundary of the system (Schaar van Ouden Doel is the East boundary), an east-west concentration profile can be calculated as function of the water quantity. For this, the hydrological model Sobek 1D/2D has been used in combination with the chemical model Delwaq.

4.2.3 Ecotoxicological stress

The calculated concentrations for Copper and Zinc are then used in the OMEGA model to calculate the ms-PAF and the contribution of each model to the ms-PAF, Figure 4.9. Since the Sobek 1D/2D model calculates the total dissolved concentration, the output of this model could be used as direct input for OMEGA.

The variation between the different organism groups and the total population was also calculated. Groups of organisms in OMEGA are:

- Fish
- Benthic Invertibrates
- Phyto Plankton
- Macrophytes

The number of species and abundance of the population is dependant on the specific ecotope, Figure 4.10. The ecotope in the Western Scheldt estuary is not constant. Main factors determining the ecotope are:

- Salinity
- Mud content of the sediment
- The inter-tidal zone (sub-littoral, mid-low-littoral and upper-littoral)
- Water depth and flow velocity

Within an ecotope, a certain species diversity, species density and species biomass is expected. This defines the good ecological quality status.

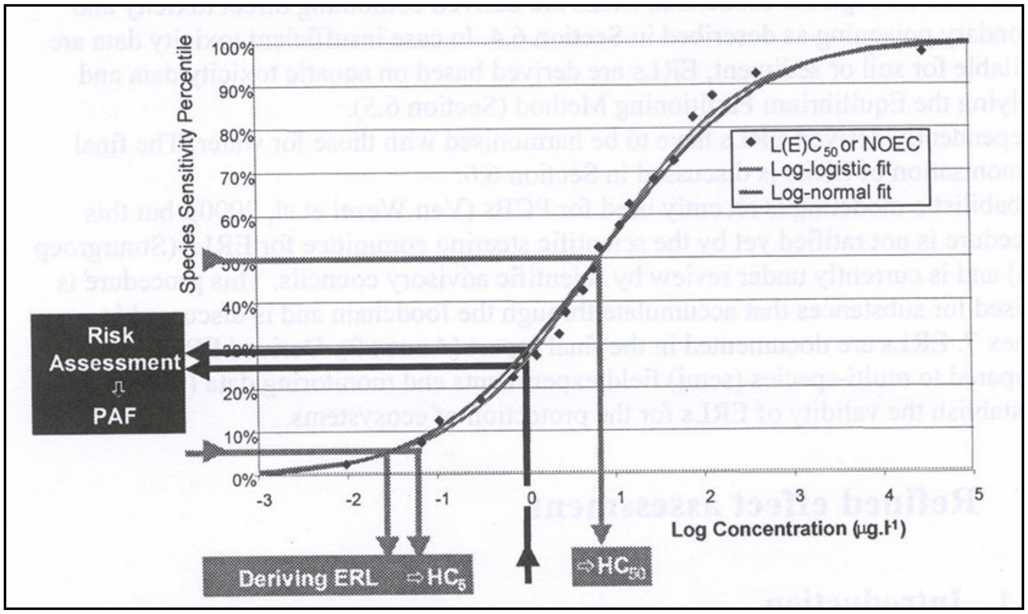


Figure 4.9 The method behind OMEGA: from NOEC's to Potential Affected Fraction (PAF)

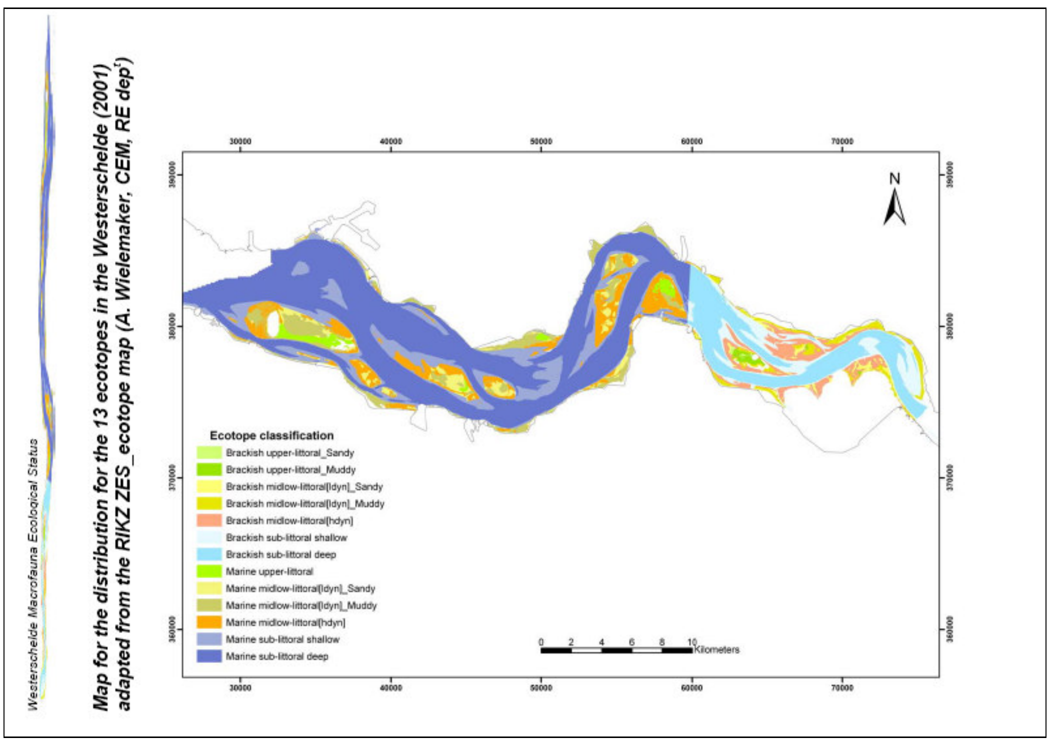


Figure 4.10 Ecotopes for the Western Scheldt (from MEP/GEP WESTERSCHELDE, appendix 1)

4.3 Results and discussion

4.3.1 Chemical status

The Sobek 1D/2D model was calibrated for a period of one year (2000).

- Number of chemical observations of water quality on the boundaries: 26 observations

- Partitioning between dissolved and suspended sediment of pollutants
- Hydrological time step conditions: 2 hours

The result was compared with the measured concentration, Figure 4.11.

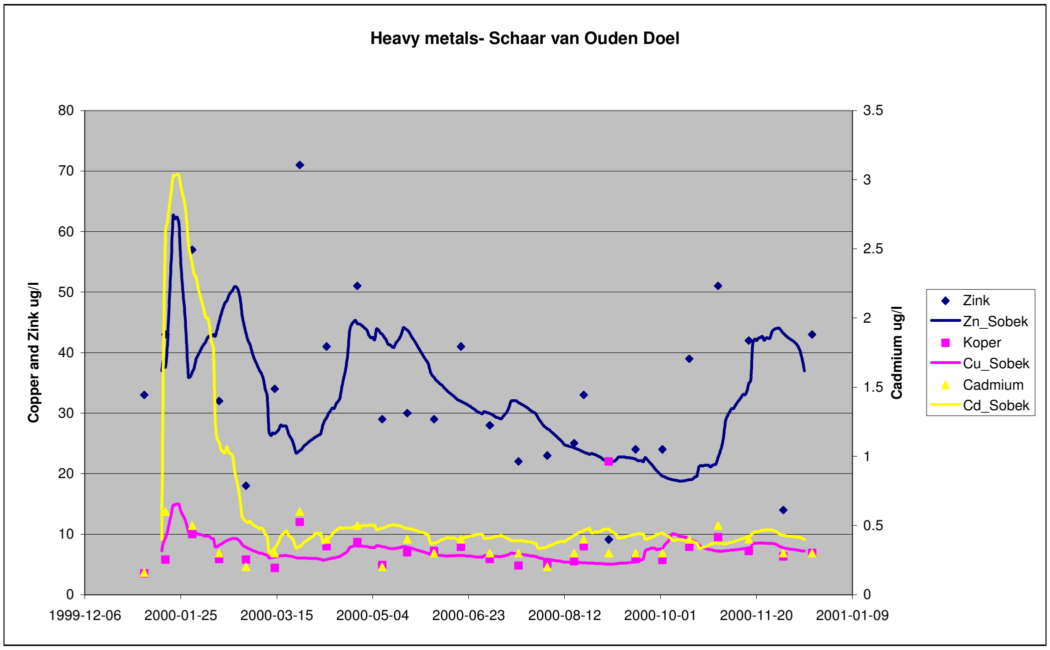


Figure 4.11 Measured (dots) and calculated concentrations (lines) at eastern calibration point (Schaar van Ouden Doel)

The tidal influence and difference in load on the boundaries translates into a time dependant concentration of pollutants, Figure 4.12.

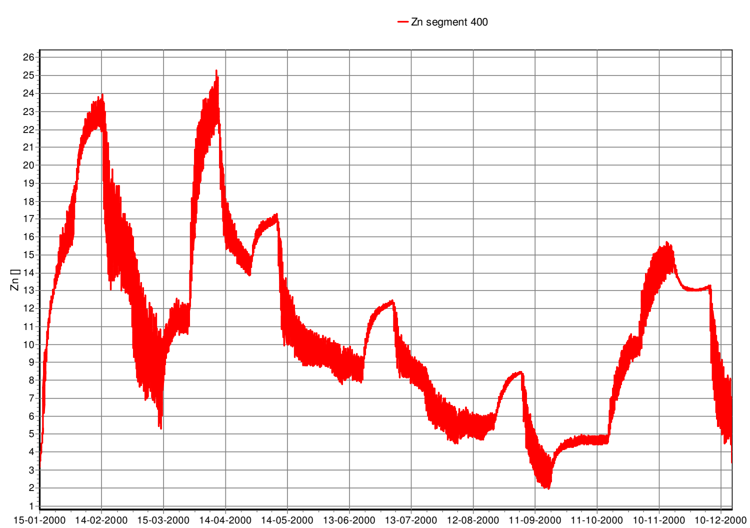


Figure 4.12 Example of zinc concentration in the middle of the Western Scheldt during one year (2000), including tidal fluctuation

Besides the tidal influence, the pollutant concentrations also vary during the year and as a function of place, Figures 4.13 to 4.17.

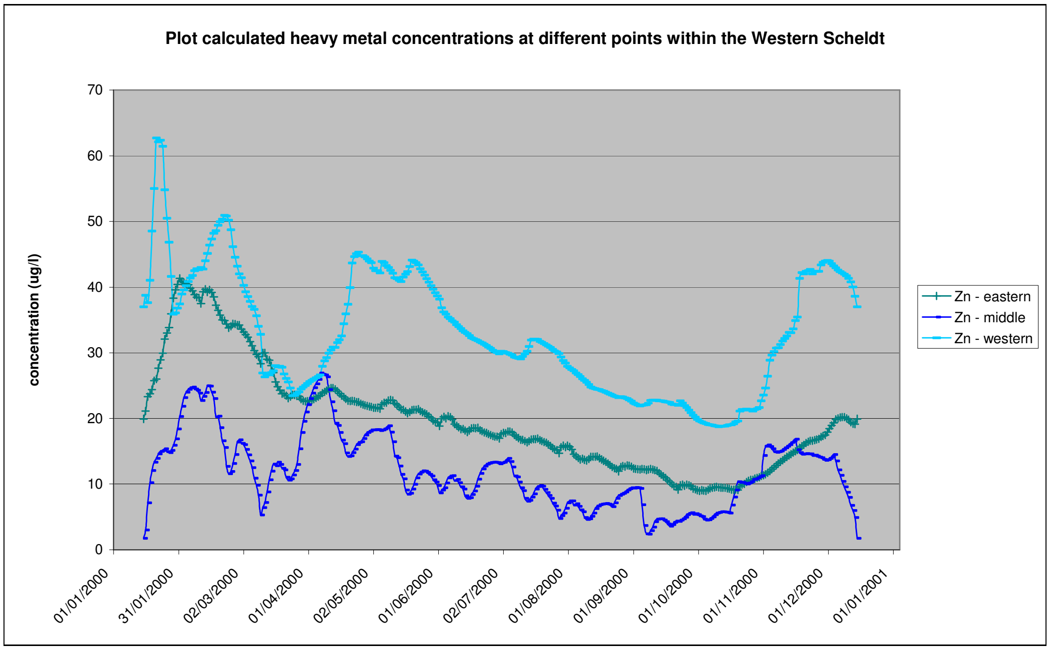


Figure 4.13 Example of zinc during the year and as function of the place

Eastern part	= Schaar van Ouden Doel, near Antwerp
Middle part	= Terneuzen
Western part	= Vlissingen, near North Sea

The total dissolved concentration of pollutants in the Western Scheldt can be plotted as a concentration contour map at different time intervals:

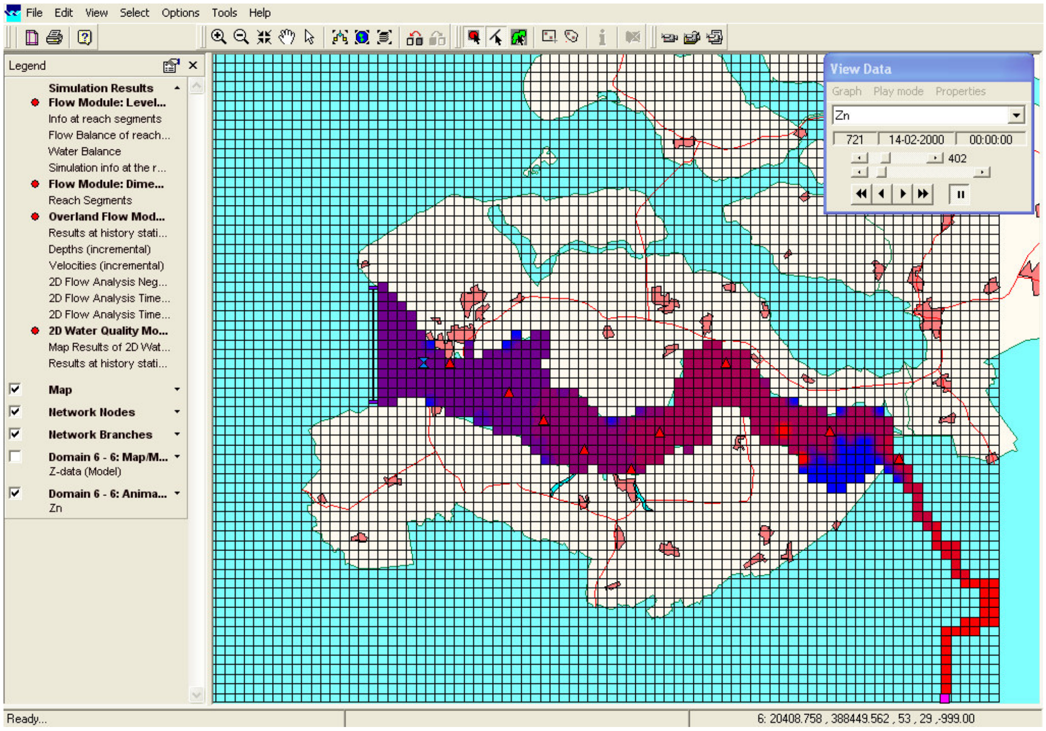


Figure 4.14 Zinc concentrations on 14-02-2000 (spring peak)

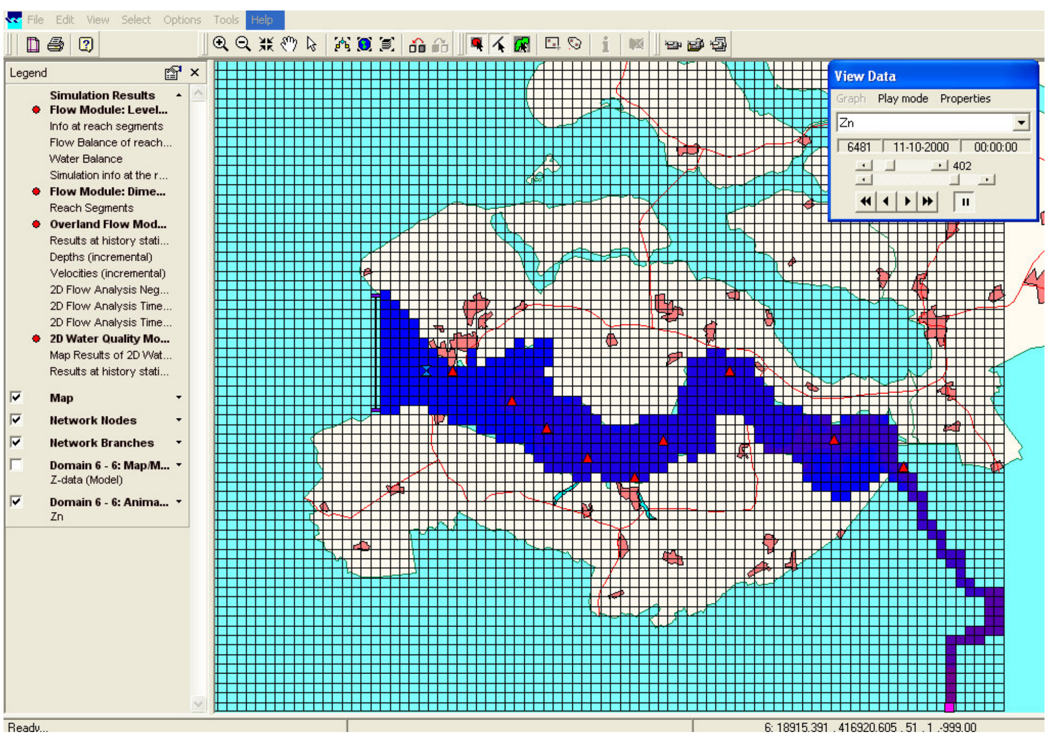


Figure 4.15 Zinc concentrations on 11-10-2000 (autumn low)

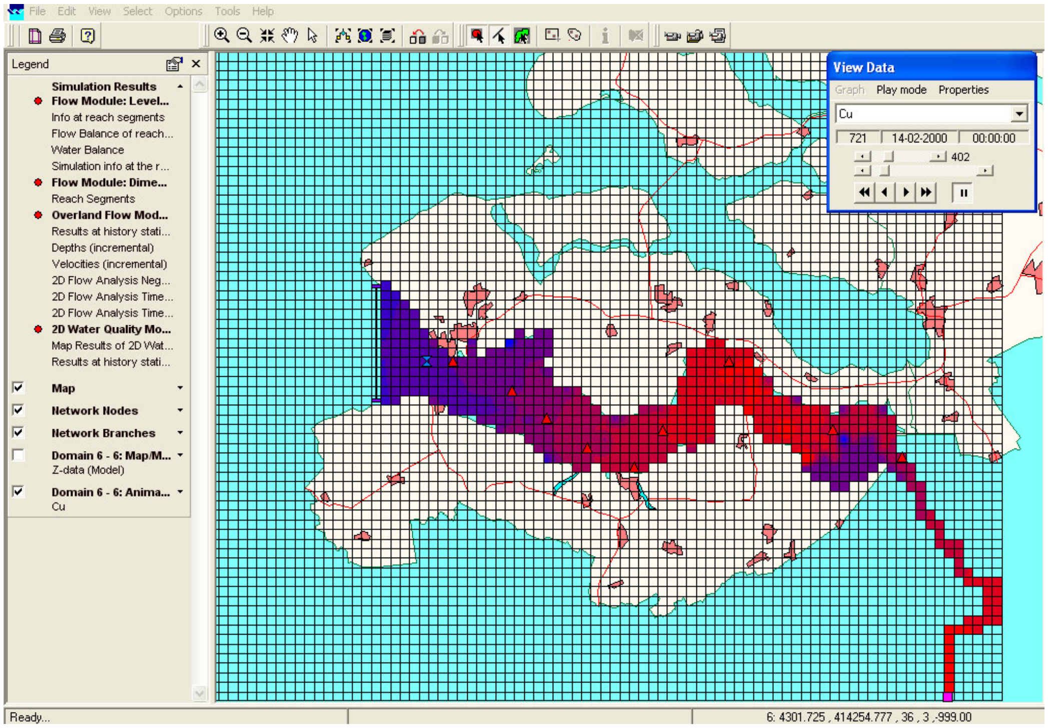


Figure 4.16 Copper concentrations on 14-02-2000 (spring peak)

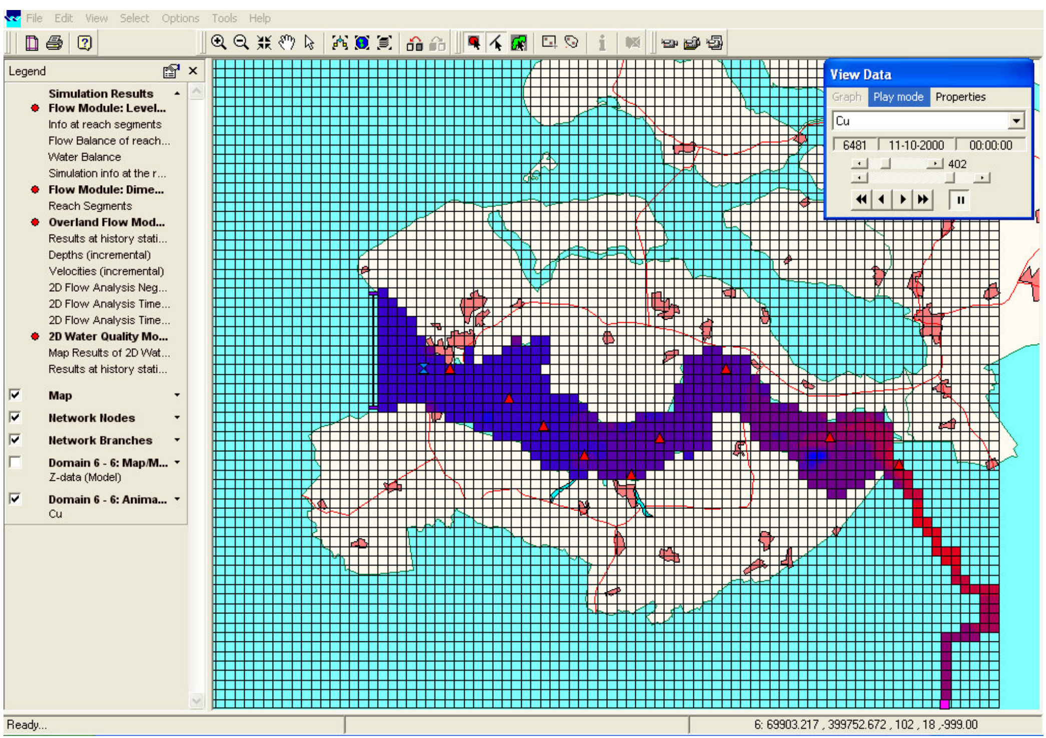


Figure 4.17 Copper concentrations on 11-10-2000 (autumn low)

4.3.2 Ecotoxicological stress

The ms-PAF has been calculated based on the water quality for an area in the South-West part of the Western Scheldt, the nature protection zone 'Verdronken Land van Saeftinghe', Figure 4.18.

