Status of ecotoxicological assessment of sediment and dredged material in Germany and The Netherlands

with a short description of the situation in Belgium, France, and Great Britain

DGE

Dutch-German Exchange on dredged material

Dutch-German Exchange (DGE) on Dredged Material

- Part 5 -

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1. Introduction

The Dutch German Exchange (DGE)

The Netherlands and Germany have large river systems such as Danube, Rhine, Meuse, Elbe, Weser and Ems, which have important hydrological and shipping functions and where dredging is essential. In case of the rivers Rhine, Meuse and Ems both countries have a common interest in the sound management of its sediment because their river basins lie in both countries. Finally both Germany and The Netherlands have large (sea) harbours such as Hamburg, Bremen/Bremerhaven, Rotterdam and Delfzijl, which receive large amounts of sediment both from the sea by tidal processes and from upstream areas by rivers. Therefore, both countries are equally subject to the cross-national (European) dimensions of dredging.

Against this background the competent governmental authorities in The Netherlands and Germany have started a Dutch-German Exchange on Dredged Material (DGE) in the year 1999. This DGE was started as an informal bilateral platform for exchanging knowledge, information and experiences in the field of sediment management. Since its start, several meetings were organized, in which subjects such as legislation, risk assessment, sediment treatment, hazardous substances, ecotoxicological and ecological assessment were discussed. The results of these discussions have been put down in thematic reports. The DGE has thus achieved an increased understanding of management of dredged material both on policy level (national) and practical (project) level.

The **Dutch-German Exchange** (**DGE**) was started as an informal bilateral platform for exchanging knowledge, information and experiences in the field of sediment management. Since its start in 1999 the discussion of subjects such as legislation, risk assessment and sediment treatment have been put down in thematic reports.

First having set the basis on the bilateral situation and second several developments in the international scene require a broader perspective for the exchange of information on sediment and dredged material. These developments are:

- Implementation of the WFD, leading to questions how to deal with relocating dredged material or in how far sediment remediation is an effective measure to improve water quality;
- Climate change, leading to increased risk for flooding, therefore increasing the need for dredging water ways or river realignment;
- Integration of river management in other policy making processes, such as Marine Strategies and Soil Protection Strategies.

For this reason, the Dutch and German participants in DGE have invited other countries to participate. The exchange is intended to be on the level of authorities that deal with the management of our rivers and ports. Most participants represent regulatory organizations (governments/port authorities). The exchange is informal, meaning that no official governmental statements will be prepared. However, the exchange will help to feed the contributing organizations with better knowledge and experience of sediment and dredged material management.

The aim of this report is to give a concise overview of ecotoxicological assessment procedures and hazard assessment methods for sediments and dredged material that are currently in use in Germany and The Netherlands. With the preparation of this report, representatives from Belgium, France and the United Kingdom joined the initial DGE expert group. Thus this description could be effectively supplemented by brief descriptions of the current situation in Belgium, France and the United Kingdom.

In general, two main goals for sediment and dredged material assessment procedures can be distinguished:

(i) assessment of the *in situ* quality of sediments and resulting risks at sites where sediment remediation is to be considered, and

(ii) assessment of the *ex situ* quality of dredged sediments in order to select dredged material relocation and sediment management options for maintenance and capital dredging on the one hand (e.g., free and confined disposal or treatment options) and on the other hand for the evaluation of environmental impacts of projected new-construction work.

These two objectives for conducting a risk assessment procedure are different in nature and therefore the risk assessment schemes are structured differently, as will be described in this report.

For the assessment of the risks of sediment pollution various methods can be used, ideally in an integrated form:

(i) by chemical analysis and comparison of contaminant levels with guidance values, or by modelling the “toxicological hazard”,

(ii) by an assessment of the condition of the benthic macroinvertebrate community using benthic ecology, which is reflecting reality, but is not predictive and difficult to discern subtle effects,

(iii) by an assessment of sediment/dredged material toxicity using bioassays and ecotoxicological methods which is less "real" (performed) in the laboratory, but can be predicative and can assess subtle effects,

(iv) by an assessment of the potential effect occurring through food chain poisoning.

To date it is common sense, that the most meaningful strategy for the assessment of potentially harmful effects in sediments and dredged material can be done by the integration of at least attempts (i)-(iii) comprising the analytical determination of chemical compounds as well as their effects on organisms, populations and ecosystems. Both in Germany and the Netherlands, biological assays that demonstrate toxic effects (the so called “bioassays“) are regarded as complementary to the physical and chemical measurements in sediment and dredged material. The estimation of the toxicological hazard can be regarded as an important element of a risk analysis procedure that can be used to evaluate the effectiveness of remedial action, or for the prevention of detrimental effects on the environment during and after the disposal of dredged material. Risk assessment and risk evaluation are the basis for management decisions on remediation or in the case of maintenance dredging, the disposal of dredged material (relocation, sub-aquatic, disposal at land) on a river basin scale.

Effect measurements can be performed on different levels of biological organization, often requiring different technical requirements:

(i) on the organismic level with the classical integrating toxicological endpoints behaviour (avoidance), mortality (e.g. microcrustacean lethal toxicity test with *Corophium*), immobility (e.g. microcrustacean acute immobilization test with *Daphnia*), growth and reproduction (e.g. algae growth inhibition test),

(ii) on the organismic level with physiological endpoints such as oxygen consumption (e.g. oxygen depletion test) or bioluminescence (e.g. luminescent bacteria test: Microtox test, Microtox solid phase test),

(iii) on the suborganismic level with endpoints such as cell toxicity, genotoxicity, immunotoxicity, and endocrine effects.

Bioassays can be used in order to detect toxic compounds, preferably in test batteries whose sensitivity cover the necessary groups of contaminants. Bioassays can also have a beneficial function in a screening phase, when further investigation has to be prioritised on the basis of measured toxicity. Another role of bioassays can be in a validation phase of the risk assessment, using these techniques to prove that sediment contamination leads to toxic effects. Often, bioassays are used in stepwise approaches, giving direction in a decision making process.

To ensure comparability of bioassays, standardisation after ISO and CEN of these techniques is an important step before implementation of bioassays into national, EU-wide or international directives or administrative regulations.

From the policy point of view, the need for the implementation of ecotoxicological bioassays comes from the need to translate international conventions and directives into national legislation. Examples of important international regulations are the OSPAR (1992) and HELCOM (1992) conventions on marine emission management, the EC water framework directive (2000/60/EC) and further EC directives (e.g. on the registration of chemicals).

The structure of the present report was chosen in order to describe ecotoxicological methods from different perspectives: firstly from a methodological point of view to elucidate the diversity of current techniques, and to show the state-of-the-art; secondly to describe the statutory order in the different countries. In this report the ecotoxicological assessment methods used in Germany and The Netherlands will also be compared to the current practice in other countries like Belgium, France and The United Kingdom of Great Britain and Northern Ireland that recently joined the DGE group (DGE, 2002, 2003, and 2005).

2. Ecotoxicological methods for the assessment of dredged material for relocation

2.1 Ecotoxicological investigation of dredged material in Germany

In Germany, handling and management of dredged material from federal waterways, that means from the inland waterways and the coastal waters within the jurisdiction of the Federal Waterways and Shipping Administration (WSV), are regulated by two directives introduced by the German Federal Ministry of Transport, Building and Urban Affairs (BMVBS), the Directive for the Management of Dredged Material in Coastal Waters (HABAK-WSV, 1999) and the Directive for the Management of Dredged Material in Inland Waters (HABAB-WSV, 2000) for the disposal of dredged material in inland waterways up to the freshwater limits to the North and Baltic Seas.