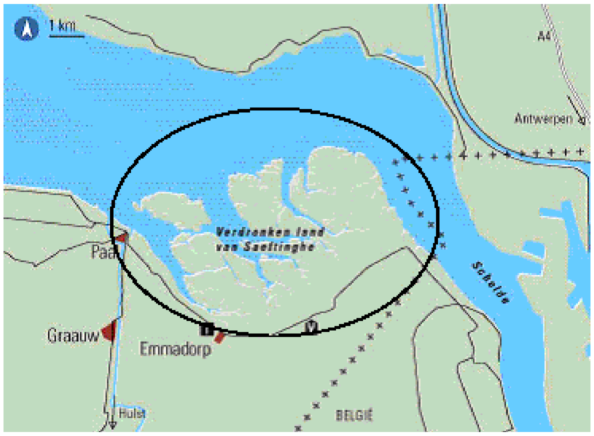


Figure 4.18 Verdronken Land van Saeftinghe'

The ms-PAF has been calculated with OMEGA based on the dissolved concentrations of Cd, Zn, Cu, (α-β-γ)HCH, HCB and naphtalene, Figure 4.19.

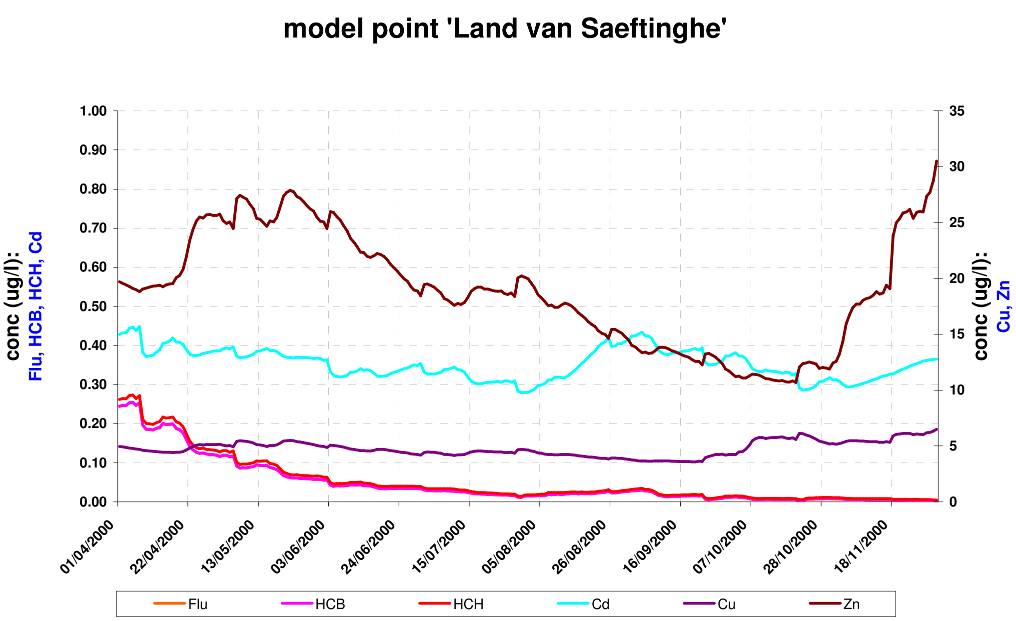


Figure 4.19 Dissolved concentrations for reference point 'Verdronken Land van Saeftinghe'

The ms-PAF is based on observed chronic stress. No additional relations (like QSAR or acute stress data) were used to fill missing data gaps, Table 4.1.

Table 4.1 Mode of action for each pollutant

pollutant	mode of action nr	mode of action
cadmium	7	Cadmium-specific
copper	10	Copper specific
HCH	24	Neurotoxicant: cyclodiene-type
fluorantheen	29	Nonpolar narcosis
HCB	29	Nonpolar narcosis
Zink	44	Zinc-specific

The ms-PAF for chronic exposure was calculated for each of the species groups: Phytoplankton, Macrophytes, Phytobenthos, Benthic Invertebrates, Fish and the Total ms-PAF, Figure 4.20.

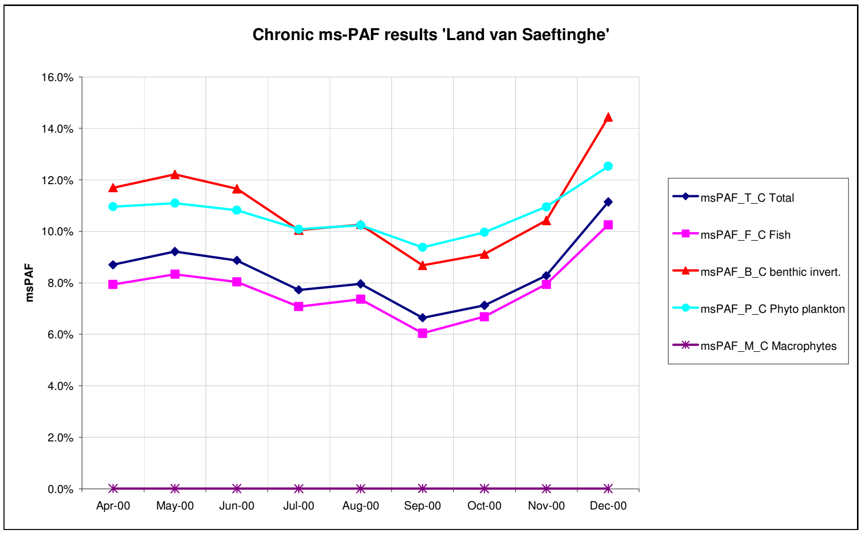
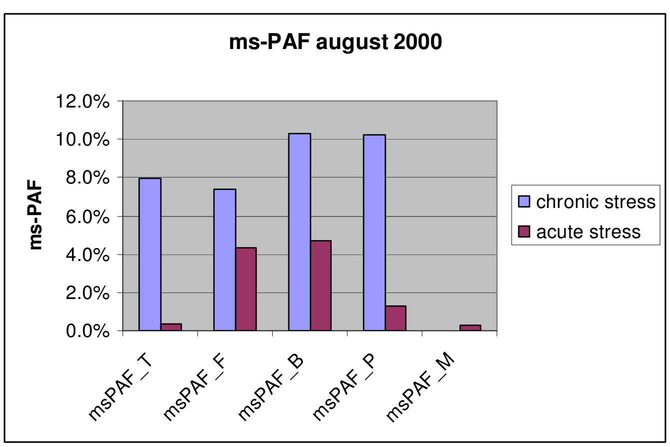


Figure 4.20 ms-PAF for chronic exposure for different groups of organisms and total-ms PAF, location Land van Saeftinghe

Toxic stress is higher for phytoplankton and benthic invertebrates (an average ms-PAF of 10%), while for fish the ms-PAF is a bit lower (around 7%), Figure 4.21. For macrophytes insufficient data is present. The overall ms-PAF for chronic exposure is around 8%. A check of the ms-PAF for acute stress in a summer month (august 2000) yields that the acute stress level is lower then the chronic stress level.



msPAF_T = Total
msPAF_F = Fish
msPAF_B = Benthic
msPAF_P = Phyto Plankton
msPAF_M = Macrophytes

Figure 4.21 Comparison of chronic versus acute stress, August 2000

The ms-PAF can be split to the contribution for each single toxicant. For this, the chronic total ms-PAF has been used. The break-down on the contribution of single toxicants has been done on 01-04-2006 and has been repeated for every two months, Figure 4.22.

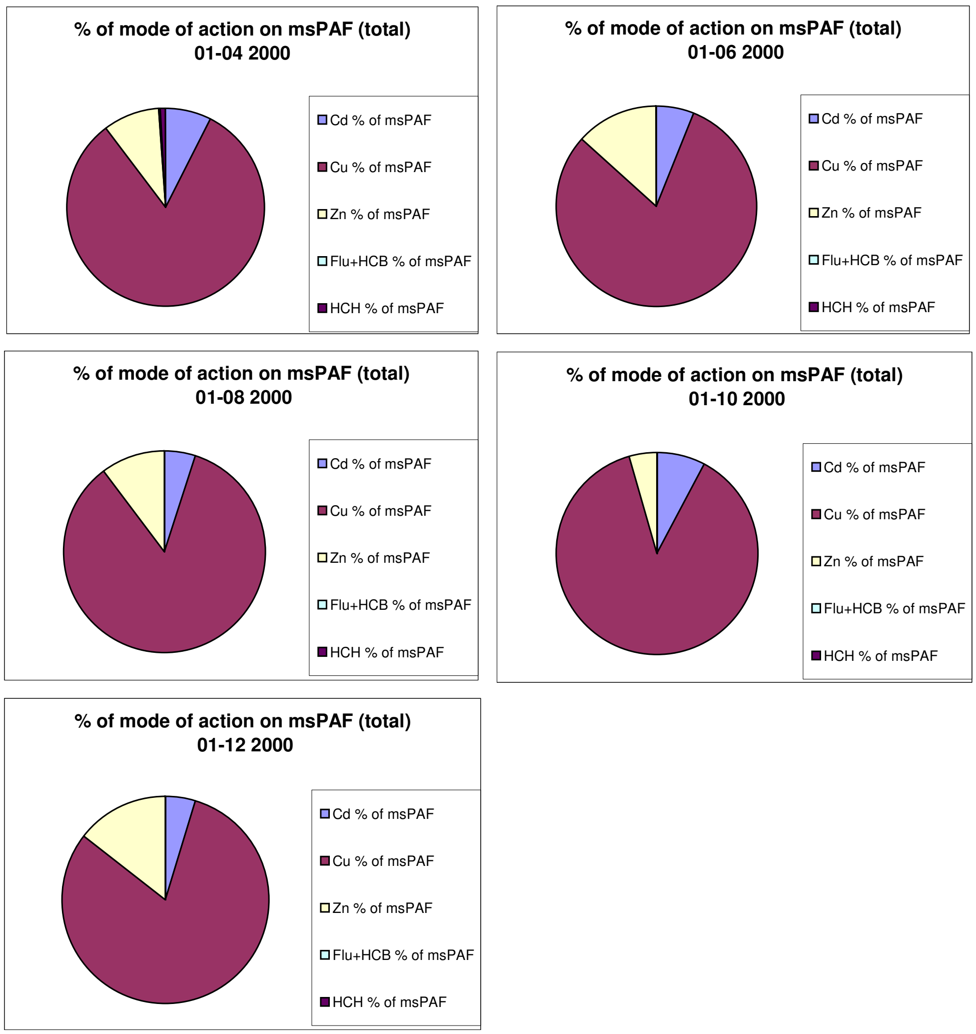


Figure 4.22 Contribution of single toxicants to the ms-PAF for chronic exposure at different moments in time, location Land van Saeftinghe

The main contribution to the ms-PAF comes from copper (80% to 90%), followed by zinc (4% to 14%).

4.3.3 Correlation of the calculated ecotoxicological stress with the observed ecological quality index

For the validation, if the ms-PAF levels are reflected in the ecological status of the system, the ecological status map for the Western Scheldt from NIOO-CEMO has been used, Figure 4.23.

Overview of the indicators and sub-indicators with their classification scaling. The weighing factor is applied by the highest intergration step from the three Scales to the scale of the whole water-body. S_{mep} represents the MEP standard estimated for the required number of species and individual species as defined in Table 10 and Figure 11.

SCALES	Ecosystem Scale		Ecotope Scale				Within Ecotope Scale			Ecological Quality Ratio
Weighing factor	1		2				2			
Eco-logical status	B_{macro}: P_{prim} gAFDW m^2/gCyr^{-1}		Mud-flats %	Shallow areas %	Sand flats %	Mussel banks ha	Macro. diversity	Macro. density	Macro. biomass	
MEP	>1/15 <2/15		>15	>15	>12	>200	S_{MEP}*	t_{calc}<t_{table}	t_{calc}<t_{table}	1.00
GEP	>2/15 <1/5	<1/15 >1/20	<15 >12	<15 >12	<12 >9	<200 >150	>S_{MEP} x0.75			<1.00 <0.75
MODE-RATE	>1/5 <1/2.5	<1/20 >1/40	<12 >9	<12 >9	<9 >6	<150 >100	>S_{MEP} x0.50			<0.75 >0.50
POOR	>1/2.5 <1/1	<1/40 >1/100	<9 >6	<9 >6	<6 >3	<100 >50	>S_{MEP} x0.30			<0.50 >0.30
BAD	>1/1	<1/100	<6	<6	<3	<50	<S_{MEP} x0.30	t_{calc} ≥ t_{table}	t_{calc} ≥ t_{table}	<0.30

Figure 4.23 Establishing the ecological quality ratio based on ecosystem scale, ecotope scale and within ecotope scale (NIOO-CEMO)

The 'within ecotope scale' benchmark is the most useful for the toxic validation since toxics will have a direct possible impact on the macro diversity, macro density and macro biomass. NIOO-CEMO has carried out field surveys within the 13 ecotopes of the Western Scheldt (see *Figure 4.10*) and reported the ecological status within the ecotopes.

Ecological status and score obtained for the four sub-indicators at the within ecotope scale and average score and qualification obtained for the whole within ecotope scale

	Ecological status	Ecological score
Diversity	GEP	98
Total macrofauna density	GEP	83
Total macrofauna biomass	POOR	42
Average at within ecotope scale	MODERATE	74

Figure 4.24 The overall 'within ecotope scale' classification of the Western Scheldt (NIOO-CEMO)

The overall macro-fauna biomass is poor, Figure 4.24. There could be a relation with toxic stress. To focus on a possible relation, the ecotoxicological study area 'Verdronken Land van Saeftinghe' was evaluated on ecotopes:

- Ecotope number 1 : Brackish midlow-littoral(hdyn)
- Ecotope number 2 : Brackish midlow-littoral(ldyn) – muddy

For these two ecotopes the diversity, total macrofauna density and total macrofauna biomass were used, Tables 4.2 to 4.5.

Table 4.2 Total number of species for relevant ecotopes 'Verdronken Land van Saeftinghe'

Total sampling surface and total number of species per ecotopes in the assessment dataset. Number of species required for the MEP, GEP, Moderate, Poor and Bad, estimated from the functions in Table 10. Ecological status and scores estimated for the assessment data.

Eco.#	Surface m2	N_species in assessment	Estimations for the required number of species					Ecological status	Ecological score
			MEP	GEP	MOD.	POOR	BAD		
1	0.615	39	26	19	13	8	<8	MEP	100
2	0.33	29	15	12	8	5	<5	MEP	100

Table 4.3 Species list for relevant ecotopes 'Verdronken Land van Saeftinghe'

Results from the study on the species list per ecotope. Surface is the surface sampled, list% the individual probability for the species in the list, N in samples, the number of species in the assessment sample; N_MEP,GEP, MOD, POOR, the number of species that are required for the different ecological status; Status and score, the ecological status and score respectively.

Eco#	Surface	list%	N in samples	N_MEP	N_GEP	N_MOD.	N_POOR	Status	Score
1	0.62	90	17	14	10	7	4	MEP	100
1	0.62	75	11	8	6	4	2	MEP	100
1	0.62	25	7	2	1	1	1	MEP	100
2	0.33	90	14	11	8	5	3	MEP	100
2	0.33	75	4	1	1	1	0	MEP	100
2	0.33	25	2	0	0	0	0	MEP	100

Table 4.4 Macrofauna density for relevant ecotopes 'Verdronken Land van Saeftinghe'

Outcomes of the t-test comparing the macrofauna density in the assessment dataset and in the reference. The ecological status and corresponding score is defined depending on the outcome of the test. The ecological status and score at the scale of the ecotope is calculated as the average of the individual ecotope scores

Eco#	Density in MEP	Density in assessment	t_{test}	$t_{0.05}$	State	Score
1	5751	6500	1.90	1.97	MEP	100
2	11500	12809	1.54	2.00	MEP	100

Table 4.5 Biomass for relevant ecotopes 'Verdronken Land van Saeftinghe'

Outcomes of the t-test comparing the macrofauna biomass in the assessment dataset and in the reference. The ecological status and corresponding score is defined depending on the outcome of the test. The ecological status and score at the scale of the ecotope is calculated as the average of the individual ecotope scores

Eco#	Biomass in MEP	Biomass in assessment	t_{test}	$t_{0.05}$	State	Score
1	2.05	2.08	1.44	1.97	MEP	100
2	9.78	6.96	2.32	2.00	BAD	0

All biological data in Tables Table 4.2 - Table 4.5 were published by NIOO-CEMO (MEP/GEP analyses for the Western Scheldt).

5. Flood induced pollution, a study case near the city of Middelburg

5.1 Introduction

During flood events, huge amounts of sediment are transported to the inundated area. The irregular supply of thousands of tons of sediment may shape the inundated area. Due to the occasional sedimentation of massive amounts of sediments is this situation not only from the morphology point of view undesired, but also from the water quality point of view. The excess of sedimentation during flood events carries with it significantly high levels of pollutant fractions absorbed to the sediment. Moreover, the transport and distribution of pollutants from storage facilities or industries (calamity events) is another risks during flood events. High concentrations of pollutants offer high ecotoxicological risks not only for plants and animals, but also for humans.

WL | Delft Hydraulics has developed a water quality model framework Sobek 1D/2D for studying the pollutant distribution and predicting the dissolved concentration at different locations and at different stages in a flood plain. The stages can be inundation versus dry top layer or running water versus stagnant water. This is done in conjunction with the hydraulic models used in Sub-Theme 1.2 (flood inundation modelling). These result in a first approach (fundamental research) with a practical spin off for knowledge rules suitable for FLOOD*site* to be implemented in Sobek 1D2D.

The case study was carried out for Middelburg and the surroundings, in the province of Zeeland, based on the collapse of dikes and the inundation of the land (see Figure 5.1). The study area considers a surface about 199 km2 with elevations that go from 1.2 m below see level in the interior to higher than 13 m above see level in the dike area close to the coast.

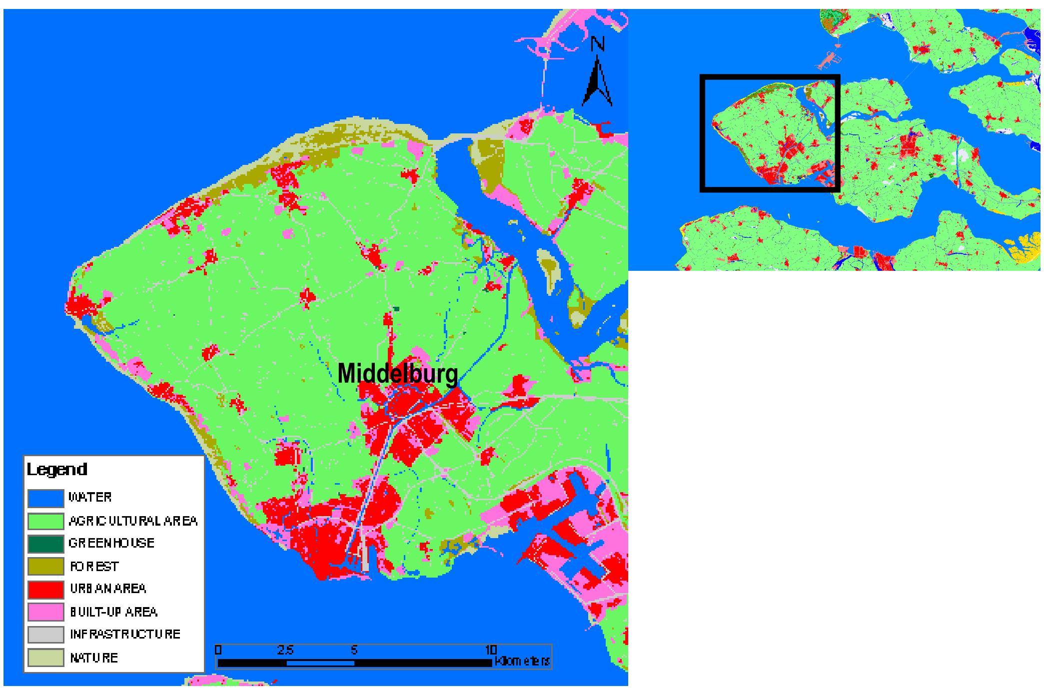


Figure 5.1 Overview of the study area Middelburg

5.2 Methods and Materials

In this paragraph the water quality model framework Sobek 1D/2D is presented. With this model the sediment transport as well as the distribution of pollutants during a flood event was studied. The model enhances the insight in the spatial distribution of sand, silt and clay on the inundated area. With regard to the pollutants, this study focuses on the heavy metals. The heavy metals cadmium (Cd), copper (Cu), and zinc (Zn) are the dominant pollutants in the Western Scheldt (Wijdeveld, 2006). A better understanding of the sediment deposition and the metal fractions absorbed to the sediment during flood events is of great use for the evaluation of measures that should avoid the excess polluted sedimentation.

5.2.1 The hydrodynamic model

Schematisation

The hydrodynamic model regards the northeast section of the hydrodynamic model for the Western Scheldt that has been used in the simulation of dam breaks and flooding (Sub-Theme 1.2: flood inundation modelling). The water quantity model has been setup in the 1D Channel Flow and the 2D Overland module of Sobek. Sobek is software set for the hydrodynamics and the water quality simulation in undimensional channels and flood plains (two dimensions).

The 1D model simulates the Channel over Walcheren. The 2D schematisation covers the whole study area (Middelburg and the surroundings) and has a 300*300 m^2 grid. This model considers an elevation map, as well as a friction map. The friction values for the roughness are between 0.1 and 10 $s/m^{1/3}$.

The inundated area is about 50 km^2, one quarter of the total area of study. This is shown in Figure 5.2. The simulation period for the hydrodynamic period is from January 4th to January 29th 2010. The dam break is simulated to happen on January 6th at 00:00. In order to study the pollutant distribution at different stages in the 2D flood plain an adaptation was done to the hydrodynamic model. From January 15th it is assumed that the dam is somehow repaired and that during the next 15 days (until January 30th) no more water is inundating the area. By doing this it is possible to study the stage 'inundation versus dry top layer' (or 'running water versus stagnant water').

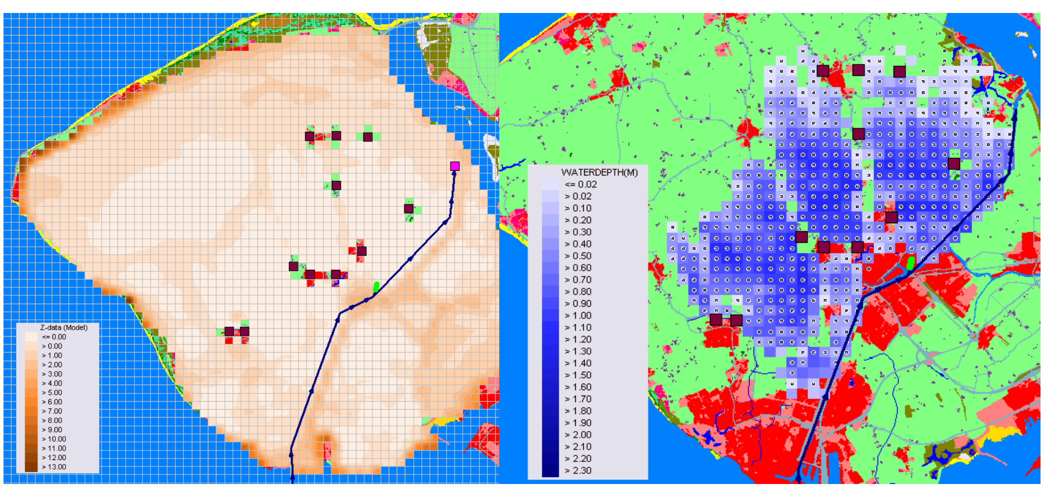


Figure 5.2 Elevations in the study area (right). Inundated area (left)

Inflow and outflow

The inflow water level and the outflow discharge are simulated as boundary conditions of the 1D model. The water level of the inflow 1D boundary is given in Figure 5.3. From January 15th the water level is set at 0 m above see level, which is a lower level than the elevation of the cell where the water flows into the 2D model.

The outflow discharge is set on 1 m^3/s until January 14th at 24:00. For the following period (until January 30th) the outflow discharge is assumed to be zero.

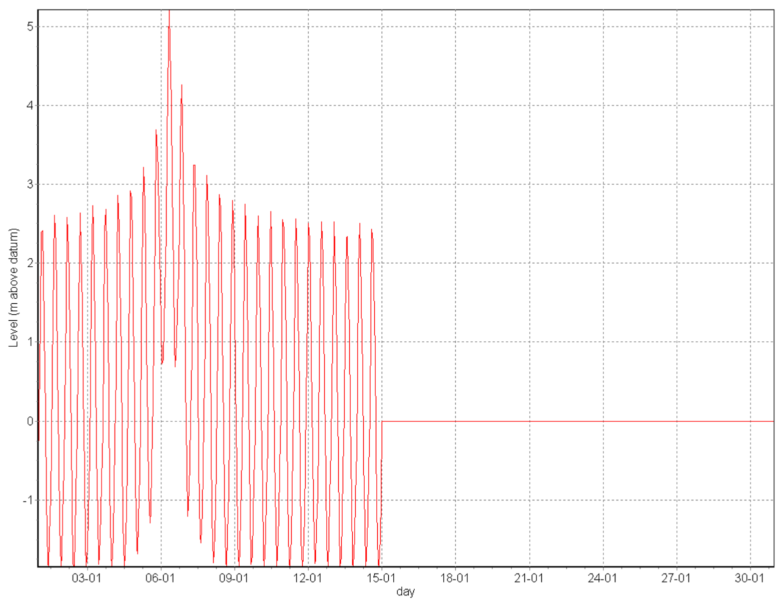


Figure 5.3 Water level at the inflow boundary of the 1D model

5.2.2 The water quality model

The level of detail and the number of substances added in the model depend on the water quality problem to be investigated and of course of the behaviour of the water system. The problems that are studied in this simulation refer to the transport of sediments and to the dispersion of heavy metals in sediment and in the water column.

The water quality model has been setup in the 1D2D Water Quality module of Sobek. The water quality model uses the process library Delwaq as the central core. The process library Delwaq contains different process equations with a large range of substances and problems of water quality. The Sobek version that has been used corresponds to 2.11.000.16. It is recommended to consider the installation of newer versions of Sobek in the future because the module Sobek 2D Water Quality is still in a developing phase.

The 1D Water Quality module and the 2D Water Quality module can be used simultaneously. In this way the exchange of sediments as well as the water quality between channels and inundated areas can be simulated. The hydrodynamics are computed jointly for 1D and 2D in one integrated calculation. For water quality, the calculation procedure is different: the 1D domain and the 2D domain are calculated separately as two different domains. The two Delwaq simulations exchange information about the 1D-2D connections during every time step. The simulation procedure is shown in Figure 5.4, Channel Flow and Overland Flow calculations are integrated; the 1D and 2D Water Quality calculations are running simultaneously.

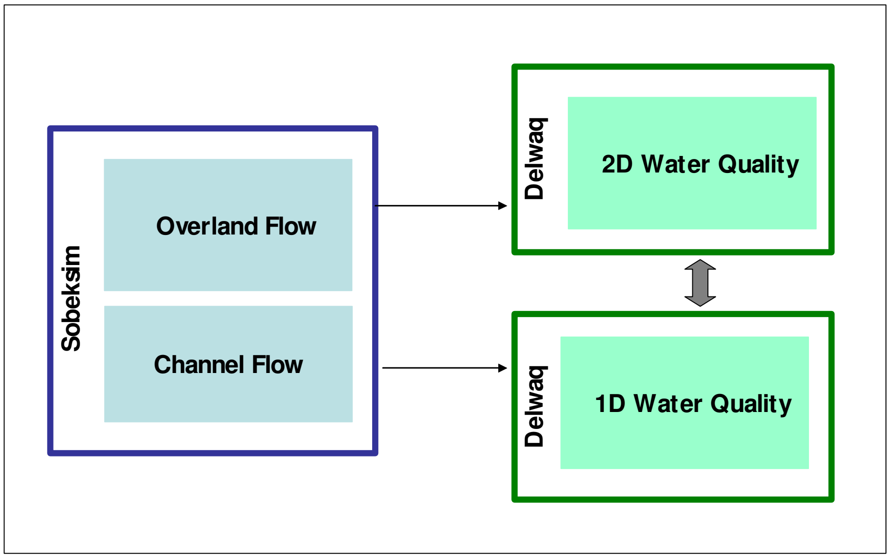


Figure 5.4 Configuration of the Sobek modules that are used for the sediment transport calculations.

Suspended and in the sediment solids

The water quality model considers three fractions of sediment: sand (IM1), silt (IM2) and clay (IM3). Each fraction has its own sedimentation characteristics. The sediment is defined as one layer (S1).

The sediment transport is modelled with a re-suspension - sedimentation approach. The bed shear stress is the determining parameter for re-suspension and sedimentation. The bed shear stress is a function of the flow velocity, the roughness of the river bed and the water depth. At low bed shear stresses, sedimentation of suspended particles prevails. At intermediate bed shear stresses, sedimentation and re-suspension are in a state of equilibrium. When the flow velocity and bed shear stress exceed a certain threshold level, the sediment layer goes into re-suspension, see also Figure 5.5.

The sedimentation velocity and the critical shear stress for sedimentation are different for each suspended solids fraction. The re-suspension velocity and the critical shear stress for re-suspension, on the other hand, apply to all three sediment fractions. These three sediment fractions are assumed to be mixed within the sediment layer. The re-suspension flux of each sediment fraction depends on the relative amount of the particular fraction in the sediment layer.

The sediment model has been set up by use of the processes library Delwaq. The sediment model is available as a pre-defined subset, so the sediment model can be coupled easily to other model schematisations. Table 5.1 summarises the output variables of the water quality model.

The re-suspension of the sediments is controlled by the tangential tension at the bottom caused by the flow and by the waves created by the wind. However in this model the influence of the wind is not considered.

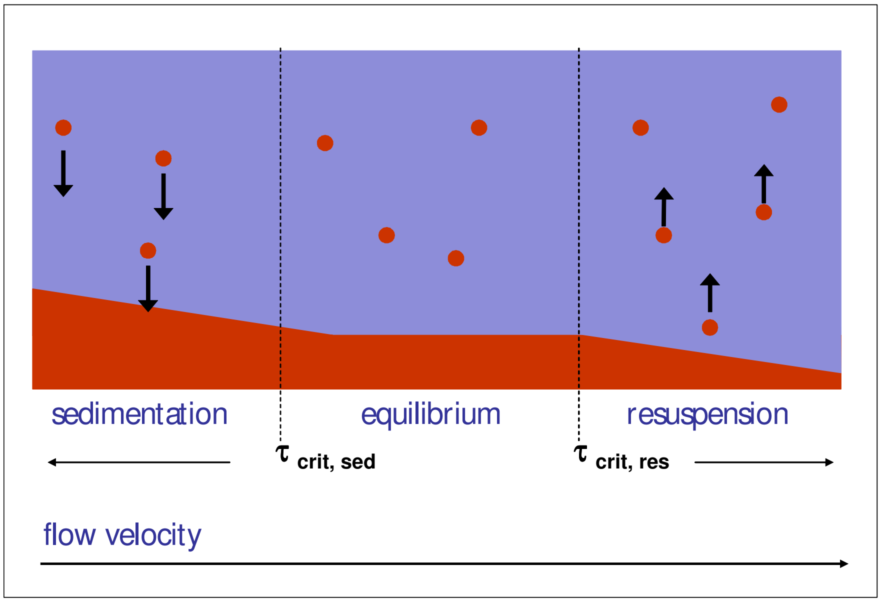


Figure 5.5 Re-suspension –sedimentation in the sediment model.

Table 5.1 The output variables of the sediment model

variable	name	units
IM1	mineral fraction 1 – sand	g/m^3
IM2	mineral fraction 2 – silt	g/m^3
IM3	mineral fraction 3 – clay	g/m^3
IM1S1	mineral fraction 1 in sediment	g/cell
IM2S1	mineral fraction 2 in sediment	g/cell
IM3S1	mineral fraction 3 in sediment	g/cell
IM1S1M2	amount of mineral fraction 1 in sediment	g/m^2
IM2S1M2	amount of mineral fraction 2 in sediment	g/m^2
IM3S1M2	amount of mineral fraction 3 in sediment	g/m^2
fSedIM1	sedimentation flux of mineral fraction 1	g/m^2,d
fSedIM2	sedimentation flux of mineral fraction 2	g/m^2,d
fSedIM3	sedimentation flux of mineral fraction 3	g/m^2,d
fResS1IM1	re-suspension flux of mineral fraction 1 from the sediment	g/m^2,d
fResS1IM2	re-suspension flux of mineral fraction 2 from the sediment	g/m^2,d
fResS1IM3	re-suspension flux of mineral fraction 3 from the sediment	g/m^2,d
Tau	bottom shear stress	N/m^2
ActThS1	actual thickness of sediment	m

Heavy metals

In water quality model three heavy metals are considered: cadmium (Cd), copper (Cu) and zinc (Zn). The heavy metals are absorbed to the suspended solids and to the particles of the sediments. The affinity of the heavy metal for a sediment fraction in particular can be different and each affinity can be specified in the water quality model. Moreover, it is possible to adjust the adsorption coefficient for each metal in the water and in the sediments. Figure 5.6 and Table 5.2 show the processes in the model for copper. The processes are similar for cadmium and zinc.

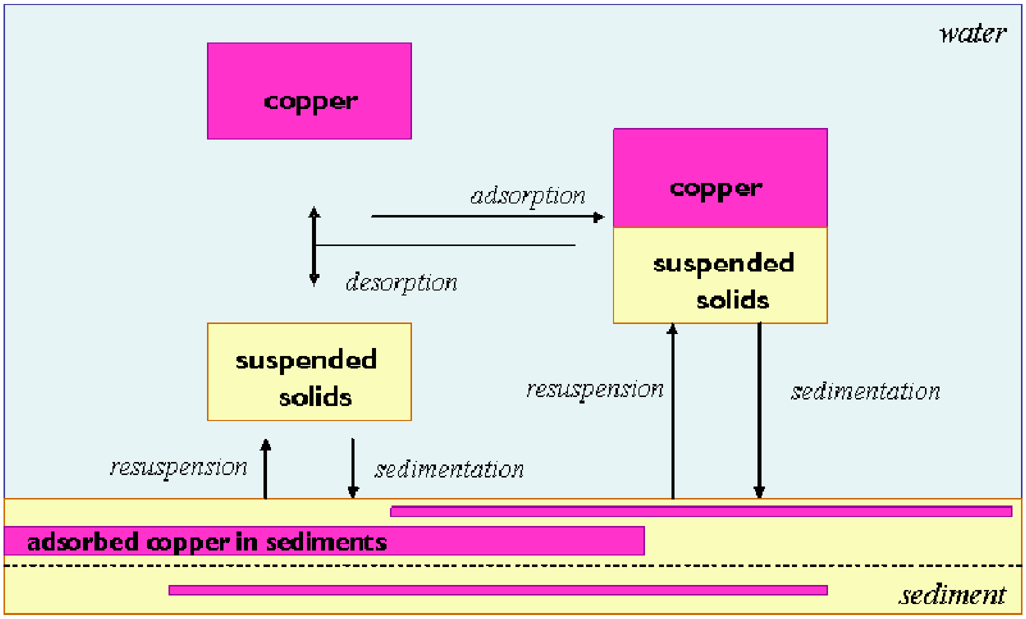


Figure 5.6 Model for copper in the water and in the sediment.

Table 5.2 General relation of the processes in the heavy metal model.

Process	Description
PartS1_Cu	Division Cu en S1
PartS2_Cu	Division Cu en S2
PartWK_Cu	Division Cu in the water column
Res_Cu	Re-suspension adsorbed Cu
Sed_Cd	Sedimentation Cd

Emission and calamity modelling

During flooding events an important risk is the transport of pollutants from urban and rural areas. Storage facilities, industries, pomp stations, farms located in the flood area become then sources of pollutants. But also the pollutants accumulated in the (agricultural and urban) ground. In the water quality model these emission and calamity locations are modelled, as a first approach, with 2D boundary nodes. It is assumed that 6 hours after the cell has been inundated the pollutants are released with a flow of 1 m3/s. In order to know when the grid cell (where the risk location is situated) will be inundated a first run of the hydrodynamic model is done for the entire period.

Figure 5.7 shows the locations of the risk groups with dangerous substances in Middelburg and surroundings (www.risicokaart.nl). This group is defined in the map as the companies that storage, produce or work with dangerous substances. This map also shows the transport routes of dangerous substances through water (in blue) and land (in red), as well as locations with the risk of been inundated. The dangerous substances in the study area belong to three groups: LPG-tank stations, propane gas and companies/factories with dangerous substances (VROM, 2006). In this case, it is referred to the factory Dutch Cleaning Mill bv, which is a factory that by means of riddles and sieves cleans all kind of beans, nuts and seeds.