The environmentally sound quality of the dredged material is the prerequisite for approval to relocate that can be performed by disposal into the running stream, by hydrodynamic dredging (e.g. water injection procedure) or local deposition. The quality of the dredged material is assessed in a tiered approach on the basis of physical, chemical, biochemical and ecotoxicological criteria.

The inherent risk management tools in these directives encompass an approach, including physical, chemical, and ecotoxicological characterizations of sediments to be dredged. For assessing contaminant loads in dredged material, chemical guidance values are given for the freshwater sediments (relative guidance values) and the coastal sediments (fixed guidance values).

For the ecotoxicological-effect assessment pore water and elutriates obtained from these sediments are used. Additional sediment contact tests (whole-sediment tests) are part of the risk-assessment procedure. Ecotoxicological testing serves to determine lethal and sublethal, acute and chronic ecological harm. The ecotoxic effect is generally to be determined and assessed for the dredged material as well as for the sediments of the scheduled site of placement – the latter especially in the marine area.

The accumulation of contaminants in organisms (bioaccumulation) is only to be tested at the site of disposal within the framework of the monitoring programme, if substances with a high potential for bioaccumulation are present in the dredged material. The same samples are used both for chemical and ecotoxicological testing.

Contaminants that are not determined through chemical analysis, but are supposed to be bioavailable in effective concentrations in the dredged material, can be detected by means of appropriate toxicity testing.

Assessing freshwater sediments/dredged material, the biotest battery comprises bacteria, algae and daphnia as indicator organisms for toxicological effects. Ecotoxicological testing of marine and brackish sediments and dredged material includes acute and chronic tests with three different taxonomic groups (algae, bacteria, and microcrustaceans) in compliance with the directives for dredged material of the OSPAR Convention (OSPAR, 1992).

2.1.1 Quantification of the toxic potential of dredged material with the pT-method

The pT-method was developed as a management tool to incorporate bioassay data into the decision-making process for assessing and comparing the relative toxic hazards of sediments and dredged material. Once pT-values and pT-indices have been determined, the dredged material can be allocated to different management categories (Krebs 2005a, b).

The **pT-method** is used here to assess the quality of solid media such as sediments and dredged material. It appraises the toxicity of different compartments and phases of solids (e.g., whole sediment, porewater, elutriates, chemical extracts) with standardized bioassays, using a dilution series in geometric sequence with a dilution factor of two. The measured endpoint of toxicity corresponds to the first dilution stage at which the test material is no longer toxic to test organisms. Numerically, toxicity is reported as a **pT-value** related to the negative binary logarithm of the first non-toxic dilution factor identified. The pT-value indicates the number of times a sample must be diluted at a ratio of 1:2 with standardized dilution water or clean sediment, respectively, until adverse effects on the test organisms cannot be measured anymore.

While individual toxicity tests measure specific endpoints, a single test cannot be used to adequately reflect the general hazard potential of a sample. A hazard assessment can only be approximated using a multidisciplinary approach based on a large number of different toxicity tests within a test battery. An adequate strategy is the application of a multi-trophic testing scheme. The pT-value of the most sensitive organism within a test battery is known as the **pT_{max}-value**, and it determines the **toxicity class** of an investigated sample. All bioassays in a test battery are considered equal in rank, and Roman numerals are assigned to each toxicity class based on the magnitude of toxic effects observed in the most sensitive test organism. For instance, if the highest pT-value is 7, the tested material is then assigned to the Toxicity Class VII which corresponds to a **pT-index** of VII (Table 1). Hence, the pT-index derived from the most sensitive organism in a test battery constitutes a numerical classification based on ecotoxicological principles. With the aid of this simple index, the potential toxic hazard of any environmental sample can be quantified in an easily understandable way.

In the case of dredged-material classification, the generally open-ended **pT-ecotoxicity scale** is restricted to seven classes (Class 0 and Classes I to VI). All pT_{max} – values higher than 6 are included in Class VI (Krebs, 1999). In the context of dredged-material management, which may include its relocation within the water body, the seven toxicity classes determined by the pT-method are allocated to three **management categories** designated as "unproblematic", "problematic" and "hazardous" (Krebs, 2000; 2001), see Chapter 4.1.1. The **pT-ranking system** permits comparative studies with results of different test systems and sampling sites. It can also provide simple graphic representations of toxic sediment loading along the course of a river or in a whole river basin (see Chapter 4.1.1).

Table 1: Geometric dilution series, pT-values, and pT-indices for sediment investigations. The pT-values determined in real sediment samples are marked by the sign + (Krebs, 2005b).

Dilution factor as cardinal fraction	Dilution factor as decimal fraction	Dilution factor as exponential fraction	pT-value[a]	Toxicity class		Measured ecotoxicity in sediments	
				pT-index[b]	Designation	Pore-water	Elutriates
Original sample	1	2^{0}	0	0	toxicity not detected	+	+
1:1,25	0.8	$2^{-0,3}$	0	0	toxicity not detected	+	+
1:2	0.5	2^{-1}	1	I	very slightly toxic	+	+
1:4	0.25	2^{-2}	2	II	slightly toxic	+	+
1:8	0.125	2^{-3}	3	III	moderately toxic	+	+
1:16	0.0625	2^{-4}	4	IV	distinctly toxic	+	+
1:32	0.0313	2^{-5}	5	V	highly toxic	+	+
1:64	0.0156	2^{-6}	6	VI	extremely toxic "Mega toxic"	+	+
1:128	0.00781	2^{-7}	7	VII		+	+
1:256	0.00391	2^{-8}	8	VIII		+	-
1:512	0.00195	2^{-9}	9	IX	"Giga toxic"	+	-
1:1024	0.000977	2^{-10}	10	X		+	-
1:2048	0.000488	2^{-11}	11	XI		+	+
1:4096	0.000244	2^{-12}	12	XII	"Tera toxic"	-	-
1:8192	0.000122	2^{-13}	13	XIII		-	-
1:16384	0.0000610	2^{-14}	14	XIV		-	-

a) **pT-value:** The highest dilution level devoid of adverse effects is used for the numerical designation of toxicity with regard to a single test organism. The pT-value (*potentia Toxicologiae* = toxicological exponent) is the negative binary logarithm of the first non-toxic dilution factor in a dilution series in geometric sequence with a dilution factor of 2.

b) **pT-index:** The numerical toxicological classification of an environmental sample attained with a test battery. The pT-value of the most sensitive organism within a test battery determines the toxicity class of the tested material. Roman numerals are assigned to each toxicity class. If the highest pT-value is 9, for instance, the tested material is then designated as Toxicity Class IX (i.e., the pT-index is IX).

2.1.2 Ecotoxicological investigation of dredged material from inland waters

The pT-index allows the assessment and comparison of the toxic potential of sediments and dredged material. It is one example of an integrated bioassay-battery approach developed for the purpose of environmental management. This sediment assessment index relies on the use of an appropriate battery of bioassays at different trophic levels (primary producers, consumers, and decomposers) allowing the measurement of various types (acute, chronic) and levels (lethal, sublethal) of toxicity.
The Freshwater Test Battery used in inland waters according to the HABAB-WSV (2000) guideline is comprised of the following organisms:

- *Vibrio fischeri*, bacterial luminescence inhibition test
- *Desmodesmus subspicatus*, micro-algal growth inhibition assay
- *Daphnia magna*, cladoceran acute immobilisation test

General test description see Box 1.