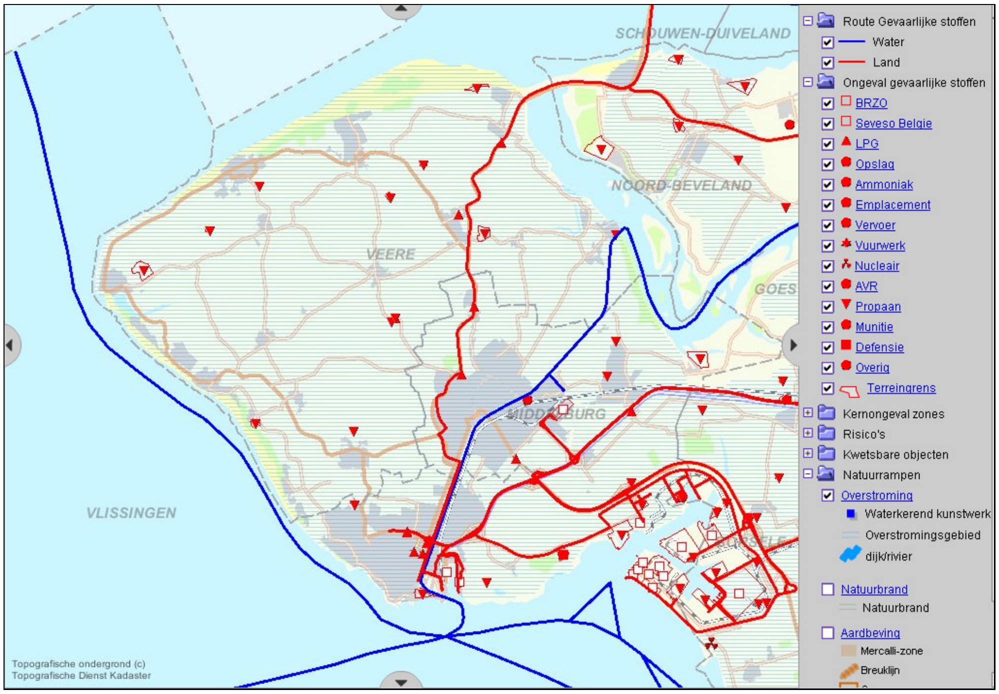


Figure 5.7 Map of risks for Middelburg and surroundings (www.risikokaart.nl)

For modelling the metals accumulated in agricultural and urban grounds, emission factors are used. The yearly national emission factors are found in RIZA (1996). Through the yearly emission factors (e.g. kg/year-house or kg/year-m2 industrial area) and the variables that defined the emission (e.g. number of houses or surface of industrial areas), gross heavy metal emissions (kg/year) can be estimated, Figure 5.8. The variables that defined the emission are found in the website of the Dutch Central Office of Statistics CBS (www.cbs.nl). The emissions of heavy metals (not including the calamity risks at the pump station) are amplified by a factor 100, in order to simulate the accumulation of the pollutants in the ground.

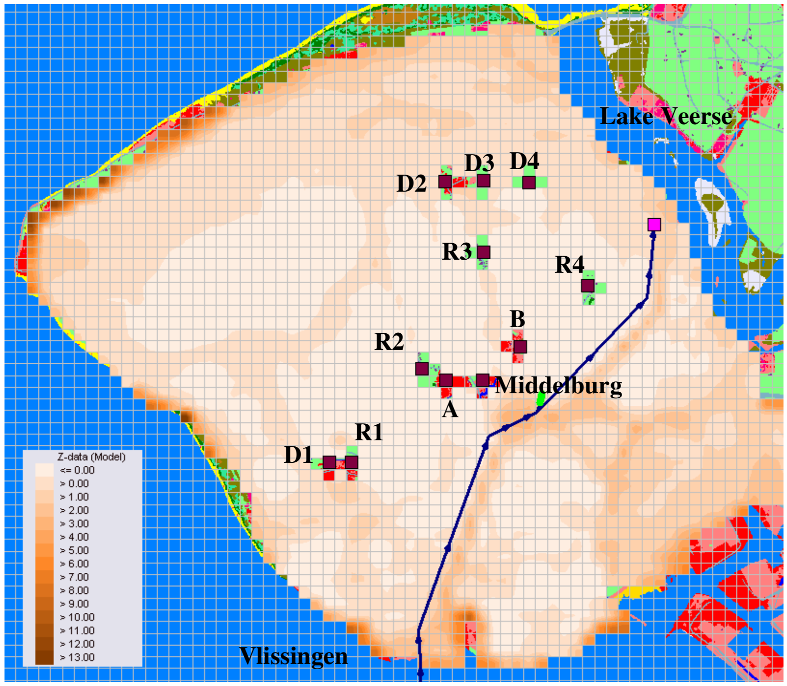


Figure 5.8 Location of sources of heavy metal emissions. D-locations are considered urban locations, R-locations are agricultural locations, A and B are industries.

Assumptions and limitations in the water quality model

The water quality model has some limitations, which are summarised briefly below:

- Bed load as such is not taken into account. However, a sediment fraction with a high density and a high sedimentation velocity can act as a sand fraction in the model.
- The accumulation of sediment at a certain location in the model does not affect the bathymetry of the hydrodynamic model. In a river bed, a significant increase of the bed level causes an increasing flow velocity, thus preventing any further sedimentation.
- As a first approach, the impact of wind induced waves is not taken into account in the model. Wind induced waves may be of importance. The extra shear stress may avoid the sedimentation of suspended solids or even cause re-suspension of the sediment.
- The simulation of emissions through a 2D boundary node is not optimal, since information is missing in the three other directions where the water is not flowing into the 2D plain.
- Complete information about the heavy metal emissions is not available. In order to make an estimation of the emissions many assumptions had to be made (see previous paragraph). The same goes for the calamity events. What is important here to see is the difference between an urban area from a rural area, as well as the spatial distribution of the sources.

5.2.3 Data for the water quality model

Initial conditions

The simulation starts on January 5th of the year 2010, at 00:00 a.m. The simulation begins about one day before the dam break and when the first sediments are transported into the modelled area. Initially, suspended solids are not present in the inundated area (2D model). The sediment layer contains no sediments at the start of the simulation for the 1D model, while the sediment fractions in the 2D model are defined locally. This is done by doing a run of the water quality model only with suspended solids and the sediment. The sediment results in the 2D model of this run are used as initial conditions. The initial conditions are summarised in Table 5.3.

Table 5.3 Initial conditions in the water quality model.

substance	description	initial condition		Units
		1D	2D	
Cd	cadmium in water	0.0002	0	Mg/l
Cu	copper in water	0.008	0	Mg/l
Zn	zinc in water	0.014	0	Mg/l
Continuity		1	1	(-)
IM1	sand in water	0	0	Mg/l
IM2	silt in water	5	0	Mg/l
IM3	clay in water	25	0	Mg/l
IM1S1	sand in sediment	0	Local	g/m^2
IM2S1	silt in sediment	0	Local	g/m^2
IM3S1	clay in sediment	0	Local	g/m^2

Boundary conditions

The inflow of sediments from the channel and through the broken dam are sources of metals and sediments in the model. The boundary in the Western Scheldt represents an inflow (Vlissingen keersluisbrug), while the boundary at Channel door Walcheren close to the lake Veerse represents an outflow (Veere schutsluis). Table 5.4 summarises the boundary conditions for the metals and Table 5.5 for the sediments in the model. Table 5.6 shows the total amount of sediment that enters the system.

Table 5.4 Boundary conditions for the heavy metals during the January 2010 event.

	From		Until		Cd	Cu	Zn
	day	Time	Day	time	(µg/l)	(µg/l)	(µg/l)
inflow 1D Vlissingen keersluisbrug	05-01-2010	00:00	15-01-2010	00:00	0.19	8	14.4
outflow 1D Veere schutsluis	05-01-2010	00:00	15-01-2010	00:00	0.09	12.1	12.8

Table 5.5 Boundary conditions for suspended solids during the January 2010 event.

	From		Until		sand – IM1	silt – IM2	clay – IM3
	day	time	Day	time	(mg/l)	(mg/l)	(mg/l)
inflow 1D	05-01-2010	00:00	15-01-2010	00:00	28	15094	15146
outflow 1D	05-01-2010	00:00	15-01-2010	00:00	6	3019	3029

Table 5.6 Total amount of sediment in the model.

inflow	sand (ton)	Silt (ton)	clay (ton)	total (ton)	percentage
inflow 1D	15.22	8206.73	8235.00	16456.95	146
outflow 1D	5.18	2608.42	2617.06	5230.66	46
Total	**10.04**	**5598.31**	**5617.95**	**11226.30**	

Emissions

The gross yearly emissions of cadmium, copper en zinc in Middelburg are given in table Table 5.7. Because no data were found about the overflow of metals in the study area, the values were taken over from a study done for the area Hunze en Aa's (De Straat Milieu-adviseurs B.V., 2003). The emissions from built-up areas are calculated proportional to the emissions in Middelburg based on the surface between the built-up area and Middelburg (see Table 5.8).

Table 5.7 Gross yearly emissions (kg) of Cadmium, Copper and Zinc in Middelburg

Source	Cd	Cu	Zn
Corrosion roof houses (kg)	-	-	9.94E+02
Corrosion roof companies (kg)	-	-	3.77E+00
Corrosion zinc street ornaments (kg)	-	-	9.21E+01
Wear tires (kg)	1.28E-02	5.24E-05	2.66E-04
Overflow (Hunze en Aa's) (kg)	2.56E-04	4.42E-02	5.58E-02
Load Middelburg Total (kg)	**1.30E-02**	**4.43E-02**	**1.09E+03**

Table 5.8 Gross yearly emissions (kg) of Cadmium, Copper and Zinc in five built-up areas

Built-up areas	Fraction area with regard to Middelburg	Cd	Cu	Zn
D1	0.075	9.67E-01	3.97E-03	8.26E+04
D2	0.038	4.83E-01	1.98E-03	4.13E+04
D3	0.038	4.83E-01	1.98E-03	4.13E+04
D4	0.015	1.93E-01	7.93E-04	1.65E+04
D5	0.023	2.90E-01	1.19E-03	2.48E+04

The sources of metals considered in the rural areas are the wear of copper rails, oil leaks, the pig manure and the artificial manure. In the case of the wear of copper rails and oil the leak, each rural area is assumed to emit 25% of load of heavy metals from both sources in the country side. This is summarized in Table 5.9.

Table 5.9 Gross yearly emissions (g) of heavy metals from the wear of copper rails and oil leaks

Rural areas	Fraction of total emissions	Cd (g)	Cu (g)	Zn (g)
R1, R2, R3, R4	0.25	3.50E-01	3.93E+03	2.26E+02

Pig manure is a source of copper. According to literature values about 15 mg of copper are content in each kg of pig manure (Foundation Biologic Pig Farming, 2003). The pig manure produce in Middelburg in 2005 was about 37231000 kg (CBS data). It is assumed that the load of copper due to this source is equally divided in the four rural areas. This is summarized in Table 5.10.

In the website www.kunstmest.nl information can be found about artificial manure. About 0.4 until 1.6 g Cd can be found per hectare. For this study it is assumed a factor of 1 g Cd/hectare. This is summarized in Table 5.10.

Table 5.10 Cadmium and copper emissions from artificial and pig manure

Rural areas	Area (ha)	Cd-Artificial manure (g)	Cu-Pig manure (g)
R1	657	657	139616
R2	1440	1440	139616
R3	1683	1683	139616
R4	1134	1134	139616

The modelling of calamities focuses on the risks around the pump station and industries. Based on the emission factors defined by RIZA (1996), the fraction between metal and oil is estimated. The volume of the tank in a pump station is assumed to be 150000 litres. The density of oil is defined at 895 kg/m3. The loads of the metals are presented in Table 5.11. The resulted loads are added to the rural area 2.

Table 5.11 Load of metals coming from a pump station in a calamity

	Fraction Metal/Oil	Mass (g)
Cadmium	1.28E-06	172
Copper	3.37E-05	4530
Zinc	8.24E-04	111000

In the case of the industries, yearly emissions are taken from the results of the study in the area Hunze en Aa’s as a reference (De Straat Milieu-adviseurs B.V., 2003). This is shown in Table 5.912.

Table 5.12 Yearly loads (g) of heavy metals from industries

Industry	Cd (g)	Cu (g)	Zn (g)
A	1.00E+01	3.50E+03	1.45E+04
B	1.00E+01	3.50E+03	1.45E+04

The emissions are modelled in the 2D model as boundary nodes. This means that the loads have to be converted to concentrations. In order to estimate the concentrations of heavy metals from the sources here presented, the surface and the water depth of the respective inundated areas (urban and rural) are estimated based on the results of the hydrodynamic model. The volumes (m3) of the inundated areas are presented in Table 5.12. The water depth in the 2D plain is calculated with the hydrodynamic model. The area is estimated according to the grid. The final concentrations (g/m3) given in Table 5.14 and added to the model are a factor 100 bigger.

Table 5.123 Calculated volumes (m^3) of the inundated areas.

Inundated areas	m3
Middelburg	2806200
D1	18000
D2	1800
D3	54000
D4	108000
R1	816750
R2	2269350
R3	2841300
R4	1962900
Industry area A	701550
Industry area B	701550

Table 5.13 Total concentrations of heavy metals (g/m^3) at the different areas considering amplification with a factor 100.

	Cd	Cu	Zn
Middelburg	9.58E-03	1.58E-00	4.08E+03
D1 (g/m3)	5.37E-03	2.20E-05	4.59E+02
D2 (g/m3)	2.69E-02	1.10E-04	2.29E+03
D3 (g/m3)	3.58E-04	1.47E-06	3.06E+01
D4 (g/m3)	2.69E-04	1.10E-06	2.29E+01
A (g/m3)	1.43E-03	4.99E-01	2.07E-00
B (g/m3)	1.43E-03	4.99E-01	2.07E-00
R1 (g/m3)	2.01E-02	4.39E-00	6.90E-03
R2 (g/m3)	1.59E-02	1.58E+00	1.47E-02
R3 (g/m3)	1.48E-02	1.26E-00	1.98E-03
R4 (g/m3)	1.44E-02	1.83E-00	2.87E-03

Process coefficients

The number of process coefficients in the sediment model is limited. For each sediment fraction the density of the material, the sedimentation velocity and the critical shear stress must be defined. The re-suspension velocity and the critical shear stress for re-suspension are two coefficients that apply to all sediment fractions. The porosity applies to the entire sediment layer. The model assumes that the three sediment fractions are completely mixed within the sediment layer. Table 5.15 gives the Kd coefficients of metals for the three sediment fractions.

Table 5.15 Kd coefficients of metals for sand, silt and clay

Metal	Default	IM1 10% Sand	IM2 30% Silt	IM3 60% Clay
Cu	50	8	25	50
Zn	110	18	55	110
Cd	130	22	65	130

Table 5.14 shows the values of the process coefficients used in the model.

As a first approach, the process coefficients for the sedimentation of sand, silt and clay are the same as the default values. The density of the sediment particles and the porosity of the sediment layer determine the simulated thickness of the sediment layer.

Table 5.14 Process coefficients in the sediment model, with default (non-calibrated) coefficient values.

coefficient	Name	Default value	Used value	units
VSedIM1	sedimentation velocity IM1 (sand)	24	30	m/day
VsedIM2	sedimentation velocity IM2 (silt)	4	7	m/day
VsedIM3	sedimentation velocity IM3 (clay)	0.01	0.3	m/day
TauScIM1	critical shear stress sedimentation IM1	0.05	0.07	N/m^2
TauScIM2	critical shear stress sedimentation IM2	0.01	0.05	N/m^2
TauScIM3	critical shear stress sedimentation IM3	0.005	0.02	N/m^2
VResDM	first order re-suspension velocity	0.01	0.01	1/day
TauRSiDM	critical shear stress re-suspension	0.2	0.2	N/m^2
RHOIM1	density IM1	$1.3\ 10^6$	$1.3\ 10^6$	g/m^3
RHOIM2	density IM2	$1.3\ 10^6$	$1.3\ 10^6$	g/m^3
RHOIM3	density IM3	$1.3\ 10^6$	$1.3\ 10^6$	g/m^3
PORS1	porosity sediment layer 1	0.50	0.5	-
MinDepth	minimum water depth for sedimentation	0.01	0.01	m
KdCdIM1	partition coefficient Cd-IM1	130	78	m3/kgDM
KdCdIM2	partition coefficient Cd-IM2	130	65	m3/kgDM
KdCdIM3	partition coefficient Cd-IM3	130	130	m3/kgDM
KdCdIM1S1	partition coefficient Cd-IM1 in layer S1	130	22	m3/kgDM
KdCdIM2S1	partition coefficient Cd-IM2 in layer S1	130	65	m3/kgDM
KdCdIM3S1	partition coefficient Cd-IM3 in layer S1	130	130	m3/kgDM
KdZnIM1	partition coefficient Zn-IM1	110	18	m3/kgDM
KdZnIM2	partition coefficient Zn-IM2	110	55	m3/kgDM
KdZnIM3	partition coefficient Zn-IM3	110	110	m3/kgDM
KdZnIM1S1	partition coefficient Zn-IM1 in layer S1	110	18	m3/kgDM
KdZnIM2S1	partition coefficient Zn-IM2 in layer S1	110	55	m3/kgDM
KdZnIM3S1	partition coefficient zn-IM3 in layer S1	110	110	m3/kgDM
KdCuIM1	partition coefficient Cu-IM1	50	8	m3/kgDM
KdCuIM2	partition coefficient Cu-IM2	50	25	m3/kgDM
KdCuIM3	partition coefficient Cu-IM3	50	50	m3/kgDM
KdCuIM1S1	partition coefficient Cu-IM1 in layer S1	50	8	m3/kgDM
KdCuIM2S1	partition coefficient Cu-IM2 in layer S1	50	25	m3/kgDM
KdCuIM3S1	partition coefficient Cu-IM3 in layer S1	50	50	m3/kgDM

Simulation period

The hydrodynamic model starts at January 4th 2010. The dam break occurs at January 6th at 00:00. The inflow of sediments at the 1D boundaries of the model lasts from January 5th until January 15th. At this moment it is assumed that the dam somehow is repaired. The sedimentation of suspended solids continues for some time until January 29th, after the inflow of material has ended. The sedimentation velocity of the finest sediment fraction is very low and sedimentation may continue several days or weeks after the transport of sediments to the lake has ended. January 30th is the end data of the flow simulation.

5.3 Simulation results

In this chapter the results are given for the water quality model in terms of the solids and the heavy metals. They refer mainly to the concentrations in the sediment layer S1. The results of the concentration of the substances in the water column are in this report not presented in order to give emphasis to the deposition of the sediments and the pollutants absorbed on to them after the break of a dam.

The results give also the pollutant distribution at different stages in the 2D flood plain. The first stage is the dam break until January 15th. From January 15th it is assumed that the dam is some how repaired and during the next 15 days no more water is inundated the area. By doing this it is possible to study the stage ‘inundation versus dry top layer’ or ‘running water versus stagnant water’.

5.3.1 Results of the solids in the sediment

The results of the sediment model consider the mass of the three fractions of sediments: sand, silt and clay. A main factor here is the tangential tension. The results of the tangential tension are given in Figure 5.109.

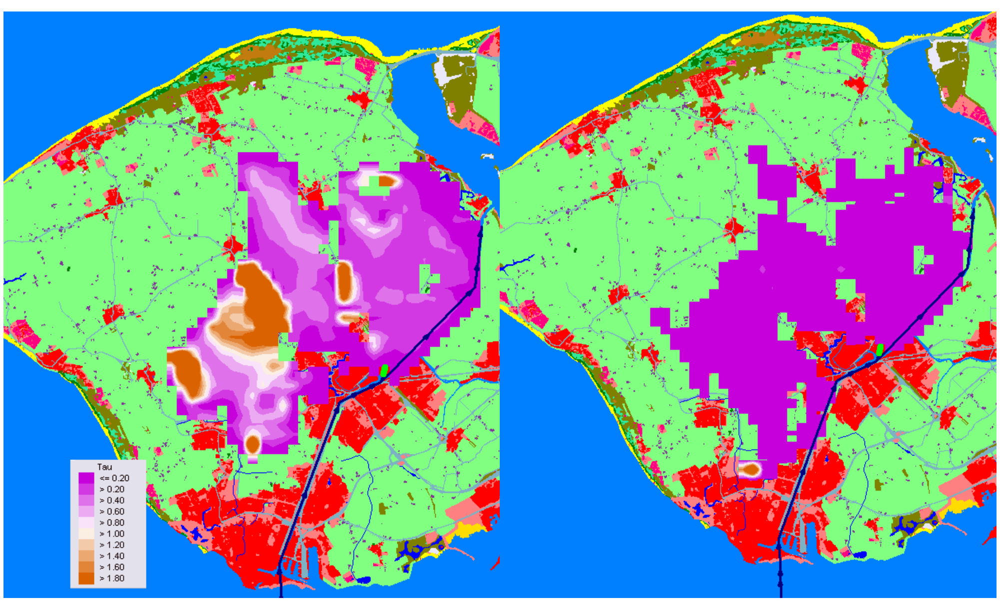


Figure 5.9 Tangential tension results on January 15th and January 29th.

The sediment balance was made for the period January 5th until January 29th. Figure 5.110 to 5.12 show the spatial distribution of sediment mass (g) in the inundated area for sand, silt and clay respectively.

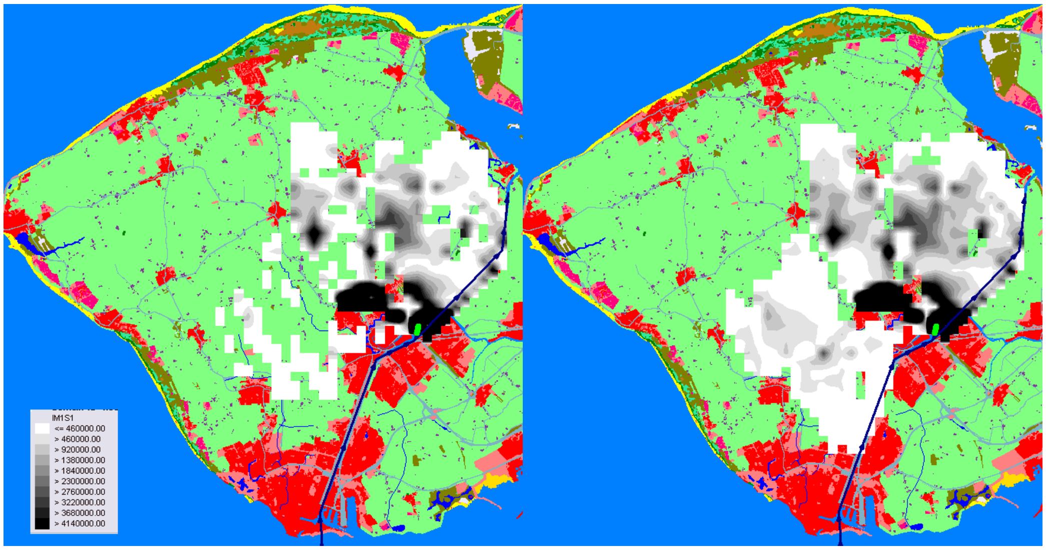


Figure 5.10 Amount of freshly deposited sand (g) on January 15th and January 29th. The sedimentation velocity of sand is 30 m/d.

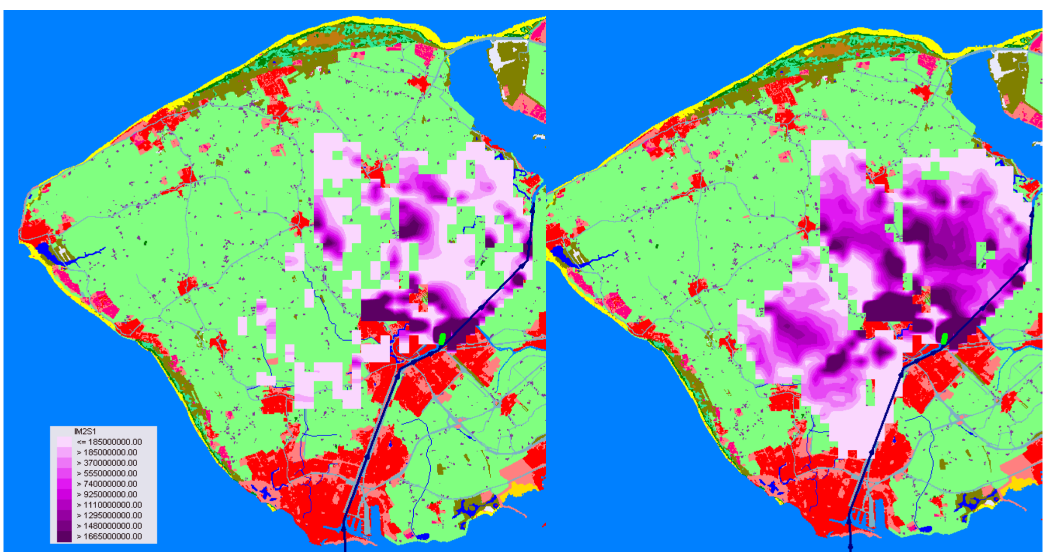


Figure 5.91 Amount of freshly deposited silt (g) on January 15th and January 29th. The sedimentation velocity of silt is 7 m/d.

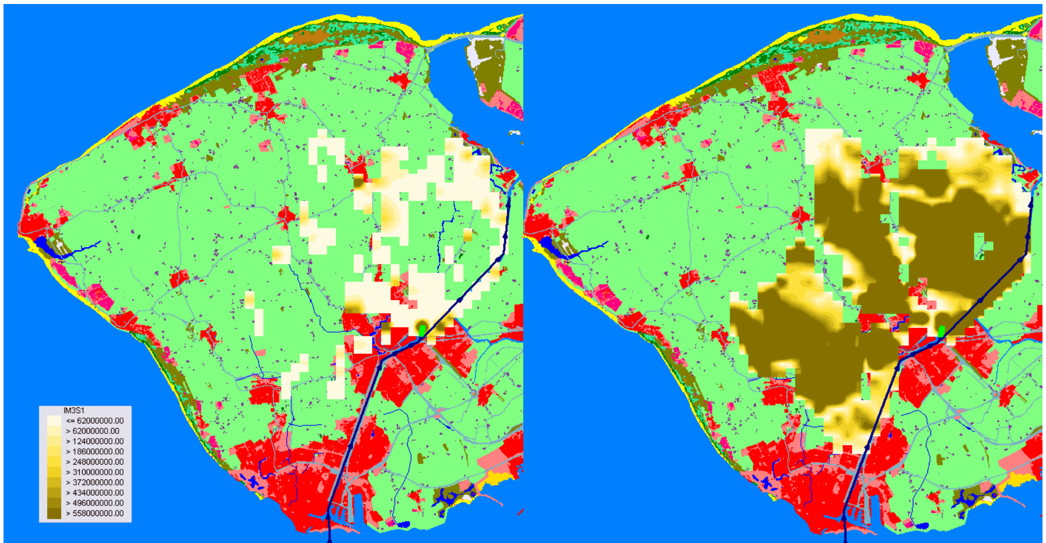


Figure 5.12 Amount of freshly deposited clay (g) on January 15th and January 29th. The sedimentation velocity of clay is 0.3 m/d.

The mass balance for the three sediment fractions in the 2D water quality model is given in Table 5.15. Both stages are presented in the table: January 15th (running water) and January 29th (stagnant water).

Table 5.15 Mass balance for the sediment in the 2D model for both stages

January 15th						
	IM1	IM2	IM3	IM1S1	IM2S1	IM3S1
Total mass in system	1.8526E+08	2.8645E+11	5.1301E+11	7.2050E+08	2.3241E+11	2.7076E+10
Changes by processes	-5.1890E+05	-2.0252E+08	-4.3489E+07	5.1890E+05	2.0252E+08	4.3489E+07
Boundary inflows	0.0000E+00	0.0000E+00	0.0000E+00	0.0000E+00	0.0000E+00	0.0000E+00
Boundary outflows	5.8544E-04	1.3977E+04	1.5808E+04	0.0000E+00	0.0000E+00	0.0000E+00
January 29th						
	IM1	IM2	IM3	IM1S1	IM2S1	IM3S1
Total mass in system	4.7501E+04	1.9659E+08	9.6737E+10	9.0571E+08	5.1866E+11	4.4341E+11
Changes by processes	-1.5720E+02	-2.7319E+06	-4.2389E+08	1.5720E+02	2.7319E+06	4.2389E+08
Boundary inflows	0.0000E+00	0.0000E+00	0.0000E+00	0.0000E+00	0.0000E+00	0.0000E+00
Boundary outflows	0.0000E+00	0.0000E+00	0.0000E+00	0.0000E+00	0.0000E+00	0.0000E+00

The simulation results are quite different for sand, silt and clay, due to their differing properties. Table 5.18 summarizes the distribution of the solids in the water and in the sediment. The sand is deposited almost immediately, near the outflow of the broken dam and also close to the cells where the flow is overtopping the embankments. The small amount of sand that is still in suspension after almost one month is trapped in the water in the areas with a low water depth. Sedimentation stops when the water depth is lower than a minimum water depth of 1 centimetre and the sand remains in suspension.

About 45% of the silt is deposited in the inundated area on January 15th while on January 29th all of the silt is settled down.

Due to the low sedimentation velocity (0.3 m/day), most of the clay remains in suspension for a long time. On January 15th 95% of the clay remains in suspension and on January 29th the percentage goes down to 18%. Furthermore, the clay sediments only on locations with low flow velocities.

Table 5.18 Distribution (percentage) of the three sediment fractions during both stages.

	IM1	IM2	IM3	IM1S1	IM2S1	IM3S1
January 15th	20	55	95	80	45	5
January 29th	0	0	18	100	100	82

The spatial distribution of sediment depends on:

- the spatial distribution of the water that is loaded with sediments;
- the time scale of sedimentation versus the time scale of hydrodynamics.

The exact location where clay, silt and sand will settle depends on the sedimentation velocity of the particles and the flow velocity of the water.

The thickness of the sediment layer and the volume of the sediment in the lake and the rice fields can be computed when the porosity and the density of the sediment is known. Figure 5.103 shows the thickness of the newly formed sediment layer. The density of the three sediment fractions is assumed to be 1.3 ton/m3, the porosity of the sediment is 0.50.

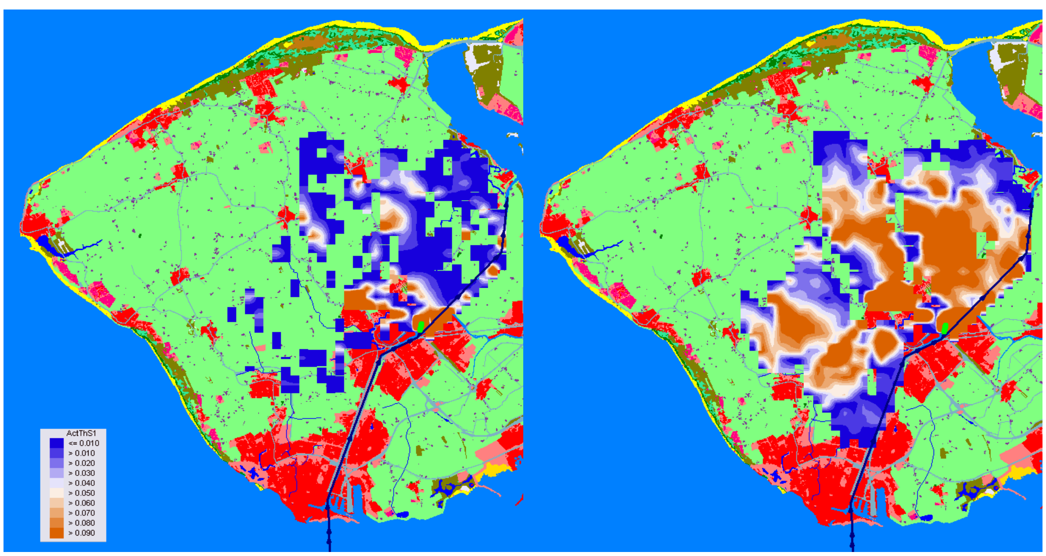


Figure 5.103 The thickness (m) of the freshly formed sediment layer on January 15th and January 29th.

5.3.2 Results of the heavy metals in the sediment

The results of the sediment model consider the mass of the heavy metals cadmium, copper and zinc absorbed to the sediment. Figure 5.11 to 5.16 show the spatial distribution of sediment mass (g) in the inundated area for cadmium, copper and zinc respectively. The spatial distribution of the heavy metals is quite similar. This is due to the fact that heavy metals are mainly absorbed to the organic matter (clay). Therefore where high concentrations of clay are distributed, also high concentrations of metals are presented.

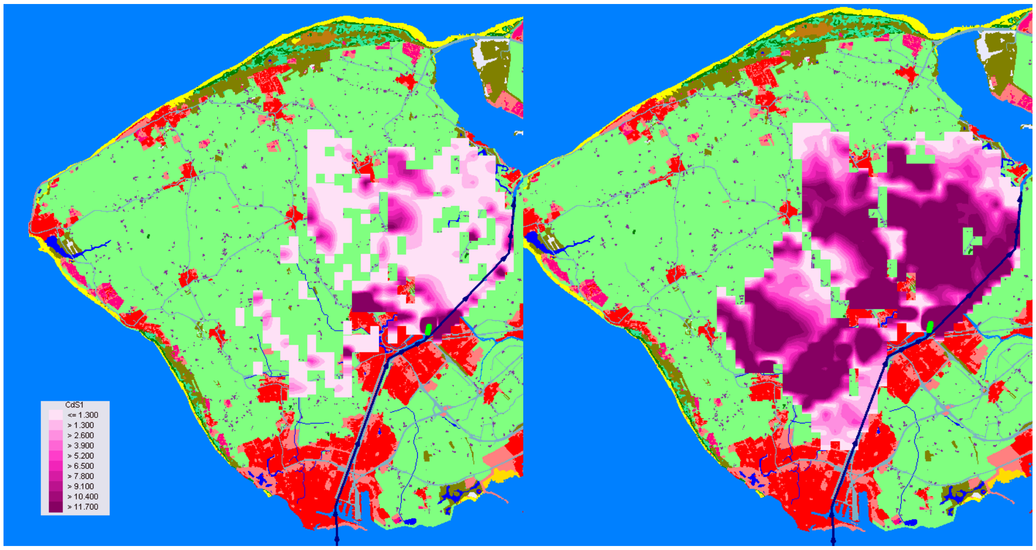


Figure 5.114 Amount of freshly deposited cadmium (g) on January 15th and January 29th.

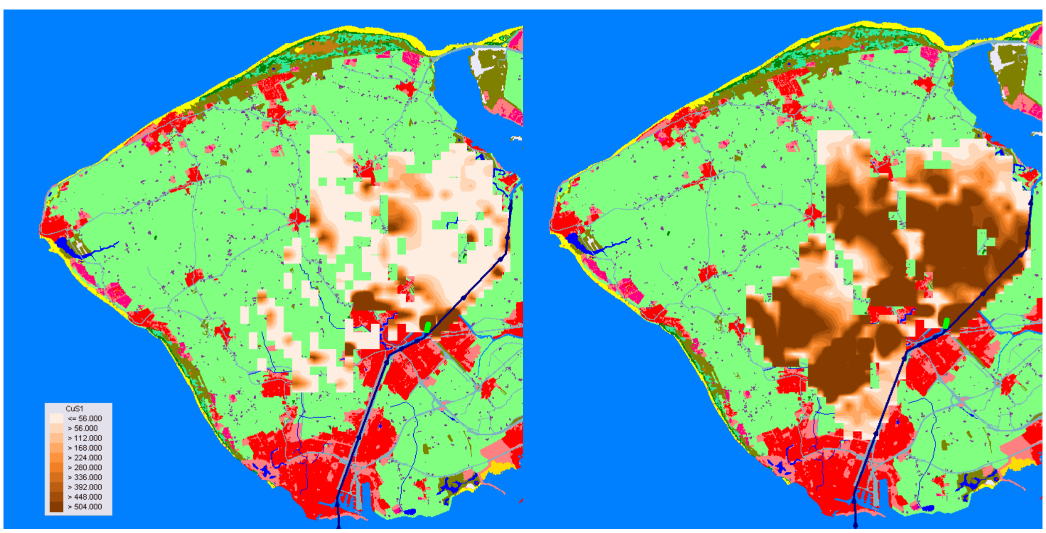


Figure 5.125 Amount of freshly deposited copper (g) on January 15th and January 29th.

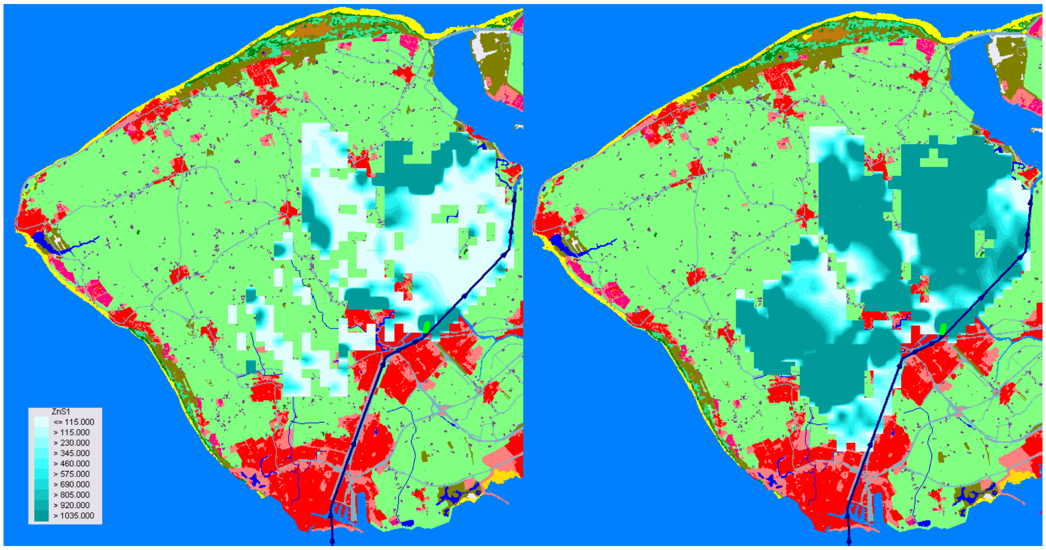


Figure 5.16 Amount of freshly deposited zinc (g) on January 15th and January 29th.

The mass balance for the heavy metals in the 2D water quality model is given in Table 5.19. Both stages are presented in the table: January 15th (running water) and January 29th (stagnant water). Table 5.20 summarizes the distribution of cadmium, copper and zinc in the water and in the sediment.

Table 5.19 Mass balance for the heavy metals in the 2D model for both stages (January 15th and January 29th)

January 15th						
	Cd	Cu	Zn	CdS1	CuS1	ZnS1
Total mass in system	5.9265E+03	2.6581E+05	1.6250E+07	2.1377E+03	1.1019E+05	4.5281E+07
Changes by processes	-5.2134E+00	-6.2356E+01	-3.2151E+05	5.2134E+00	6.2356E+01	3.2151E+05
Boundary inflows	0.0000E+00	0.0000E+00	0.0000E+00	0.0000E+00	0.0000E+00	0.0000E+00
Boundary outflows	1.9084E-04	8.0363E-03	1.4465E-02	0.0000E+00	0.0000E+00	0.0000E+00
January 29th						
	Cd	Cu	Zn	CdS1	CuS1	ZnS1
Total mass in system	8.4739E+02	3.6778E+04	1.6257E+05	7.2460E+03	3.3941E+05	6.1366E+07
Changes by processes	-3.7395E+00	-1.6230E+02	-9.5853E+02	3.7395E+00	1.6230E+02	9.5853E+02
Boundary inflows	0.0000E+00	0.0000E+00	0.0000E+00	0.0000E+00	0.0000E+00	0.0000E+00
Boundary outflows	0.0000E+00	0.0000E+00	0.0000E+00	0.0000E+00	0.0000E+00	0.0000E+00

Table 5.20 Distribution (percentage) of the heavy metals during both stages.

	Cd	Cu	Zn	CdS1	CuS1	ZnS1
January 15th	73	70	26	27	30	74
January 29th	10	10	0.3	90	90	99.7

5.3.3 Recommendations

- The present grid is 300*300 m2. In order to have a better description of the distribution of the heavy metals and sediments in the inundated area, the grid can be refined to 100*100 m2.
- The suspended solids concentrations at the boundaries of the model are constant in the present model schematisation. In reality, the suspended solids concentrations are most likely a function of the discharge and therefore a function of time. This also goes for the heavy metals.
- The effect of the wind has to be included. Wind induced waves may be of importance. The extra shear stress may avoid the sedimentation of suspended solids or even cause re-suspension of the sediment.
- Sensitivity analysis. The uncertainty by modelling of a calamity situation is considerable, especially when we want the model to be calibrated. However, a sensitivity analysis may give more understanding of the effects of the flooding in the water quality. Table 5.21 lists the parameters that can be including in a sensitivity analysis.