Box 1: Freshwater test battery according to HABAB-WSV (2000). (Krebs, 2005b)

Algal test *Desmodesmus subspicatus* (R. CHODAT) E. HEGEWALD and A. SCHMIDT, 2000, formerly known as *Scenedesmus subspicatus* CHODAT, 1926 (Taxonomy: Chlorophyta, Chlorophyceae, Chlorococcales) Test performed according to DEV L 33 – DIN 38 412 Part 33 (1991) Toxicity endpoint: Cell growth inhibition; test duration: 72 h Number of test organisms per dilution step: 10^4 cells per ml Threshold value for the determination of the pT-value: IC<20% Threshold value for NH_4-toxicity 30 mg/l NH_4-N (Wahrendorf et al., 2005)
Luminescent bacteria test *Vibrio fischeri* BEIJERINCK, 1889; LEHMANN et NEUMANN, 1896, formerly known as *Photobacterium phosphoreum* (COHN, 1878) BEIJERINCK, 1889; (Taxonomy: Bacteria; Proteobacteria; gamma-Proteobacteria; Vibrionales; Vibrionaceae) Test performed according to DEV-L 34 – DIN EN ISO 11348-3 (1998), Freeze-dried Microtox® bacteria Toxicity endpoint: Luminescence inhibition; test duration: 30 min Number of test organisms per dilution step: About 10^6 cells per ml Threshold value for the determination of the pT-value: IC<20% Special test developments in Germany (Krebs, 1992a,b)
Daphnia test *Daphnia magna* STRAUS, 1820, water-flea (Taxonomy: Crustacea, Branchiopoda, Cladocera, Daphniidae) Test performed according to DEV L30 – DIN 38 412 Part 30 (1989) Toxicity endpoint: Microcrustacean acute immobilization; test duration: 24 h Number of test animals per dilution step: 10 Threshold value for the determination of the pT-value: 90% survival

This test battery is used for a 1st tier assessment. If the results indicate "Case 2" in the ecotoxicological categorization of contamination (Chapter 4.1.1) further tests have to be performed. For this purpose, additional test systems, including sediment contact tests, are currently under development and evaluation:

- *Lemna minor*, plant-growth inhibition assay (Feiler & Krebs, 1998; 1999; 2000; 2001)
- *Myriophyllum aquaticum*, plant-growth inhibition assay as sediment contact test (Feiler, 2004; Feiler et al., 2004)
- *Caenorhabditis elegans*, nematode reproduction test (Höss & Krebs, 2003; Höss et al., 1999; 2001; 2004; 2005)
- *Danio rerio*, zebra fish, fish-egg test (DEV T6; Hollert et al., 2003)
- *Salmonella typhimurium*, bacterial genotoxicity test or umu-test (DEV T3)
- *Salmonella typhimurium*, bacterial mutagenicity test or Ames-test (DEV T4)

2.2.1 Ecotoxicological investigation of dredged material from coastal waters

The marine bioassay test-battery for the assessment of brackish and marine sediment quality is currently developed and tested in several laboratories within Germany.
The respective project is conducted in close cooperation with the national standardization working group on marine bioassays (DIN AK 5.3) and the international standardization organisation (ISO/ TC 147 „Water quality"). At present, the proposed marine test battery consists of the algae test and the luminescent-bacteria test for the water phase of brackish and marine sediments and dredged material (tests with pore water and elutriates) and a whole-sediment bioassay with the estuarine and marine amphipod *Corophium volutator* (Box 2).

For the marine bioassays artificial sea water was successfully used both for culturing the test organism and for the test procedure. The tests were examined for their suitability in the assessment of marine- and brackish-sediment samples and were further refined.

The standardized marine-algae growth-inhibition test was initially meant for single-substance testing. Now, its use has been expanded to the testing of pore water and elutriates of brackish- and marine-sediment samples within this project.

The bioluminescence test was modified for brackish and marine elutriates. The modification was validated in a round-robin test with 24 laboratories within Germany. At present, the implementation of the modification in the standard protocol (DEV L34-ISO EN DIN 11348-1-3) as an informative annex is in preparation.

Box 2: Marine test battery according to HABAK-WSV (1999), Pfitzner & Krebs (2001). The algal and the bacterial test systems are used for aquatic phases like pore water and elutriates. The Corophium test is a sediment contact test conducted without dilution series. (Krebs, 2005b)

Algal test *Phaeodactylum tricornutum* BOHLIN, 1897 (Taxonomy: Bacillariophyta (Diatoms), Bacillariophyceae, Naviculales) Test performed according to DEV 45-DIN EN ISO 10253 (1998) Toxicity endpoint: Cell-growth inhibition, test duration: 72 h Number of test organisms per dilution step: 10^4 cells per ml Threshold value for the determination of the pT-value: IC<20%
Luminescent-bacteria test *Vibrio fischeri* (BEIJERINCK ,1889) LEHMANN et NEUMANN, 1896, formerly known as *Photobacterium phosphoreum* (COHN, 1878) BEIJERINCK, 1889; (Taxonomy: Bacteria; Proteobacteria; gamma-Proteobacteria; Vibrionales; Vibrionaceae) Test performed according to DEV L 34 – DIN EN ISO 11348-3 (1998); Freeze-dried Microtox® bacteria Toxicity endpoint: Luminescence inhibition; test duration: 30 min Number of test organisms per dilution step: About 10^6 cells per ml Threshold value for the determination of the pT-value: IC<20%
Corophium test *Corophium volutator* PALLAS 1766, mud shrimp (Taxonomy: Crustacea, Amphipoda, Corophiidae) Test performed according to ISO/DIS 16712 (2003) as a liquid-phase and sediment-contact test Toxicity endpoint: Microcrustacean acute toxicity; test duration: 10 d Number of test animals per dilution step: 10 Threshold value for the determination of the pT-value: 90% survival

2.2 Ecotoxicological investigation of dredged material in The Netherlands

2.2.1 Ecotoxicological investigation of dredged material from inland waters

Disposal of dredged sediments in Dutch freshwaters is restricted by law to clean or moderately contaminated sediments (up to 2007: class 0, 1 or 2), but in practice, this option is very rarely chosen. Normally, these sediments are re-used as construction material (after ripening) of put ashore. Freshwater sediments with a class 3 or 4 quality are put into confined disposal sites, in combination with sand separation or treatment. Effect-based approaches to evaluate sediment quality before or after treatment have been proposed (Brils & Den Besten, 1995) but have not been implemented within a legal framework yet.

2.2.2 Ecotoxicological investigation of dredged material from coastal waters

The Fourth National Policy Document on Water Management in the Netherlands (Ministerie V&W, 1998) specified that specific guidelines were to be drafted to supplement the existing system for the assessment of the presence of chemical substances in dredged material. That system was based on chemical measurements alone. The supplementary system should take greater account of the environmental and biological effects of dispersing saline dredged material. This resulted in a policy implementation to switch from the existing Uniform Concentration Test to the so called Chemistry-Toxicity Test (CTT). The CTT has been implemented in 2004 in the Dutch system of permit allocation (See Table 2).