Table 5.21 Parameters in the sediment model

parameter	effect
sedimentation velocity	spatial distribution of sediments overall sediment mass balance
critical shear stress	spatial distribution of sediments overall sediment mass balance
dispersion coefficient	spatial distribution of sediments
density	thickness of sediment layer
porosity	thickness of sediment layer
inflow of sediments	overall sediment balance
hydrodynamic simulation	spatial distribution of sediments overall sediment mass balance

6. General conclusions

6.1 Development of the OMEGA model

The OMEGA model is useful as a screening tool to determine if there is a possible toxic risk in a water body. The OMEGA tool is also useful to determine which (priority) substances are causing the main toxic pressure factor. The OMEGA tool can also be useful to determine which groups of organisms are at risk.

The OMEGA model is based on dissolved water concentrations; in case total water concentrations have been measured simple algorithms have been developed to translate total water concentrations to dissolved water concentrations. This will however introduce an extra uncertainty in the model outcome. The use of these translating algorithms for sediment (to derive pore water concentrations from total sediment concentrations) is highly questionable and great care should be taken to evaluate the model outcome.

The expansion of the OMEGA model with information on the mode of action for the toxic substances within OMEGA was useful; it helped on improving on the risk addition calculation for multiple toxicants. The specification of four different groups of organisms to calculate different toxic stress levels for each group can be very helpful in evaluating the effect of toxic stress on the water body (like the toxicity of herbicides can be high for macrophytes but low for fish).

6.2 The use of OMEGA in the Western Scheldt case study

The OMEGA model was successfully applied to calculate the mixture toxicity of pollutants in the Western Scheldt. The OMEGA model yielded a Potential Affected Fraction (ms-PAF) on a chronic exposure level of around 9 % for the study area (Land van Saeftinghe). The ms-PAF varied between 11% in the winter to 7% in the summer. The main pollutant causing the ms-PAF is copper (80-90%).

Translating this outcome to risks, 9% of the species present in the study area might have an effect caused by toxicants. Since the definition of effect on a chronic level is based on NOEC (No Observed Effect Concentration) experiments, the toxic effect might vary from growth inhabitation to a higher mortality.

The ms-PAF level of 9% (on average) might have an impact on the ecological status of the study area. For validation, the ms-PAF for the study area has been compared with the ecological status for the ecotopes (2 in total) within the study area. The ecological status is based on the index for the diversity, total macrofauna density and total macrofauna biomass. Only the biomass index for one of the two ecotopes has a bad ecological status, all other indexes meet the MEP (Maximal Ecological Potential) status. Therefore the evidence that there is a toxic effect in the studied area, based on a ms-PAF of around 9%, is inconclusive at the moment. A more detailed field study in the area to observe if specific groups of organisms are indeed affected is needed to further test the effects of toxic stress.

6.3 Flood induced pollution in a case study near Middelburg

During flood events, huge amounts of sediment are transported to the inundated area. The excess of sedimentation during flood events carries with it significantly high levels of pollutant fractions absorbed to the sediment. Moreover, the transport and distribution of pollutants from cities, storage facilities or industries is another risk during flood events.

By combining a 1D/2D hydrodynamic model with erosion/sedimentation of suspended sediment and the release of pollutants of different origin a simulation map could me made of the pollution pattern during and after the flooding. In this model, different suspended sediment fractions (sand, silt, clay) were considered with different sedimentation rates and different sorption characteristics for pollutants. Because cities, storage facilities and industries also vary in the exact pollutant that is released pollutant, the distribution pattern is very pollutant specific.

The simulation maps can help decision makers in predicting where to look for historically distributed pollutants during earlier flooding or help in planning a clean up operation after a future flooding.

7. References

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Appendix A: Water storage in nature conservations areas: does current knowledge suffice?

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Abstract

Some years ago, water management policy in the Netherlands and surrounding countries focused on fighting drought damage to nature conservation areas. Recently, however, the 1993 and 1995 river floods and excessive rainfall in 1998 and 1999 caused attention to shift to the question of how to deal with flooding problems. Both quantitative water management problems relate to climate change, which is expected to aggravate these problems in the future. For both problems, water storage is proposed. It is even suggested that both problems can be solved with one and the same measure, in one and the same area. After all, the nature conservationists wanted more water for their nature conservation areas? In this paper we question whether we all mean the same with water storage and whether water management functions can combined, and we review the existing knowledge on the relationship between quantitative and qualitative (ground)water variables and biota. Finally, we discuss what extra research is needed to achieve at sound management decisions and we give some recommendations about what decisions can already be made without endangering our remaining natural values.

Water management problems

In densely populated countries, such as the Netherlands, nature conservation areas are patches within a matrix of other functions, primarily agricultural land, with different requirements on water level and quality. To achieve the required water regime for a specific function the water system is being fully controlled by draining, pumping, canalizing and damming. In polder areas the existing nature is adapted to a good regulated and detailed surface water level. In addition, careful management of the vegetation by mowing or extensive grazing has gradually increased the nature values either by being a species rich peat land with nutrient poor conditions or by providing a habitat for valuable meadow birds (Klijn & Penning, 2002).

In spite of this control, conflicts occur because the drainage of agricultural land and the extraction of drinking water leads to a lowering of (ground) water levels resulting in drought damage to both nature conservation areas and natural values within agricultural areas. In some areas water is externally supplied in order to diminish this drought damage. However, the supplied water is mainly derived from the large rivers in the Netherlands, which have often a lower water quality than water from the nature areas itself, which are often dominated by clean carbon rich or brackish groundwater. This illustrates a dilemma for Dutch water managers: do we let the nature areas suffer from drought or do we accept the effects of water supply? Since the 1980s both nature and water managers ask attention for the desiccation problems in large parts of the Netherlands. According to the desiccation map of 2000 about 500.000 hectares of nature area is desiccated. This is 12% of the total nature area in the Netherlands.

Extreme events in the last decades have raised questions about the robustness of water management strategies in low-lying densely populated areas. For example, in the Netherlands the floods in 1993 and 1995 have raised the awareness of the vulnerability of living in a delta (Silva et al., 2001; Vis et al., 2001; Vis et. al., 2003). As a result of high water levels people along the rivers Rhine and Waal were evacuated. Areas along the Meuse were flooded. Inhabitants and companies experienced inconveniences and substantial damage. In addition, several periods of sustained rainfall during the 1990s caused problems for inhabitants and farmers in the terrestrial areas.

Climate change is expected to aggravate both problems flood and drought by an increase of precipitation surplus in winter and a decrease in summer (EEA 2004). Furthermore, climate change will increase the inter-annual variation of the river Rhine discharge: summer flow reduces and winter flow increases (Middelkoop et al., 2001). In terrestrial areas intensified rainfall in winter and spring may lead to water on surface level. In summer less precipitation is expected, resulting in lower (ground) water levels. In conclusion: in the future we have to deal more often with extreme situations with water excess and water shortage. A measure to diminish inconvenience by flooding and water stress is the storage of water. This requires room. However, space is scarce in densely populated countries and combining functions seems a valid remaining option.

Policy makers tend to select nature conservation areas to provide the room they need for water storage, as nature is generally regarded to suffer from water deficit and the economical damage in these areas is less than in build-up or agricultural areas. However, is nature really suffering from droughts and is water storage a solution for the drought problem in nature areas or does it creates extra stress resulting in less valuable nature areas. In other words: Do we know enough about the impacts of water storage on nature, to combine water storage and nature conservation in the same area? If we do not know enough what decisions can we make with the current knowledge? And what knowledge is lacking and cannot be missed? In order to assess effect of water storage on species composition we adopted a conceptual model based on site factors, consider likely changes in site factors and subsequent implications on vegetation types. Effects on fauna are not taken in consideration. Finally, we give suggestions about the combination of water storage with nature.

Site model

The adopted site model was developed for the prediction of ecological effects of water management interventions and restoration measures (Stevers et al., 1987; Klijn, 1997; Witte, 1998; Runhaar, 1999). It was based on the assumption that there is a clear correspondence between spatial patterns in vegetation and underlying processes like soil formation and flow of water. This relation is described in terms of ecotopes classified using ecological relevant site factors. An ecotope is defined as ‘a spatial unit that is homogenous in vegetation structure, stage of succession and in the dominant abiotic factors that determine the species composition of the vegetation’ (Stevers et al., 1987).

Site factors are environmental conditions which determine the occurrence and development of plants. The site model distinguishes conditioning and operative site factors, which influence the vegetation respectively indirectly and directly. Conditioning factors, like soil composition and hydrology, occur on a large scale and are determined by large scale processes, like climate, wind, sedimentation and geology. Conditioning factors on their turn influence operative factors, which correspond with a more detailed scale. For example the conditioning factor surface water regime influences the moisture content and may influence the nutrient availability through the transport of nutrients. Operative factors are in fact the physical characteristics in which plants live. Examples of operative site factors used in the conceptual model for terrestrial vegetation are: moisture regime, nutrient availability, acidity, salinity and dynamics (Table 1).

The species composition of the vegetation (ecotopes) is determined by operative factors. For terrestrial ecotopes also the vegetation structure is taken into account. With this characteristic, which is not a site factor, time (vegetation succession) and vegetation management can be taken into account in the classification. For aquatic ecotopes water depth is used as operative site factor in stead of moisture content. Conditioning factors, like soil type and hydrology, can be used for ecosystem classification in ecosystem types at a higher scale (e.g. ecozones, ecoseries etc.; Klijn, 1997).

A process chain of the site factor model is presented in figure 1. Each arrow presents a process-effect relation. The ecosystem classification describes the relation between the current abiotic conditions (site factors) and the occurrence of vegetation types. Hydrological models are used to determine the process-effect relation between water management and conditioning and operative site factors. The relation between operative site factors and the species composition of vegetation is determined with field research or by expert knowledge and is often part of ecohydrological or habitat models. These relations are generic in its region but as flora differs, different for other parts of the world.

Table 1. Site factors and classes used for the classification of terrestrial and aquatic ecotopes.

Site factors	Classes
Moisture regime (terrestrial) or Water depth (aquatic)	Wet, moist, dry or Shallow, deep
Nutrient availability	Low, moderate, high
Acidity	Acid, weakly acid to neutral, alkaline
Salinity	Fresh, brackish, saline
Dynamics	Sand drift, perturbation
Vegetation structure (only terrestrial)	Pioneer, grassland, tall herbaceous, dwarf shrub, shrub, forest

An operational version of the site factor model is available through the ecohydrological model DEMNAT (Witte, 1998, Van Ek et al., 2000) en Natles (Runhaar et al., 2003). The advantage of the site factor model method is that it integrates abiotic conditions and biotic characteristics of an ecosystem. As the ecotope types are described in terms of site factors, impact of water management measures on ecotopes can be determined by following the process-effect chain: measures determine conditioning factors, which determine operative factors, which influence the growth of plants and of course also the species composition. For example, the water management measure 'water storage' will have an impact on conditional site factors, like (ground) water level, seepage and water quality. These factors will on their turn influence the operative site factors, like moisture content, acidity, nutrient availability, and salinity, which will finally influence the vegetation composition.

A disadvantage of the current applications of the site factor model is the use average conditions of the site factors in terms of yearly mean value or the mean of the growing season. Moisture regime is the only factor including some dynamics. As water storage is often temporary, dynamics have to be included for determining its effect on nature. For example, one of the most striking characteristics of water storage is the moistening of the area as a result of higher (ground) water levels or flooding and afterwards the de-hydratation. Both processes have their own influence on site factors. Some site factors will also change only during the storage itself, while others might be affected structural. A temporary change of site factors does not clearly indicate a change in vegetation types on a long time scale.

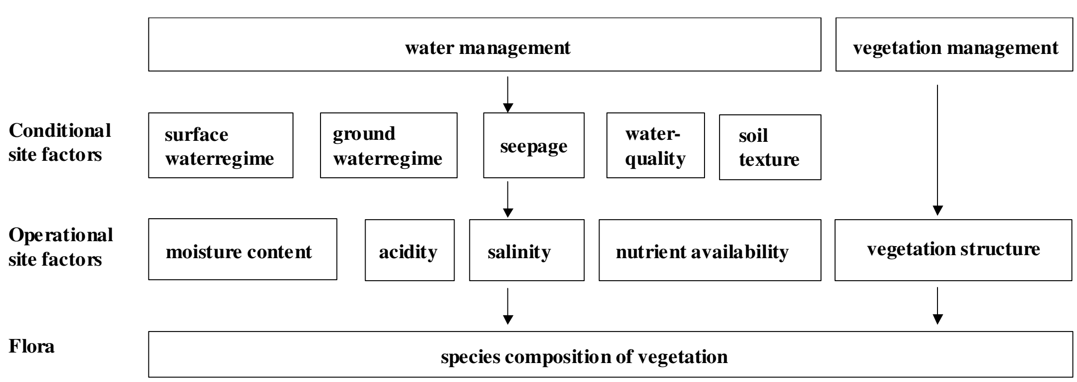

Figure 1. Conceptual site factor model.

The use of mainly average conditions in the current ecohydrological models is due to the correlative relations between site factors and ecological effects, which is determined through field experiences (data or expert judgement). Field experience is usually the result of several years of measurements or of measurements from steady states. Measurements after events are exceptional because they do not occur often and also science and society were not so interested in these effects. In the Netherlands, this knowledge is even more lacking as surface water levels are regulated within cm's and consequently lost the natural dynamics in most of the areas. The lack of hydrodynamics in the models was not much of a problem as the models were developed to assess long-term effects of water management and land use strategies on nature, which did not include large hydrodynamics.

The challenge in using the adopted conceptual site factor model for assessing the effects of water storage on vegetation can be found in the description of the hydrodynamics and in the formulation of process-effect relations between these (temporary changed) site factors and vegetation (the latter is probably the most difficult part). This might need the model to be extended.

Water storage strategies

Before we can assess the effects of water storage we must first consider the characteristics of water storage. In general, two purposes of water storage on vegetation can be distinguished:

- **flood control**: temporary controlled storage of water from heavy precipitation or from peak river discharges to prevent flooding elsewhere.
- **water conservation**: preservation of water in periods with a water surplus in order to have this available in periods with water shortage.

Within these two purposes we can distinguish four water storage strategies (Penning and Duel, 2003). These strategies have each their own characteristics in terms of origin of water, flood duration and timing, resulting in different effects on vegetation (table 2; figure 2).

Table 2. Characteristics of different water storage types.

	Water storage type	Strategy	Frequency (events/year) and duration	Place	Water level fluctuations	Relative water quality
water conservation	**Summer storage**	Short and local storing of precipitation especially during summer to prevent intake of eutrophic water for other parts of the system	Many times per year; a few days to weeks inundated	in surface and ground water	a few centimetres till decimetres (often below ground level)	high
	Winter storage	Storing water during winter half year to prevent water shortage in agricultural and nature areas in summer half year	Every winter half year ; many months continuously inundated	in special designated areas or ponds	half to several meters above ground level	high
flood control	**Excess storage**	Temporary storage of water to prevent flood risk in other parts of the system, by stopping pumping, or flooding of low lying polders to prevent damage in other areas	Annually – 1: 100 years. Each event lasts a few days	in surface & ground water and above ground	few centimetres till decimetres above ground level	low
	Emergency storage	Temporary storage by water inlet from rivers or main channels to prevent serious flood damages	1:100- 1:10.000 years; a few days till weeks.	in special designated areas	one to several meters	very low

The summer storage strategy involves short and local storage of water in the summer half year. In other words, during summer the water level regime is more flexible, by allowing the water levels to rise before the pumps are activated and water is drained to get back to the target water level. As the stored water is derived from precipitation, the water quality is relatively good. The water level rise is around cm to decimetres in summer and is often below the ground level.

Winter storage focuses on saving precipitation in winter to prevent shortages in dry periods (summer). According to Penning and Duel this water is used in nature areas and for agricultural purposes. In the latter case, effects on vegetation may be different. Water levels may rise decimetres to meters and can get above the ground level, especially in winter. Both water conservation strategies store water on an annual basis.

Excess and emergency storage aim at preventing water damage because of high water levels in the (often large) rivers. Consequently, this will occur in periods with heavy rainfall in the whole river basin (winter and early spring). During high water in the river, water is let into the storage area reducing the high water peak. The flooding duration will be as long as the high water on the river continues and the time needed to empty the area (few days to weeks). The excess storage strategy is less frequently and involves less water, while the emergency storage is much more exceptional including large areas of flooding. Both involve storage of water above the ground level.

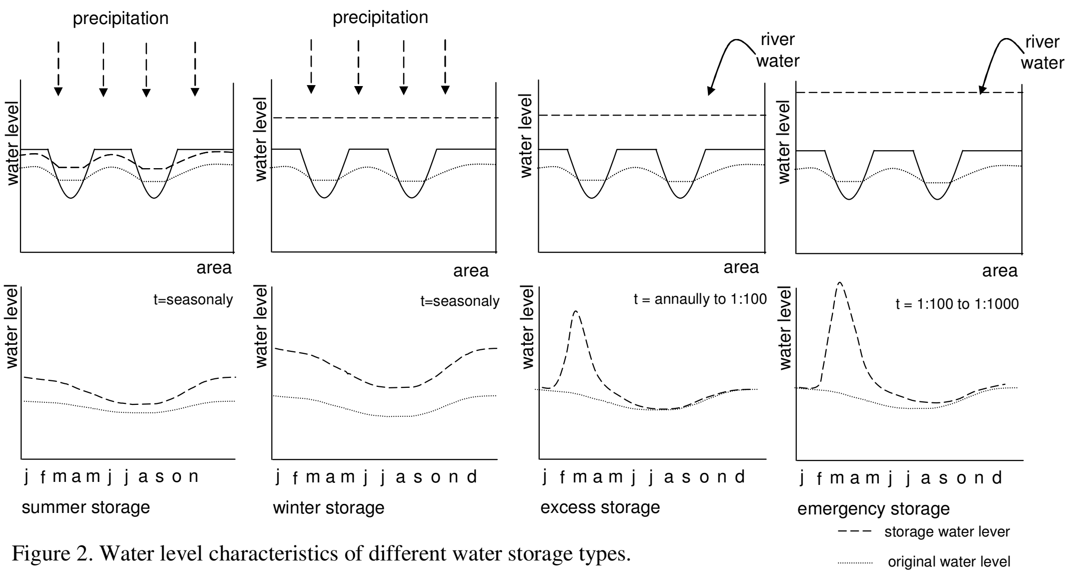

Figure 2. Water level characteristics of different water storage types.

Effects of water storage on site factors

Now we defined the characteristics of water storage in terms of hydrology and time aspects, we can describe the effects of flooding on the specified site factors.

Moisture content

In terrestrial areas, water storage increases the moisture content through a higher groundwater level or through inundation. This results in an **increase of the moisture content** at the expense of air (= oxygen). The degree of this increase in moisture content depends on the final (ground) water level and soil characteristics. A soil with small particles (e.g. fine sand and clay) has a larger capillary rise of water than a soil with large particles (e.g. coarse sand), resulting higher moisture contents above the groundwater level. Large amounts of storage water may eventually lead to inundation. In aquatic areas, water storage results in a **water depth increase**.

The timing of these wetter conditions depends on the start of storage period, which is obviously under (extreme) weather conditions like an intense or long period of rainfall, and stops either directly after or during dry periods in case of summer storage. In other words, the moisture content and water level regime change and more in particular, they will be **more dynamic and higher**. Higher (ground) water levels will occur during the water storage period, which is in winter and spring for winter, excess and emergency storage. The duration varies from days (excess storage), to weeks (emergency storage) to months (winter storage). Furthermore, very dry conditions may occur less in case of summer storage. For winter and summer storage this is a frequent situation (yearly), while excess and emergency storage are events and occur occasionally.

Increase of moisture content through inundation or an increase of the (ground) water level is the most visible result of water storage, which influences vegetation directly as water is a crucial factor for the life of plants. Indirectly, an increase of moisture content has also some other effects, which we will discuss further below.

As the diffusion of oxygen is much slower through water-filled pores than through a porous soil, flooding or a groundwater level close to surface level finally leads to an **oxygen deficit** in the root zone. This starts after several hours to a few days after inundation and sometimes after 22 days (Gambrell & Patrick, 1978; Savant & Ellis, 1964; Meuleman, 1999). The actual rate of oxygen reduction depends on temperature and the amount of organic material for microbial respiration. At temperatures above 5 ^{o}C bacterial decomposition and plant activity increase, resulting in an increase of oxygen use. The anaerobic conditions prevent plants from normal respiration.

In the absence of oxygen other substances are used as oxidator or in other words as catalyst for the decomposition of organic material (Mitch & Gosselink, 2000). The following substances are respectively used as oxidator: nitrate (NO_3^-), mangaic (Mn^{4+}), ferric (Fe^{3+}), sulphate (SO_4^{2-}) and carbon dioxide (CO_2). First nitrate is used as oxidator resulting in gaseous nitrous oxide (N_2O) or nitrogen (N_2). In addition, mineralisaton of organic nitrogen forms ammonium (NH_4^+) both under anearobic and aerobic conditions. Nitrogen is followed by the reduction of iron and manganese. Insoluble ferric iron is reduced to the soluble ferrous iron, while manganic is reduced to manganous. After longer periods of flooding sulphate (SO_4^{2-}) is used and converted into sulphide (S^{2-}) or hydrogen sulphide (H_2S). Finally, under extremely reduced conditions, bacteria (methanogens) form methane (CH_4) from carbon dioxide during the methanogenesis process (Mitch & Gosselink, 2000). These processes will influence the macro-nutrients availability (N, P and K).

Some of the substances formed during the above mentioned processes reach potential **toxic levels**. Toxic effects of ammonium may be reduced by immobilisation binding on negative loaded soil particles like in clay. Manganese and iron are both toxic in their reduced form. Sulphide and hydrogen sulphide are toxic through immobilisation of zinc and copper and reduced availability of sulfur due to precipitation with metals (Ponnamperuma, 1972). The toxicity of sulfide can be reduced after immobilisation through binding with ferrous iron in iron-rich areas. The presence of these toxic substances occurs probably only during the flood period as after dehydration the upfide oxydizes again (Lucassen, 2003).
Besides a temporary change, a more structural change of moisture content may locally occur as well. Inflow of sediment together with inlet storage water, in case of excess and emergency storage, can change the surface height and soil texture and consequently the (ground) water levels and groundwater flow. When an area is flooded, dissolved sand and clay particles precipitate after the stream velocity decreases. Under natural circumstances this corresponds with natural embankments, but as a result of water storage this occurs only very locally close to the inflow of the water. This process is therefore not further considered.

Acidity

The acidity decreases (and pH increases) if during anaerobe conditions other substances are used as facilitator for the decomposition of organic material. This may result to neutral pH conditions even in acid soils after approximately two weeks (Ponnamperuma, 1984; Lamers et al., 1998). Furthermore externally supplied water, in case of excess or emergency storage, often contains higher concentrations of bicarbonate which may increase the pH (and decrease the acidity). The acidity influences site factor the nutrient availability through its effect on the velocity of the mineralisation process; a low acidity reduces mineralisation and a high acidity increases the microbial activity, resulting in an increase of the nutrient availability. Van den Brink et al. (1993) found higher concentrations of bicarbonate (1,5 to 2 times larger) in floodplain lakes fed with seepage water than in lakes which were frequently flooded, indicating that flooding with water from the large rivers increases the acidity.

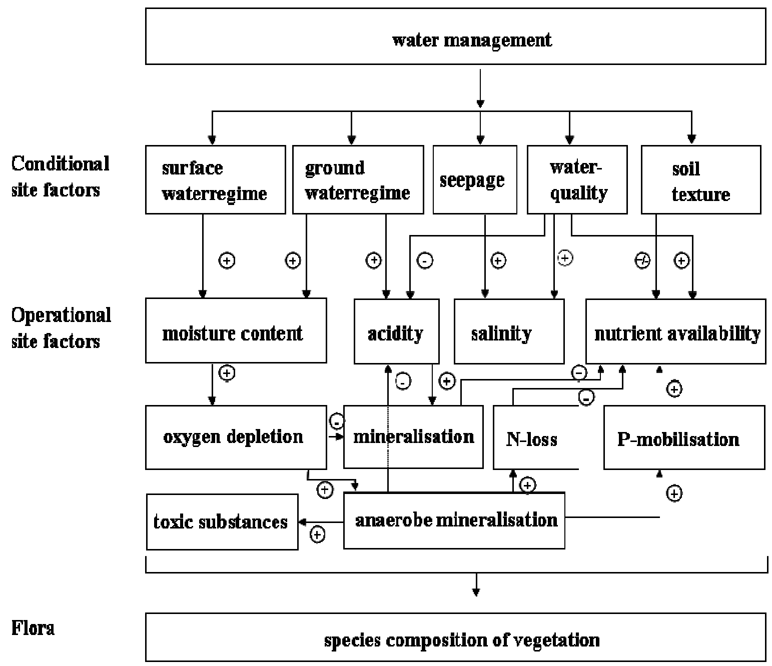

Figure 3. Effects of water storage in the site factor model. Example changes in site factors due to water storage. This is a general overview which is more applicable to excess/emergency storage than to summer storage. Increase of surface and ground water result in increase of moisture content. After a while, this results in oxygen depletion. Other substances are used as oxidator, resulting in a higher pH and decrease of the acidity. This stimulates mineralisation, which increases nutrient availability. Anaerobe mineralisation mobilises P due to binding on soluble Fe3+ iron instead of insoluble Fe2+ but results in loss of N. Externally supplied water can introduce more nutrients. Also bicarbonate rich water stimulates mineralisation, while sulphate or calcium rich water can stimulate P-mobilisation. Less bicarbonate rich groundwater from seepage and relatively more input of rain water, can decrease pH and diminish mineralisation. Storage of rain water, in case of winter or summer storage, often results in a higher acidity, which reduces mineralisation.

Salinity

Externally supplied water may have a higher or lower salt concentration than the original water, resulting in an increase of the salinity. Higher (ground) water levels decrease the influence of upward seepage surface water and in the root zone. In areas with brackish upward seepage, this can change the operational site factor salinity. In lakes frequently flooded with water of the river Rhine (67 days/year) the chloride concentration can be 1,5 to 4 times greater than in respectively lakes and ponds which were not flooded (Van den Brink et al. 1993).

Nutrient availability

Nutrient availability is the last operational site factor that can change due to water storage. The effects on this site factor are more complicated to estimate as it is (in) directly influenced through several paths. The most obvious cause is the inflow of nutrients through externally supplied storage water, either in dissolved form or attached to sediments, which are often referred to as **external eutrophication**. Part of the dissolved matter will precipitate ones the stream velocity decreases. This type of effect is obviously only the case if water is stored from outside the storage area, for example in case of excess storage or emergency storage. Extra supply of dissolved nutrients may be typical temporal effect as most of the dissolved nutrients will flow out of the area after it is emptied. Lakes frequently flooded by the river Rhine in the Netherlands had 4 to 9 times higher amounts of dissolved phosphorous and 10 times nitrate than lakes which were mainly supplied with rain and seepage water (Van Brink et al., 1993). Van der Lee et al. (2004) found that nutrient retention from the river Rhine was insignificant for nitrogen (<3% of total load) but the phosphorus retention was around 18%. For both nutrients sedimentation was the most important mechanism instead of uptake by plants, denitrification and binding to soil. For lower Cosumnes river in California the effectiveness of the floodplain reducing nitrate was limited to the 0.6-4.4% of the annual N load during a typical wet year (Sheibly et al., 2006). A model simulation of Baptist et al. (2006), taking into account elevation, drainage capacity and flow velocities indicated that 30% of the suspended sediment and adsorbed phosphorus that enters the retention area during an extreme flood (1:100 years) along the river Sava (Lonjsko Polje) deposits in there.

Spatial differentiation in substrate types as a result of sedimentation during the flooding as described above will also influence the nutrient availability. Coarser sand subsiding closer to the water inlet contains fewer nutrients than finer soil elements subsiding more land inwards.

Indirectly the nutrient availability may also change as a result of the oxygen depletion as described before. The availability of nitrogen may be reduced if the inundation (or groundwater level at surface level) is long enough to result in oxygen depletion as the denitrification results in a **loss of nitrogen** to the atmosphere in the form nitrogen gas (N_2 and N_2O) (Mitch & Gosselink, 2000, Wienk et al., 2000). Temperature influences the activity of bacteria and consequently the velocity of denitrification.

The availability of phosphor can change indirectly as well. Phosphor is often adsorbed to soil particles together with Fe, Al, Ca and Mg. If iron is reduced during the anaerobe decomposition the iron-phosphor combination will loose its connection to soil particles dissolve in water (Khalid et al., 1977, Lamers et al., 1998). Caraco (1984) found a strong relation between phosphor release from sediments with sulphate concentrate of surface waters. The sulphate or calcium rich storage water can enhance this process as both substances can replace phosphor (Lamers et al., 1996, 1998). This process is often referred to as **internal eutrophication.** Sulphate is an extra stimulator of the mineralisation process as the reduction of sulphate results in more bicarbonate increasing the alkalinity, and it results in sulphide which replaces phosphate from its binding with iron, resulting in more soluble and available phosphate (Lamers et al., 1996, 1998; Roden & Edmonds, 1997; Lucassen et al., 2003). This effect is expected to be temporarily and not immediately. For a peat area in the Netherlands a ten-fold increase of the phosphate concentration was measured after two weeks of flooding. Flooding with sulphate- rich water increased the phosphate concentration more. After two weeks a significant increase due to sulphide was measured (Lamers et al., 1996). A study along the floodplain of the river Overijsselse Vecht confirmed the raise of phosphate caused by reduction of iron. The increase of phosphate was only measured during the inundation. After desiccation the phosphate concentrations became low again (Loeb & Lamers, 2003).

Both nitrogen and phosphor are released during **mineralisation**. Under anaerobic conditions the decomposition of organic matter (mineralisation) will be slower; up to half as fast as when oxygen is used (Updegraff et al., 1995). Close to root cells, the opposite process may occur through oxidation of ferric to ferrous iron as a result of oxygen leakage, resulting in the immobilisation of phosphorus. The velocity of mineralisation also depends on the temperature, composition of the organic matter and pH. In summer the mineralisation rate is higher than during winter. An increase of temperature from 5 to 20 ^{0}C can raise the anaerobe microbial reduction during flooding and consequently the phosphor availability significantly (Loeb & Lamers, 2003). In aerobe soils the mineralisation velocity doubles at a temperature increase of 10 ^{0}C (Reddy & Patrick 1984; Koerselman et al. 1993). This process lies almost still at temperatures below ^{0}C. In winter the effect of water storage on mineralisation is probably very low. The optimal pH for mineralisation is neutral to basic (between 6.5 and 8.5) (Wienk et al., 2000; Reddy & Patrick, 1984) A decrease of the acidity (to low pH), decreases the microbial activity (McKinley & Vestal, 1982). The shift to other electron acceptors after oxygen depletion due to anaerobe conditions generates alkalinity (Ponnamperuma 1984). Moreover, a higher pH resulting from the anaerobe decomposition processes stimulates the mobility of phosphate.

As acidity influences mineralisation, this process is also affected by amount and composition of externally supplied surface water and the inflow of groundwater, like described earlier. Externally supplied water can increase the acidity through the **introduction of bicarbonate**. This is often the case when water from our large rivers is used (Lamers, 1996; Wienk, 2000). Consequently, the quality of the used water for storage determines also the nutrient availability; in case it is alkaline, it enhances mineralisation. However, if the amount of inlet water is very large resulting in flooding and anaerobe conditions, the mineralisation is slowly anyway.

Externally supplied water can also introduce extra nitrate and sulphate, stimulating the possibilities for mineralisation partly because they can take over the role of oxygen as oxidator.

The extra pressure of the storage water can reduce the inflow of **groundwater** including the dissolved substances. One of these substances is bicarbonate which originates from the dissolution of calcium under the influence of carbon dioxide. Less inflow of calcium rich groundwater can therefore reduce mineralisation. Another substance delivered through seepage water is iron. Less influence of groundwater may therefore decrease the iron content and lower the insoluble phosphate-iron bindings, resulting in more soluble phosphor available for plants. In addition, the iron dissolved in upward seepage can immobilise the phosphorus again through binding with ferrous iron. This is not applicable during anaerobic conditions, so this is probably more applicable to winter and summer storage. Furthermore, high nitrate concentrations in groundwater restrict the mobilisation of phosphate as nitrate is an energetically more favourable electron acceptor than ferric and sulphate (Lucassen et al., 2003). High nitrate loads function therefore as a redox buffer limiting the reduction of ferric and sulphate. This diminishes the binding of sulphide to reduced iron, leaving insoluble iron-phosphate connection intact, reducing internal eutrophication. Nitrate may also oxidise ferrous iron to ferric iron capable of binding with phosphate. The buffer of high nitrate conditions and the oxidation can however also limit the amount of soluble ferrous iron, needed as nutrient for plants (Smolders et al., 1997). However, if sulphate concentrations are high, due to the concentration in the water or due to the nitrate induced oxidation of pyrite, eutrophication may still occur as result of sulphate reduction. These effects through groundwater are relatively small compared to the amount of externally supplied water as the amount of water is much less. It can be more important in case rainwater is stored during summer under dry conditions. Together with the more acid rainfall, less groundwater, under wetter but still aerobe conditions the mineralisation may be reduced.

The extent of increased nutrient availability due to internal eutrophication and mineralisation depends of course on the soil type: soils with high amount of organic material have more potential for nutrient availability.

Summarizing, the nutrient availability can change either directly through inflow of nutrients or indirectly through redox and mineralisation processes. Some effects increase and others decrease the nutrient availability. To assess the total effect, we should consider the different types of storage types separately. In case of excess and emergency storage, the nitrogen availability may temporary increase through inflow of nutrients from the externally supplied water or may decrease by anaerobic oxidation processes. Phosphorous may increase through mobilisation and inflow of nutrients. This effect is reduced by slower mineralisation under anaerobe conditions. The total effect is probably a temporary increase of nutrients during storage, part of it precipitates and is left there after storage, but after a while these nutrients are taken by plants and washed out by the rain. After some time the system will return to its original conditions. On the other hand, if the flooding is more frequently the system has no time to recover. This might be the case for excess storage, which in a way is similar to parts riparian floodplains where the vegetation is kept at an early succession stage.

In case of winter or summer storage, local water is stored. This is either precipitation surplus (polders) or water from upstream the catchment (rivers). Winter storage results in inundation during winter and probably also during spring. In summer the groundwater levels should be higher. Here, the effects as a result of the anaerobe mineralisation will be relatively more important than the inflow of nutrients as mentioned for excess and emergency storage. Phosphor availability may increase. The availability of nitrogen depends on whether rainwater is stored or water from a catchment which collected nutrients on its way. If it is only rain water, then the availability will probably decrease, otherwise it may increase or stay equal. In case of summer storage, rainfall is stored in the ground, resulting in higher groundwater levels, but still mainly aerobe mineralisation. The nutrient availability will probably not change much. Mineralisation could decrease a little as a result of lower pH (through rainfall) and higher groundwater levels. But this could be compensated by the increase of P mobility after a lower acidity.

Effects of water storage on vegetation composition

Effects of changes operational site factor depend on timing, duration and depth of the inundation on the one hand and the changes in site conditions and the (adaptation of the) current vegetation on the other hand. The above described conditions differ from average conditions on which most process-effect relations are based. This makes it difficult to use the available relations for impact assessment of water storage on ecosystem types. We will describe effects of water storage by looking at existing situations with comparable (hydro)dynamics and vegetation and describe effect for hydrological interesting water storage and nature areas: river floodplain wetlands and brook valley wetlands.

River floodplain wetlands

Along rivers it is more likely to store water to prevent flooding than to conserve water for dry periods, like excess and emergency storage. The water storage characteristics to estimate the effects are therefore: large amounts of water occurring in mostly in winter or early spring with a low frequency. Site factors corresponding with river floodplain wetlands involve wet to dry, neutral and (moderate) nutrient rich conditions.

Flooding is a dominant site factor determining the distribution of plant species in river areas (van Eck, 2006; Skyrira et al., 1988; Crawford, 1992; Vartapetian & Jackson, 1997; Pollock et al,. 1998; Van de Steeg & Blom, 199; Vervuren et al., 2003; Van Geest, 2005). Flood tolerant species occur at low elevations with high flood durations, while flood intolerant species occur at highly elevated areas (van Eck 2004,Vervuren 2003). Flood duration is therefore often used in environmental impact studies on water system changes in river floodplain areas. Often the total number of flooded days during a year is used to determine the distribution of vegetation types for example in ecological modelling (Baptist et al., 2006; Haasnoot & van der Molen, 2005). In the site factor model, this involves the moisture content and the water depth.

Due to lack of oxygen, one of the fundamental requirements of plant life, anaerobe conditions resulting from flooding will after a while result in deterioration and eventual death of plants. A longer the inundation, results in more damage and less survival of plants. During summer, and more in particular in the growing season, effects of flooding are more significant than during winter (van Eck et al., 2004; Van Eck, 2006; Blom et al., 1990; Klimesova, 1994; Sparks et al., 1998). In the winter period the impact on the survival of plants is less as they are not active during this period (Brock et al. 1987). Extreme events in summer, e.g. as a result of water storage for flood prevention, determine the lower distribution of species along a river for a long time. For example, the effects of a major summer flood 12 years ago were still visible in the distribution of grassland species along the river Rhine (Vervuren et al., 2003, Van Eck et al., 2006). This is probably due to the limited ability to re-colonise lower positions in the floodplain. Winter flooding, which is more regular, influences the distribution of very intolerant species. This is not applicable for species which can rapidly colonise new areas like bare ground created by floods. Upper limits of plant distribution along rivers are expected to be determined by drought conditions (van Eck et al., 2006; Lenssen et al,. 2004).

Besides flood duration and timing also water depth and under water light intensity (determined by suspended load and water depth) are important environmental factors determining the effects of flooding (Vervuren et al., 2003). Deep water diminishes possibilities to achieve aerial oxygen, while turbid water limits under water photosynthesis. Inflow of nutrient poor groundwater may diminish or compensate the effect of nutrient inflow due to externally supplied water. This depends on the frequency and the amount. Furthermore, helophytes, brushwood and forest can remove nutrients from the water, creating chances for nutrient poor sites further away from the river and enhance biodiversity (Wassen et al., 2003). For this effect large floodplains, which are absent in densely populated deltas, are needed.

Effects of flooding are species-specific; some species are better adapted to flooding than others. First species normally growing on (very) dry site factors will die, followed by species of moistly conditions. The tolerance of species to anaerobic conditions can vary from a few hours to many days or weeks depending on the species (Vartapetian & Jackson, 1997). Adaptations to anaerobic conditions are either metabolic, morphological or concerned with reproduction. Reeds and bulrushes have air tubes in roots and stems to transport oxygen from the air to the roots and other parts of the plant (aerenchyma). Also, these tubes are used to create a oxygen rich environment around the roots in order to detoxify soluble reduced substances by oxidation and precipitation in the rhizosphere. The floating plant water lily derives its oxygen through air transport through young leaves via aerenchyma downwards into the plant and returns through the older leaves. Several aquatic plants and some grassland species are able to elongate there shoots to reunite shoot with air after an increase of water depth (Mitch & Gosselink, 2000; Visser et al., 2003; Sman et al., 1993; Blom et al., 1990). Small plants in early summer flooding have less capacity to reach air again, which is contributing to the fact that the timing of the flood is important for the eventual effect. Renewed contact with air can stimulate the development of aerenchyma root system in Rumex species (Blom et al., 1990). In many trees and herbaceous species (e.g. willow) the hormone ethylene stimulates the growth of extra roots above the anaerobic zone (Mitch & Gosselink, 2000).

Metabolic adaptations involve the access of stored carbohydrates for energy production when submerged (Crawford 2003) and under water photosynthesis (Vervuren et al., 1999; Vartapetian & Jackson, 1997). Outside growing season, when temperature is lower, the carbohydrate depletion is slower, resulting in an extension of survival.

Another mechanism plants use to survive flooding is to change their timing and reproduction by either the delay of flowering and seed production during flooding and survive as a vegetative plant or by acceleration of flowering during short dry periods to produce seeds in short intervals between successive floods. Sometimes flowering occurs after flooding but the seed production is then reduced (Sman et al., 1998).

If species lack such adaptation or change to anaerobe metabolism, which is a very inefficient process with regard to carbohydrate consumption, biomass will strongly be reduced (van Eck et al. 2004).

In the next part, we will describe results of literature review for species occurring in four characteristic vegetation types of floodplains (backwaters, grassland, forest).