The CTT contributes to improving the marine environment by raising the level of prevention for dispersing toxic dredged material in saline waters. The formal implementation of the CTT took place in 2004 by means of publication in the Government Gazette (Staatscourant, 2004), whereby the assessment context for the dispersability of dredged material in the Wvo (Surface Water Contamination Act) and Wvz (Maritime Water Contamination Act) is applicable.
In recent years about ten bioassays have been investigated in order to determine their suitability for the routine biological effect assessment of estuarine dredged material (Stronkhorst et al., 2001, Schipper et al., 2004). Three bioassays were selected and subjected to 4 years of trials (1999 to 2002) to determine whether they are sufficiently robust to be used in the routine application of the CTT standard. The bioassays have been selected for routine application known as the Chemical-Toxicity-Test, CTT approach, with an amphipod (the mud shrimp *Corophium volutator*) toxicity test, a bacterial test (Microtox Solid Phase) and a genetically modified cell line that reacts specifically to substances with a dioxin-like mode of action (DR-CALUX). These tests have the following characteristics (Box 3).

At present there is only a measurement obligation and the bioassays are being used only to signal the presence of unusually high levels of contaminants. In addition, further protocols and certification will be required before uniform and reliable analyses of bioassays can be developed and included in the CTT. English protocols are published on www.zeeslib.nl.
The CTT was evaluated in 2006 (Schipper & Klamer, 2006) (see Chapter 4.2.1).

Table 2: Chemical and ecotoxicological criteria for the Chemical-Toxicity Test (CTT) for evaluation of dredged marine sediment.

Chemical name	Group	Units	Test value[3]	Signal value[5]
Mud shrimp, amphipode *C. volutator*	Combination toxicity	Mortality (%)		50
Microtox SP, bacteria *V. fischeri*	Combination toxicity	Bioluminescence $(1/EC_{50})$[1]		100
DR-CALUX cell line rat hepatoma cells	Dioxin-like	ng TEQ/kg dw		50
Tributyltin	Organometal	µg Sn/kg dw	100-250[4]	
Copper[2]	Metal	mg/kg dw	60	
Arsenic[2]	Metal	mg/kg dw	29	
Cadmium	Metal	mg/kg dw	4	
Mercury	Metal	mg/kg dw	1.2	
Chromium	Metal	mg/kg dw	120	
Zinc[2]	Metal	mg/kg dw	365	
Nickel	Metal	mg/kg dw	45	
Lead	Metal	mg/kg dw	110	
Sum 10 PAH	PAH	mg/kg dw	8	
Hexachlorobenzene	OCP	µg/kg dw	20	
Sum DDT/DDD/DDE	OCP	µg/kg dw	20	
Mineral oil C10-40[2]	Oil	mg/kg dw	1250	
Sum 7 PCB	PCB	µg/kg dw	100	

1) EC50MSPt expressed as a reciprocal of sediment concentration (dry weight basis) and corrected for fine silt particles.

2) These parameters are subject to a 50% test rule.

3) The test value is a firm upper limit, subject to the provision that no more than two of the parameters to which the 50% test rule applies may exceed the test value, and that they may do so by no more than 50%.

4) The test value for tributyltin is expressed as a range, within which the actual test value is to be determined in relation to each individual application for the dispersal of dredged material in saline waters. In deciding such applications, account will be taken of the fact that there may be no significant variation from the trend in the volumes of dredged material being dispersed in saline waters.

5) If the signal value is exceeded, the permit or exemption-holder must investigate the cause of the exceedance but the batch of dredged material will not be automatically disqualified. For the record, it should be pointed out that the signal values in question are not related to those mentioned in the Fourth National Policy Document on Water Management (*Vierde Nota Waterhuishouding*) as part of the system for assessing the need to remove contaminated or seriously contaminated sediments.

Box 3: Marine test battery according to Chemical-Toxicity Test (Schipper & Schout, 2004).

DR-CALUX test

Rat hepatoma cells (H4IIE) transfected with the plasmid pGudLuc 1.1.
This plasmid contains three DREs (Dioxin Responsive Elements) and the luciferase gene from the firefly *Photinus pyralis* as the reporter gene .

The DR-CALUX test reacts specifically to compounds with a dioxin-like action (dioxins are highly toxic, complex organic compounds). The number of dioxins in the dredged material extracts can be determined by measuring the amount of light emitted by the cells.

Test performed according to RIKZ/SPECIE -07 (Schipper and Stronkhorst, 1999) describes the determination of 2,3,7,8-TCDD toxic equivalents (TEQ) in prepared sediment extracts.

Toxicity endpoint: bioluminescence; 24 hours
Exposure of transfected H4IIE cells: from sediment samples after homogenisation extracted with a mixture of hexane and acetone.

Luminescent bacteria test

Vibrio fischeri (BEIJERINCK ,1889) LEHMANN & NEUMANN, 1896, formerly known as *Photobacterium phosphoreum* (COHN, 1878) BEIJERINCK, 1889;
(Taxonomy: *Bacteria*; Proteobacteria; gamma-Proteobacteria; Vibrionales; *Vibrionaceae*)
The **Microtox Solid Phase test** is followed by an assessment of the impact on the bacteria's metabolic processes, which is then compared to the CTT standard.

Test performed according to RIKZ/SPECIE-02 as suspension test sediment (Schipper and Stronkhorst, 1999).
Freeze-dried Microtox® bacteria
Toxicity endpoint: luminescence inhibition; test duration: 20 min
Number of test organisms per dilution step: about 10^6 cells per ml

Corophium test

Corophium volutator PALLAS 1766, Mud shrimp
(Taxonomy: Crustacea, Amphipoda, Corophiidae)
The amphipod test with *Corophium volutator* is a bioassay in which these mud shrimps are exposed to dredged material. After ten days the degree of survival is then compared to the CTT standard.
Test performed according to RIKZ/SPECIE01 as **sediment-contact test** (Schipper and Stronkhorst, 1999).
Toxicity endpoint: microcrustacean acute toxicity; test duration: 10 days
Number of test animals: 10

3. Ecotoxicological methods for the assessment of the in situ quality of sediments for remediation

An important factor in the risks caused by sediment-bound chemicals is the degree of exposure encountered by sediment-dwelling organisms. As a result of sorption of contaminants to sediment particles, actual exposure levels are lower than would be expected on the basis of the total concentrations of compounds in sediment (Hamelink et al., 1994; Kraaij, 2001). However, it is also known that mixtures of contaminants can have additive or synergistic effects (Hermens et al., 1984; von Danwitz, 1992), which may not be well addressed by single sediment quality guidelines (SQGs). Even the identity of usually a significant part of the contaminants is often not known. For these reasons, ecological risk assessment and sediment quality assessment have been based not only on chemical measurements, but also on biological endpoints. Biological endpoints integrate the effects of all contaminants present at their actual bioavailability (and detect possible combination or synergistic effects). Finally, ecological risk assessment of contaminated sediments should also account for possible exposure through diet (Lee et al., 2000a; Lee et al., 2000b).

3.1 Fresh water sediments in Germany

In Germany, the assessment of the in situ quality of sediments for remediation has not been implemented in policy documents as yet.

A first tier sediment classification using chemical data is done in Germany according to the ARGE Elbe classification. This classification is based on chemical analysis data that has been proposed by the Working Group for the Advancement of Water Quality of the River Elbe (*Wassergütestelle der Arbeitsgemeinschaft für die Reinhaltung der Elbe*, ARGE Elbe). The classification approach takes into account the loads of metal and organic compounds in sediments. For each compound, a classification into one of 7 classes is done. A certain number of points are given for each class. These points are added and the results can be classified (ARGE Elbe, 1996; Calmano, 2001; Maaß, 2001; Heise et al., 2005).

A concept for deriving quality objectives from ecotoxicological test results for the protection of inland waters against hazardous substances was also developed by the Working Group of the German Federal States on Water Problems (LAWA), (LAWA 1997a, b, 1998a-d, Krebs et al., 1985; Gottschalk et al., 1986; Gottschalk, 1994; Schudoma, 1994)

An overview of sediment studies conducted in German rivers and coastal waters is given in Table 3.

3.2 Marine sediments in Germany

The assessment of the in situ quality of marine sediments for remediation has not been implemented in German policy documents so far.

For the ecotoxicological assessment of estuarine and marine sediments, and the monitoring of sediment quality in marine waters, both bioassays and surveys of the macrozoobenthic community are used in monitoring studies. The ecotoxicological tests used are given in Box 2.

Tests covering bioaccumulative effects in organisms (e.g. nematodes, mussels) and the use of highly specific bioassays, e.g. for the assessment of dioxin-like effects are currently being evaluated. An prominent example is the EROD assay that is currently under ISO standardization for the assessment of adverse dioxin-like effects towards the fish fauna closely related to contaminated sediments (examples are given in Table 3).

Table 3: Selected literature on ecotoxicological and chemical sediment studies in German rivers and coastal waters

Study area	Publication
Oder and its tributaries: Neiße	Claus & Feiler (2001), Duft et al. (2003), Kase (2004)
Berlin water ways: Rivers Havel, Spree, Dahme, Teltow channel	Claus et al. (2000), Wittekindt et al. (2001), Dizer et al. (2002), Alcock et al. (2003), Hansen & Huschek (2005), Hansen et al. 2006, Huschek & Hansen (2006)
Elbe and its tributaries: Saale	ARGE Elbe (1996), Heininger et al. (1998, 2003) Wittekindt et al. (2000), Claus et al. (2000, 2002), Claus & Feiler (2001), Gratzer & Ahlf (2001), Neumann & Francke (2001), Feiler et al. (2002), Duft et al. (2003a), Förstner et al. (2004), Kase (2004), Heise et al. (2005), Reifferscheid et al. (2005a)
Hamburg Harbour	Schmidt (1994), Maaß et al. (1997), Maaß (1999, 2001), Heise et al. (2005)
Weser	Duft et al. (2003a), Kase (2004)
Ems	Duft et al. (2003a)
Rhine and its tributaries: Neckar, Main, Mosel, Saar, Ruhr	Feiler & Krebs (1998, 1999, 2000, 2001) Feiler et al. (2002, 2006), Ulrich et al (2002), Duft et al. (2003a), Hollert et al. (2003), Heise et al. (2004), Kase (2004), Köthe et al. (2004), Kosmehl et al. (2004, 2006b), Krebs (2005b), Reifferscheid et al. (2005a), Seiler et al. (2005)
Neckar	Hollert et al. (2000a, 2000b, 2002a, 2002b, 2003a, 2005), Brack et al. (2005)
Danube	Duft et al. (2003a)
Storage lakes, impounding reservoirs, impoundments	Henschel et al. (2001), Henschel et al. (2001b,c,d; 2003a,b)
North Sea	Bresler et al. (1999), Lozán et al. (1990, 1994), Hansen & Pluta (1994), Lozán & Kausch (1996), Herbst & Nendza (2000), Netzband (2001), Hansen (2003), Schmidt et al. (2003), Skouras et al. (2003), **Ems:** BfG (2001, 2002), **Jade:** BfG (2003a), **Weser:** BfG (1999, 2003b), **Elbe:** BfG (1995, 2005), **Wadden Sea:** BfG (1997), NP-SH (1998)
Baltic Sea	Baumgard et al. (1999), Lozán et al. (1996), Hansen et al. (1999), Hansen (2001), Kase (2004)

3.3 Fresh water sediments in The Netherlands

For freshwater systems in The Netherlands, assessment of in situ sediment quality is required within the legal framework of the Soil Protection Act (ref). This act requires risk assessment as a second tier for sediments with contaminant concentrations above the intervention value and for which in a first tier assessment toxic pressure has been concluded above a certain level[1]. After this first step, the Triad approach (basic principle as described by Den Besten, 1995) is used in order to determine the urgency of remedial action. The two-tiered risk assessment is believed to supply the necessary information to evaluated whether remedial action would support achievement of WFD goals (Guideline for sediment remediation, 2006). Ecotoxicological information is to be combined with information about the interaction between sediment and water quality. Ecological observations play a dominant role in the Triad approach, but they are described in a separate document (DGE, 2007). Another important element in the Triad is the measurement of toxic effects in (sediment) bioassays. In the Netherlands, much experience exists with the following bioassays:

- *Chironomus riparius* (test with midge larvae in whole sediment samples)
- *Daphnia magna* (water flea; test with sediment pore water)
- *Photobacterium phosphoreum* (also known as Microtox assay; sediment pore water)

In addition, three other bioassays have been applied in a small number of studies:

- *Thamnocephalus platyurus* (sediment pore water bioassay with a freshwater crustacean)
- *Brachionus calyciflorus* (sediment pore water bioassay with rotifers)
- *Ephoron virgo* (test with mayfly larvae)

For the first three bioassays criteria have been set in order to classify effects as non-toxic, moderately (Criterion 1) or highly toxic (Criterion 2) as depicted in Table 4 (see Van Elswijk et al., 2001).

The risks of food-chain poisoning as a result of bioaccumulation can be assessed in two ways (Van Elswijk et al., 2001):

- By collecting organisms from the field. In The Netherlands bioaccumulation has been measured as part of studies on the risks caused by sediment pollution. Contaminant levels have been measured in chironomids, oligochaetes, fish and Cormorant eggs (Den Besten et al., 1995). The contaminant levels were evaluated by comparison with maximum tolerable risk (MTR) levels that have derived specifically for concentrations in different types of food (see Den Besten et al., 1995; MTRs are lower when the energy content of the food is high, e.g. in the case of birds that catch eel).

- By performing the accumulation bioassays with aquatic worms according to the method described by Maas et al., (1993) and Den Besten, (2003). Aquatic worms are exposed for 4 weeks to samples of sediment in the laboratory after which the organisms are collected and processed for analyses of the contaminant levels. Accumulation is evaluated by comparing the accumulation levels with reference values and with MTRs as described above.

[1] The toxic pressure on aquatic organisms is calculated using the model OMEGA (ref), and with bioavailable concentrations of contaminants in the sediment as the model input. The model will calculate the potentially affected fraction of species (PAF; Posthuma et al., 2002). The same methodology has been developed for the assessment of risks of soil contamination (Mesman et al., 2003). With this model, direct effects and effects as a result of foodchain poisoning can be distinguished. In the Netherlands, mild extraction techniques with $CaCl_2$ or Tenax are used for measurement of the contaminant concentrations considered to be bioavailable (Cornelissen et al, 2001; see also Van Elswijk et al., 2001).

Table 4: Criteria for the classification of sediment toxicity on the basis of effects measured in bioassays (Van Elswijk et al., 2001).