Exceptional flooding of floodplains along the river Rhine during the summers (May-July) of ‘70, ‘78, ‘80 and ‘83 resulted in changes of the aquatic and marshland vegetation (Van de Steeg, 1984; Brock et al., 1987). *Nymphoides peltata* disappeared, while a slight decrease of *Iris pseudacorus*, *Nuphar lutea* and *Nymphae alba* was noticed. Although the *Nymphoides* were the first to reach the surface of the water with their leaves by elongation, the leaf stalks were so fragile, that they washed away under normal wind and wave conditions. During flooding, the growth of seedlings of *Nuphar* and *Nymphaea* is largely inhibited due to shading by turbid water or by floating leaves of full-grown species (Guppy, 1897). Several helophytes and marshland species, like *Typha angustifolia, Sparganium erectum, Equisetum fluviatile, Ranunculus lingua, Scirpus lcaustris, Scirpus maritimus* were diminished in abundance. Most of them disappeared after 3 weeks of flooding (Brock et al., 1987). The depth of flooding (3 m) was so high that the aerenchyma of *Typha* and *Scirpus* species did not help them to survive as the oxygen is consumed very quickly (Sale and Wetzel, 1983). *Glycerina maxima* and *Phragmites australis* showed a reduction in vitality, but they could maintain themselves. *Carex acuta* and *Butomos umbellatus* had benefit from the flooding and increased its area. External eutrophication can change transparent floodplain lakes into an algae dominated system causing the deterioration of macrophytes.

For floodplain grassland species results of inundation experiments on survival of these species are available (Sman et al,. 1993; Van Eck et al., 2004, 2006; Vervuren et al., 2003). These experiments show that the amount of days where most of the species died (survival < 20%) during summer conditions varies between 10 to 50 days for species growing on high elevation to 50 to > 60 days for species growing on a low elevation (Van Eck et al. 2004, van Eck et al. 2006). Under winter conditions the amount of days ranged from 20 to > 100 days (Van Eck et al. 2006). During these simulations air-leaf contact was prevented through a large enough water depth and the water was kept clear. A larger light intensity and shallower water enhances the survival (Vervuren et al., 2003). On average the survival time was diminished with 50% when plants were completely submerged (1.6 m) compared to shallow water (0.4 m). The largest increase of survival rate occurred at low light intensities, but there was a large difference in the reaction of species. *Rumex crispus* was found extreme tolerant to submergence with turbid water, which is probably related to anaerobic energy metabolism.

Floodplain forests are often adapted to a certain amount of flooding. Willows can produce adventitious roots during periods of inundations. These roots are able to take up water and adsorb oxygen and nutrients from the river. Seeds of floodplain forest are released late spring or early summer. They thrive in moist bare zones exposed by the receding of the river. It is therefore critical that floods occur just before seed release, too early or too late will reduce survival of seedlings.

Field research indicates that some species are adapted to flooding events, but a lot of plants are more or less flood sensitive; they either dye or have severe damage. However, regular flooding of backwaters by river water in winter or early spring is needed to maintain aquatic and marsh vegetation, as winter and spring flooding prevent a succession towards a marsh vegetation. In addition, a lake drawdown is favourable for the distribution of some species. During extreme low water levels, the soil becomes dehydrated, increasing the consolidation. An increased sediment density stimulates conditions for vegetation, as highly fluid sediments impede establishment of propagules and the anchoragement of rooted macrophytes (Van Geest , 2005). Furthermore, this exposed bare ground stimulates the of seeds of long emergents (ter Heerdt and Drost , 1994). However, seeds of *Nuphar* and *Nymphaea* species are killed after several months of desiccation. Forests need some hydrodynamics for germination of seeds. So hydrodynamics are needed to keep the vegetation characteristics of the river floodplains.

Brook valley wetlands

Brook valleys are geographically too far away from large rivers for emergency storage. As flash floods do not occur in low lying countries, emergency storage is not considered here. Winter or summer storage are the best water storage options to take into account. Although some areas might also be applicable for excess storage store water from the main drainage channels ('boezem') or to conserve water by preventing fast drainage in hilly areas. Storage conditions to consider, range from few cm groundwater level increase for a couple of times a year to an increase of the water level to dm above ground level once a year or less often.

The vegetation in brook valleys exists of brook forests with *Alnus glutionosa* in the higher parts of lowland brooks, hay meadows with *Calthion palustris*, fen meadows with *Cirsio-Molinietum* and *Caricetum nigrae* and terrestrializing vegetation of shallow water. The latter areas occur in peat areas in the western part of the Netherlands, which were created by peat harvesting during preceding centuries. This resulted in shallow lakes and rectangular parcels with water areas in between. In some areas the shallow lakes are reclaimed by drainage systems. Nature in brook valleys is the nature suffering from desiccation. Site factors for the vegetation types are most to wet, (weakly) acid and nutrient poor to moderate nutrient rich.

Flooding is not a rare phenomenon in these areas. The wet anaerobe often acid conditions caused the development of peat and its vegetation. Also, from historical and paleo data we know that these areas are strongly influenced by flooding (Van Diggelen et al., 1991). Nowadays we can still find similar conditions along e.g. the Biebzra floodplain in Poland. Here, *Scirpo-Phragmitetum, Glycieretum maximae, Caricetum actae, Caricetum elatae, Calamagrostietum* vegetation and also species characteristic of nutrient-poor conditions, like *Menyanthes trifoliata*, *Potentilla palustris, Carex diandra, Parnassia palustris* are flooded in spring with river water (Wassen, 1996). The latter mentioned species were found in higher areas further away from the river, which are less frequently and less deep flooded.

Aquatic systems

The ranges of water quantity changes as a result of water storage are within the range of the natural conditions. Effects of changes in water quality are in these areas a more important factor to consider when analysing the possibilities for the combination of nature and water storage. We will describe these effects first for aquatic and than for terrestrial species.

For aquatic species winter or summer storage by conserving rain water seems like natural conditions. The water depth may increase but this is probably small en most species can keep up with this increase by elongation. The change to more acid conditions is probably neglectable as the amount of rain water is relatively. Excess storage, however may lead to an increase of the nutrient availability through internal and/or external eutrophication. This may result in turbid water through algae blooms, the decline of characteristic species like *Stratiotes aloides* and *Potamogeton compressus* and an increase of eutrophic species like *Lemna minor* (Smolders et al., 1996; Smolders et al., 2003). These effects were measured after the use of water from outside the nature area, which is comparable to the conditions under excess or emergency storage (because of the externally supplied water). Besides eutrophication also toxic effects of sulphide are measured in aquatic systems. Roots of *Stratiotes aloides* and other species were deteriorated by accumulation of sulphide and died (Smolders et al., 1996). The toxic effect of sulphide can be diminished by binding on ferrous iron. However, in this form iron is unavailable for plants, raising the risk on lack of iron, which may lead to damage of the plant as a result of chlorosis (yellowing of leaf tissue due to a lack of chlorophyll). For example, leaves of *Stratiotes aloides*, *Alnus glutinosa* and *Juncus acutiflores* turned yellow as a result of the lack of iron by the binding on sulphide (Smolders et al., 1996; Smolders et al., 1997; Lucassen et al., 2003). Upward seepage water, which often contains ferrous iron, can diminish this effect.

The magnitude of the effect of externally supplied water depends not only on the water quality and quantity but also on the inlet place and the route. In case of water storage in ditches, it takes some time for all the original water is replaced or mixed with the storage water. A study in nature area 'de Meije' indicated that the storage water was spread through the whole area only in July and August, while storing water since spring (Meuleman et al., 1996). Close to the inlet the phosphate concentration was high, and further away the concentration decreased, probably because of the uptake by plants. The vegetation reflected this pattern: in the first part of the ditches system eutrophic floating species like *Lemna gibba* and *Limna minor* were found, than eutrophic submerged species like *Lemna triscula* and in the last part of the system species characteristic for nutrient poor to moderate nutrient rich conditions like *Scirpus fluitans, Hottonia palustris, Myriophyllum verticallatum* were found.

Terrestrial systems

The terrestrial areas may suffer from acidification as a result of the conservation of rain water in case of winter or summer storage. Species characteristic for wet alkaline nutrient poor conditions like *Viola persicifolia, Cirsium dissectum, Dactylorhiza majalis*, *Orchis morio* and hay meadows will decline. On the other hand these are good conditions for species characteristic of wet acid nutrient poor conditions like *Molinia caerulea, Eriophorum angustifolium* and *Sphagnum* species. Flooding with polder or river water or upward seepage water used to neutralise the groundwater, but nowadays this does not occur anymore due to leakage to surrounding areas. In areas with enough upward seepage water with bicarbonate the groundwater can be neutralized as well. The introduction of externally supplied water in case of excess storage, could mimic this flooding or raise the groundwater levels. However, as the inlet water is often phosphate and sulphate rich this can cause eutrophication. In the nature area 'de Meije' neutralised the groundwater within a few meters from the ditch, saving the alkaline species, although eutrophic helophytes were decreasing the habitat from the ditches site (Meuleman et al., 1996).

The effect of sulphate rich water on the biomass of six species groups was measured by Lamers et al. (1998). This was done for flooding with and without sulphate enriched and effects after several weeks. A decrease of biomass was noted for Carex spp. After 21 and 32 weeks of flood with sulphate enriched water. The same accounts for Graminae spp. Although for these species also the waterlogging itself was decreasing the growth. Galium spp. increased growth after 13 weeks but sulphate rich water decreased it again.

For brook forests we did not find any literature on effects of flooding. Under natural conditions these forests are adapted to flooding. So, if the water storage results in natural conditions (some flooding in winter and spring and aerobe conditions in summer), we do not expect any severe effects.

Concluding, flooding in spring corresponds with natural conditions in brook valleys. Winter and summer storage can diminish desiccation problems in areas with species characteristic for acid conditions. However, most valuable areas are the ones with alkaline species. In these areas the stored rainwater needs to be neutralized with river or upward seepage water.

Discussion

What are the effects of water storage? In order to assess effect of water storage on species composition we adopted a conceptual model based on site factors, consider likely changes in site factors and subsequent implications on vegetation types.

Following the site factor concept, we learned from the review on the relationships between quantitative and qualitative (ground) water variables and biota that changes in site factors as a result of water storage can have an effect on the vegetation composition. The raise of the (ground) water level increases the moisture content resulting in anoxic conditions. The nutrient availability is affected through several ways, either directly through water inlet or indirectly through mineralization. The total effect depends on the water level increase and the origin of the water: rain or river water. If it is only rain water then the total nutrient availability may be either increase through P-mobilisation, stay equal or decrease as a result of slower mineralization under (partly) anaerobe acid conditions. River water introduces nutrients and may increase the nutrient availability through anaerobe mineralization, which is extra stimulate through sulphate-rich water.

The vegetation composition will change in most cases. The effects of water storage are species specific and depend on the storage period, duration, water depth, frequency, water quality and the conditions the species are adapted to.

A decline of species richness is then the obvious result. A decline of calcicole species can be caused by increased influence of rain water in case of summer storage. Species characteristic for nutrient-poor conditions decline in number in case of excess and emergency storage. This may also account for fresh water species. On the other hand species characteristic for nutrient-rich conditions may come in replacement, although they probably already occur somewhere close. This corresponds with other studies which indicate that species richness declines with the increase of dynamics through disturbances in environmental factors (Gosselink & Turner, 1978; van den Brink 1994; Weiher & Keddy, 1995; Pollock et al., 1998; Casanova & Brock 2000). The effect of disturbance by flooding is not only substractive but also additive; other species were stimulated under subsequent conditions (Casanova & Brock, 2000).

In general, species richness along rivers increases with height and is higher at lower inundation duration and lower water depth (Gosselink & Turner, 1978; van den Brink, 1994; Wassen, 1996; Casanova & Brock, 2000). A high spatial variation of flood frequencies results in high species richness (Gosselink & Turner, 1978; Pollock et al., 1998). At very low flood duration the species richness decreases again according to Pollock et al. (1998). They found species rich sites at intermediate flood frequencies and low to intermediate levels of productivity. Species poor sites had low or high flood frequencies and low productivity. Unproductive frequently flooded sites were dominated by r-selected data, like *Epilobium* an *Agrostis* spp, while productive undisturbed sites were dominated by comparatively slower growing shrubs and clonal herbaceous plants (k-selected species), like *Carex aquatilis*, *Calamagrostis canadensis*, *Vaccinium ovalifolium* and *Oploanax horridus*. Although flood duration is often mentioned as of great predictive value in wetlands, the duration of individual events plays a big role as well. Casanova & Brock (2000) found that flood events of two weeks at a time produce one plant community, flooding events of longer than two weeks produced a different plant community. Vervuren et al. (2003) found that a flood event during summer still influenced the vegetation composition years later. Long dry phases in between flood events give an opportunity for terrestrial species to establish.

Recovery of species can however occur very fast. Some effects are just temporary, like extension of flowering or growth, or the vegetation composition restores quickly through a quick restoration of the site factor conditions. Some species have a bad year, but come back the next year. For pioneer species along the river this is a normal situation which they thank their existence to. Flood events may even create additional habitat, like bare soil, which are exploited by species like willows.

Competition between species may play a role in the recovery. Good re-colonisers are in favour (Lensen et al., 2004). In the end the former species may not all be able to return again. This may change the species composition forever.

The recovery of the vegetation composition depends roughly on first of all the restoration of the site factors and secondly on the succession stage of the plant and the r- or k-strategy in other words the time needed to re-colonise and to re-growth. In case of a very frequent and long storage site factors have no ability to restore (like for winter and summer storage which both occur yearly). With a lower frequency (> year) site conditions can restore, but vegetation of last succession stages, like forest, do not always have the opportunity to re-growth again. As a rule of thumb we could say: if the frequency of a flood event is larger than the lifetime cycle of the ecosystem type than plants have time to recover from the event. Effects are temporary (year to decades, depending on vegetation type) and appear mainly in the fact the ecosystem is set back in succession. Consequently, excess storage can be less well combined with vegetation types with long life cycle, but for types with a shorter cycle, like pioneer and grassland vegetation chances are better (years to growth for pioneer 1 year, grassland years, shrub 5 years, forest 5 – 20 years).

Furthermore, we questioned ourselves, whether it would be possible if the changes could be more permanent or in other words if water storage could not only change the vegetation composition but also a shift to a different ecotope type with different site factors. This may occur as a result of winter and summer storage. Moisture content and acidity will change in case of summer storage, while in case of winter storage with externally supplied water the nutrient availability may change enough as well. This is not always an unwanted situation: it may contribute to rehabilitation of the nature conservation area. Excess and emergency storage will be executed in areas which are often already nutrient-rich and wet, so here the change will not be so extreme that it changes into another ecotope. Except for the river floodplain with moderate nutrient-rich sites like tranparent ponds with lots of macrophytes. Here the nutrient load may change into a species poor nutrient-rich turbid pond.

A permanent change could also appear through another mechanism: the presence or absence or ecological networks. Re-colonisation of species depends also on the availability of seeds, either from the soil, through connection through the water or from other nature areas. Absence of nearby nature areas which can contribute to seed supply decreases the ability to re-colonise the area. Furthermore, fauna species which use the water storage as their habitat should be able to move to other (not affected) areas. Sometimes this is needed for a long period: the time needed for the vegetation to recover. But is there enough space elsewhere and are these areas well connected? Furthermore, we should consider if we want to wait for vegetation to be regenerate.

The site factor model is useful, to describe effects systematically and get first quantification, but the quantification of dose-effect relations is difficult and it lacks factors characteristic for determining effects of water storage. We will elaborate this further.

Most dose-effect relations, especially the ones for brook valley wetlands, have been determined for the nowadays artificial maintained water levels. So, even the conditions for winter and summer storage are similar the old natural situation it is difficult to quantify them. Reference areas with still natural conditions may be helpful to overcome this lack of knowledge. In addition, nutrient availability is influenced in so many ways that difficult to assess the direction of the total effect, not to mention the quantitative relations.

Dynamics in terms of effects of an event in terms of transitional situations (succession) are not included in the concept model. It gives the new vegetation composition after a new equilibrium is settled. The model could be improved by adding flooding frequency and duration, maybe one for the growing season and the rest of the year. This makes it also more applicable for rivers as for these areas much more information of effects of flooding and flood events are available than in brook valley wetlands.

Other effects which can not be included (yet) with the site factor are inflow of heavy metals, mechanical stress of water by waves and wind exposure (Coops & van Velde, 1996), and increased connectivity. The latter stimulates the dispersal of seeds and propagules (Boedeltje, 2004, Brock et al., 1984).

Do we know enough about the impacts of water storage on nature, to combine water storage and nature conservation in the same area? And what knowledge is lacking and cannot be missed?

Knowledge lacking to make a good assessment whether water storage and nature conservation can be combined includes the sum of effects on nutrient availability. Does it increase in case of summer and winter storage? Is it temporary in case of excess or emergency storage? When will the original conditions return? If the plants are affected, what are the recover possibilities and periods? Most studies include effects on several species and not the whole species composition.

Information on natural or reference conditions is not numerous but could help to overcome the lack of knowledge to make good decision on the combination of water storage and nature conservation. We consider long-term experimental research essential for to quantify relations and determine the long-term effects of events.

If we do not know enough what decisions can we make with the current knowledge? Finally, we give suggestions about the combination of water storage with nature.

Recommendations for policy makers

Flood events in the last decades and the expected climate change make society believe we have to act now. Although not all the effects of water storage on ecosystems are clear, we can already make some recommendations to water managers.

Can flood and drought problems be solved with water storage in nature conservation areas? Solving both problems within the same area is not possible as the specific nature areas suffering from drought problems are too far away from the areas where water storage is needed for flood control. So we could say that water managers and nature conservationist are talking about two different things: about summer and winter storage on the hand and emergency and excess storage on the other hand.

Conservation of rainwater and preventing leakage of water from nature to surrounding areas, corresponding with summer storage, can be a good combination of water storage and nature conservation. In most brook valleys this water storage type can diminish desiccation problems and restore brook forests, acid fen meadows and hay meadows. For highly valuable alkaline mesotrophic vegetation more carefulness is needed as these plants are very sensitive to acidification. Rehabilitation of these nature reserves is only possible if the rainwater is mixed enough with bicarbonate-rich groundwater to neutralize the rainwater or with nutrient poor surface water.

Winter storage involves more water and flooding. These conditions suit mesothropic wetlands if there are anaerobic conditions during the growing season. This is not in favour of alkaline poor wetlands.

Excess and emergency storage results in flooding of large areas with a lot of water and introducing nutrients. Consequently, most species will be damaged. The extent of the damage depends on the timing, duration and water depth. In the worst case most species die. However, the next year you can see already recovery. In most cases the area is just set back into succession. More carefulness is needed for rare vegetation types and vegetation types which take years to develop like hardwood forest.

Current Dutch rivers have a tide jacket in the form of the embankments. There is not much room for a high spatial variation of flood frequencies. Creating more room for the river could enhance species richness and at the same improve safety. Furthermore, helophytes, brushwood and forest can remove nutrients from the water, creating chances for nutrient poor sites further away from the river and enhance biodiversity (Wassen et al., 2003). For this effect large floodplains are needed. In this way flood management and nature conservation can combine their objectives. Unfortunately, we have to admit that room was one of the reasons to question the combination of nature conservation and water storage.

Large 'bathtubs' with water should be prevented. Flooding is part of the natural conditions along rivers and brook valleys, but highly valuable nature conservation areas thank their value on the spatial differentiation in site factors and in particular flood depth and duration. Flooding determines for a large extent the other site factors and species richness. Only upward seepage can have an extra effect on the site factors.

Take care with water storage in old vegetation types, like hardwood forest. From the literature we know this type can cope with a little bit of flooding each year, but too long and to frequent results in dieback, which is too bad for this valuable vegetation.

Water storage in newly to develop nature areas is considered to be positive as it leads to larger surface areas and a more robust network of nature conservation areas. The development of the ecosystem in tune with the extraordinary circumstances ensures full adjustment to the new site conditions.

Water storage in wet nutrient-rich ecosystems will be acceptable in most cases, because the vegetation is relatively adapted to high levels of environmental dynamics, in contrast to acid or alkaline and/or nutrient-poor ecosystems. However, the acceptability of water storage cannot be adequately judged for all ecosystem types.

Take care of enough drainage capacity, as the longer the inundation the larger the effects of anoxicity to vegetation will be (Baptist et al., 2006)

To summarise, water storage and nature conservation can be combined in some cases especially in newly developed areas, but it is not the solution for all the problems. Rushed decisions on this matter should be avoided, each time effect should be considered carefully.

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