Bioassay (type)	**Parameters and criteria**			
Chironomus riparius **(sediment)**	**Parameters**			
	Mortality eggs, prior to start sediment bioassay (incubation of eggs in elutriate)	Mortality larvae	Inhibition of development	Weight reduction (relative to control)
Criterion 1	mortality > 25%	mortality > 10% mortality < 50%	inhibition > 10% inhibition < 50%	effect > 10% effect < 25%
Criterion 2	mortality ≥ 50%	mortality ≥ 50%	inhibition ≥ 50%	effect ≥ 25%
Daphnia magna **(sediment pore water)**	**Parameters**			
	NOEC-mortality (in % dilution of pore water)	Mortality in undiluted pore water	NOEC-reproduction	Inhibition of reproduction in undiluted pore water
Criterion 1	NOEC < 100% NOEC > 10%	-----	NOEC < 100% NOEC > 10%	Inhibition > 10% Inhibition < 50%
Criterion 2	NOEC ≤ 10%	mortality ≥ 50% within 48h	NOEC ≤ 10%	Inhibition ≥ 50%
Vibrio fischeri **(sediment pore water)**	**Parameters: TU= 1/EC20 (determined after 5, 15 and 30 min)**			
Criterion 1	1/EC20 > 2			
Criterion 2	1/EC20 ≥ 10			

1) Exceedance of criterion 1 results in class 2 toxicity (moderate effects); exceedance of criterion 2 results in class 3 toxicity (strong effects). Otherwise, class 1 (no toxicity).

2) For each test, the most sensitive parameter is used for classification; effects on which score depends must be significant at $p < 0.05$ using an appropriate statistical test.

3.4 Marine sediments in The Netherlands

Sediment quality in brackish and coastal areas is assessed by using the following bioassays:

- The Microtox test (*Vibrio fischeri*). In this test the bioluminescence is measured during exposure to sediment pore water or elutriates.

- The bioassay with the mud shrimp *Corophium volutator* (Schipper & Stronkhorst, 1999). Survival is the toxicity parameter in this test.

- The bioassay with the polygochaete *Arenicola marina* (Thain & Bifield, 1999). In this test the activity of the test organisms is used as a sensitive indicator of toxicity.

- The bioassay with the Japanese oyster (*Crassostrea gigas*). Oyster larvae are exposed to sediment pore water or elutriates. Effects on the embryonic development are used in this test as indicators of toxicity.

Criteria for the classification of effects as non-toxic, moderately or highly toxic are described by Van Elswijk et al., (2001).

Within the framework of the MWTL programme and JAMP/OSPAR, only a limited number of sediment, sediment pore water and water bioassays have so far been employed in estuarine and coastal waters. However, a recent Dutch advisory report provides preliminary criteria for water and sediment bioassays in this context (Maas et al., 2003). In order to effectively describe the condition of water systems, and to take the necessary measures, it is necessary to have a picture of the concentrations of the priority substances. However, both the OSPAR List of Chemicals for Priority Action, and the priority substance list of the EU Water Framework Directive include substances of which it is not or only partly known whether they represent a possible issue for Dutch marine and estuarine waters. As such, attention was first devoted to phthalates, and brominated flame retardants, priority substances of which the presence is not well documented. Research was carried out in 2003 when all analysis and bioassays were performed on sediment and suspended matter samples at various locations in the coastal waters and at open sea (Åkerman et al., 2004).

One of the main benefits of the WFD and its monitoring programme is the use of both chemical and ecological parameters. Although bioassays are not prescribed in the WFD guidelines, opportunities for bioassays can be seen in both challenges. Toxicity results from bioassays on environmental samples show that most of the toxicity is due to unknown and un-measured compounds. Two possible applications of bioassays have been proposed (Maas et al. 2004):

1. Eco-assays: the use of tests as a tool to determine the causes of below-standard ecological status of water bodies. Eco-assays can be used as part of a diagnostic system to identify or confirm chemical, ecological or hydro-morphological pressures.

2. Bio-analysis: the use of bioassays to partially replace chemical analyses of priority pollutants or other relevant compounds in chemical monitoring. The goal is not an extended analysis of water quality, but a better indication of hazard.

4. Risk assessment of dredged material for relocation

4.1 Statutory order in Germany

4.1.1 Statutory order for relocation of dredged material in inland (HABAB-WSV) and coastal waters (HABAK-WSV): Toxicity classes and ecotoxicological management categories

Management of dredged material

The solid material removed from the bed of a river, canal or harbor in dredging projects should be relocated. In the Federal Republic of Germany, the larger waterways are the property of the Federal Government, whereas a number of smaller waterways are the property of Federal States. In the practice of the Federal Waterways and Shipping Administration (WSV), dredged material is usually relocated within the water body from which it was removed. This relocation may be performed either by dumping the material directly into flowing water, by hydrodynamic dredging (*e.g.* suction dredge, water jet), or by confined disposal. Sediment removal and its relocation are considered as one continuous process performed under the sovereign administrative activity of the Federal government (Köthe and Bertsch, 1999; Köthe, 2003).

Management of dredged material is regulated in Germany in two Federal guidelines: the Guideline for Management of Dredged Material in Inland Waters (HABAB-WSV, 2000) and the Guideline for Management of Dredged Material in Coastal Waters (HABAK-WSV, 1999). For disposal of dredged material on land, only the directive for inland waters (HABAB-WSV, 2000) is applicable.

The quality of dredged material must satisfy environmental protection standards for material relocation. Its quality is examined by physical, sedimentological, chemical, biochemical (including oxygen and nutrient balances), and ecotoxicological criteria and is assessed according to definitions of the guidelines. If relocation within a Federal waterway is not possible, the material can be used for direct or indirect beneficial uses, upland disposal or disposal in waters other than Federal waterways. For these options, the guidelines provide only a general orientation, because the approval procedures fall under the jurisdiction of the Federal States.

The general procedural steps for dredging-material handling are described for inland waters in Figure 1 and for coastal waters in Figure 2.

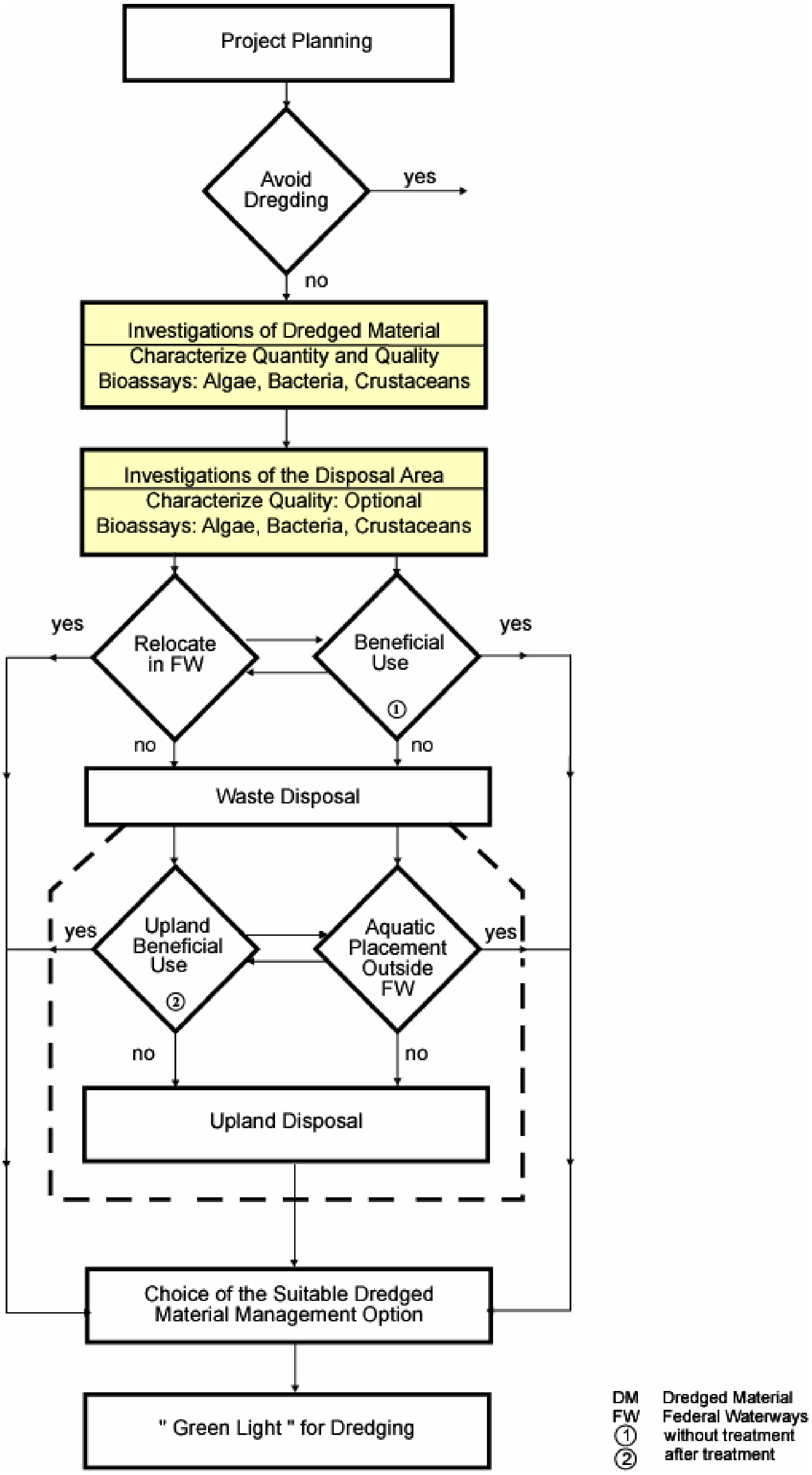


Figure 1: Dredged-material handling in inland waters. Procedural steps for the decision-making on dredged-material relocation in federal waterways according to HABAB-WSV (2000).

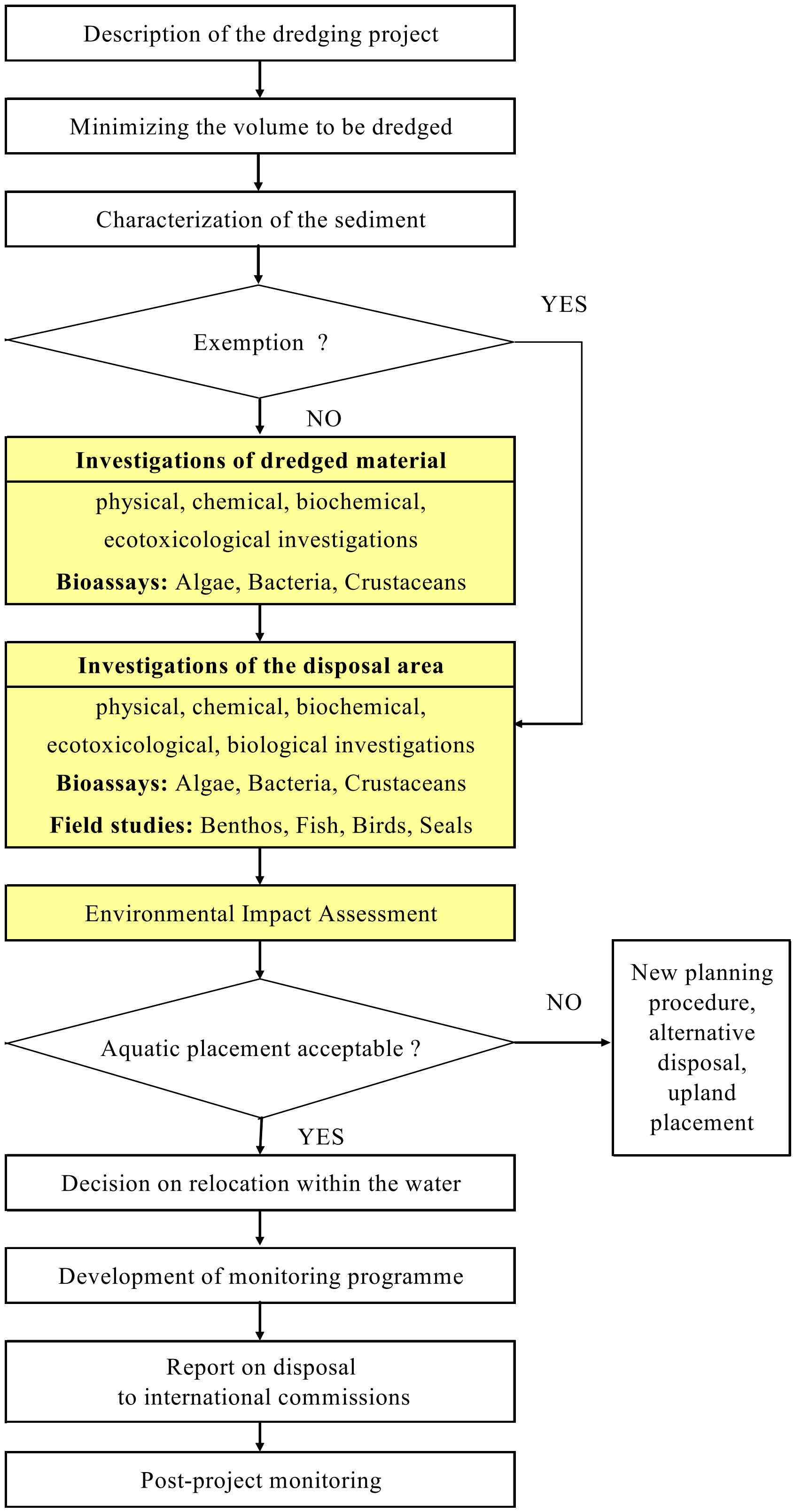


Figure 2: Dredged-material handling in coastal waters. Procedural steps for the decision-making on dredged-material relocation in federal waterways according to HABAK-WSV (1999). (Krebs, 2005b)

Management categories for the relocation of dredged material

In the context of dredged-material management, the seven toxicity classes established by pT-values are assigned to management categories labeled as “unpolluted”, "unproblematic", "problematic" and "hazardous" (Table 3). These categories then define “cases” by which dredged material can (or cannot) be relocated, as recalled below:

Case 1: Pursuant to the guideline for handling of dredged material from inland waterways (HABAB-WSV, 2000) and its counterpart for coastal waterways (HABAK-WSV, 1999), dredged material up to toxicity class II can be relocated without restriction.

Case 2: The relocation of dredged material of toxicity classes III and IV must be decided on a case-by-case basis. An impact hypothesis (prediction of potential impacts) is mandatory.

Case 3: Dredged material of the two highest classification levels (toxicity classes V and VI) must not be relocated in inland waterways according to HABAB-WSV (2000) and should not be relocated in coastal areas according to HABAK-WSV (1999).

Table 5: Toxicity classes defined by the German Federal Institute of Hydrology (BfG) for sediment assessment and ecotoxicological management categories for dredged material relocation. Ecotoxicological characterization is based on porewater and elutriate bioassay responses. Colour coding, refer to Table 6. (Krebs, 2000)

Highest dilution level without effect	**Dilution factor**	**pT-value**	**pT-index Toxicity class**		**Management category**	
			7-level system	**Designation**	**4-level assessment**	**Colour coding**
Original sample	2^0	0	0	toxicity not detected	**unpolluted**	**0**
1:2	2^{-1}	1	I	very slightly toxic	**unproblematic**	**I**
1:4	2^{-2}	2	II	slightly toxic		**II**
1:8	2^{-3}	3	III	moderately toxic	**problematic**	**III**
1:16	2^{-4}	4	IV	distinctly toxic		**IV**
1:32	2^{-5}	5	V	highly toxic	**hazardous**	**V**
≤ (1:64)	$\leq 2^{-6}$	≥ 6	VI	extremely toxic		**VI**

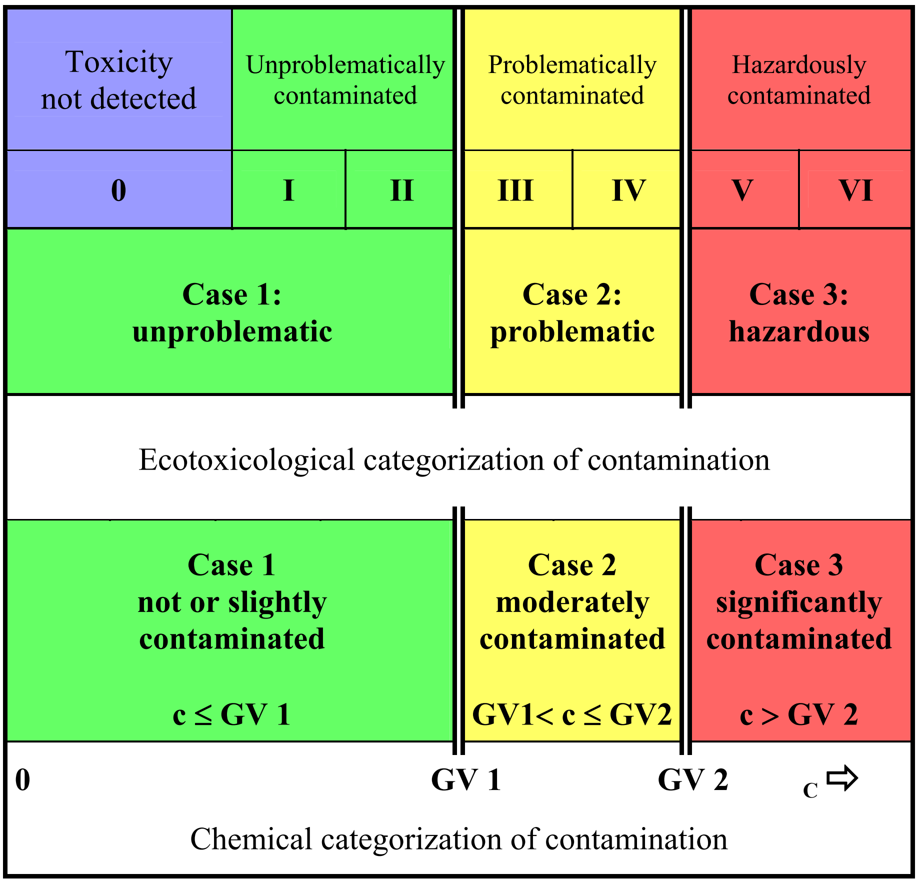
For verbal description of toxicity classes, the designations listed in Table 5 may be used, in conjunction with the number of the toxicity class and in the case of Toxicity Class VI additionally the maximum pT-value found. The combined characterization is indispensable, because the information is expressed by the pT-index and not by the verbal descriptions which may, in principle, be chosen freely and are thus subjective judgments.

As seen in Table 6, the degree of contamination of dredged material follows matching colour codes for ecotoxicological (bioassay-based) and chemical assessments, respectively, set with pT-indices and measured concentrations of specific contaminants. Hence, management categories relating to the degree of hazard of sediment material intended for dredging correspond to Cases 1 to 3, and in parallel to the colour codes (green, yellow, red) that signal decisions to be taken with respect to the relocation of dredged material.

Table 6: Management categories for dredged-material relocation used by the Federal Institute of Hydrology (BfG). Chemical and ecotoxicological criteria are those of HABAB-WSV (2000). The marine HABAK-WSV (1999) uses the same categorization (Case 1, 2, 3) but without the designations “problematically” and “hazardously contaminated”. (Krebs, 2000)

Legend:
c = contaminant concentration; GV 1 and GV 2 = Guidance Values 1 and 2 *(Richtwerte RW 1 und RW 2)*
pT-indices in Roman numerals
Colour coding for management categories for dredged material relocation:

Toxicity not detected	Case 1	blue	Toxicity Class 0
Unproblematically contaminated	Case 1	green	Toxicity Classes I and II
Problematically contaminated	Case 2	yellow	Toxicity Classes III and IV
Hazardously contaminated	Case 3	red	Toxicity Classes V and VI



In the Federal Institute of Hydrology (BfG) the pT-method as a hazard-assessment scheme (Chapter 2.1.1) is used as an ecotoxicological discriminator to map out sediment quality in polluted zones, along the course of a river or in a whole river basin. During the last years, in numerous investigations pT-values and pT-indices were obtained from sediments of the River Rhine and its tributaries Moselle, Saar and Neckar (Krebs, 2000; Krebs, 2005b).

Table 7: Sediment quality in waters associated with the rivers Saar and Moselle (old arms, harbours, and marinas). The ecotoxicological characterization is based on porewater and elutriate bioassay responses generated with algae, bacteria, and daphnids. Bioassays conducted according to HABAB-WSV (2000), refer to Box 2. For colour coding information of toxicity classes, refer to Table 6. (Krebs, 2005b)

No.	Location	Sample	Algal Test				Luminescent-Bacteria Test				Daphnia Test				Toxicity class[e]
			Porewater		Elutriate		Porewater		Elutriate		Porewater		Elutriate		
			%[a]	pT[d]	%[a]	pT[d]	%[b]	pT[d]	%[b]	pT[d]	%[c]	pT[d]	%[c]	pT[d]	
HS 1	Saarbrücken	Bed-Surface Sample	-3	0	-19	0	22	1	30	2	0	0	0	0	II
AW 1	Wadgassen		11	0	24	1	39	2	32	1	0	0	0	0	II
HD 1	Dillingen		-39	0	-25	0	3	0	-6	0	10	0	0	0	0
BM 1	Merzig		28	2	-2	0	23	1	36	2	0	0	0	0	II
BT 1	Trier-Monaise		11	0	-23	0	9	0	9	0	0	0	10	0	0
HS 2	Saarbrücken	Sediment-Core Sample	-29	0	-20	0	9	0	26	1	0	0	0	0	I
AW 2	Wadgassen		100	6	100	3	100	7	100	7	100	3	100	5	VI
BM 2	Merzig		-16	0	-10	0	-3	0	-1	0	0	0	0	0	0
AS 2	Schwemlingen		43	2	46	2	30	1	18	0	100	2	70	1	II
AS 3	Schwemlingen		63	4	49	4	33	1	39	2	100	2	100	1	IV
BT 2	Trier-Monaise		-22	0	-17	0	7	0	19	0	0	0	0	0	0

a) percent growth inhibition for the test alga, *Desmodesmus subspicatus*, in the undiluted test material (negative % values indicate stimulation) (DEV L33).

b) percent light inhibition for the test bacterium, *Vibrio fischeri*, in the undiluted test material (negative % values indicate stimulation) (DEV L34, Microtox® bacteria).

c) percent of immobilized test animals, *Daphnia magna*, in the undiluted test material (DEV L30).

d) pT-value characterizing the potential toxicity of each sediment sample compartment for a specific test organism (test-specific value).

e) the pT-value of the most sensitive organism in the test battery, the pT_{max}-value, determines the toxicity class (pT-index) of the dredged material.

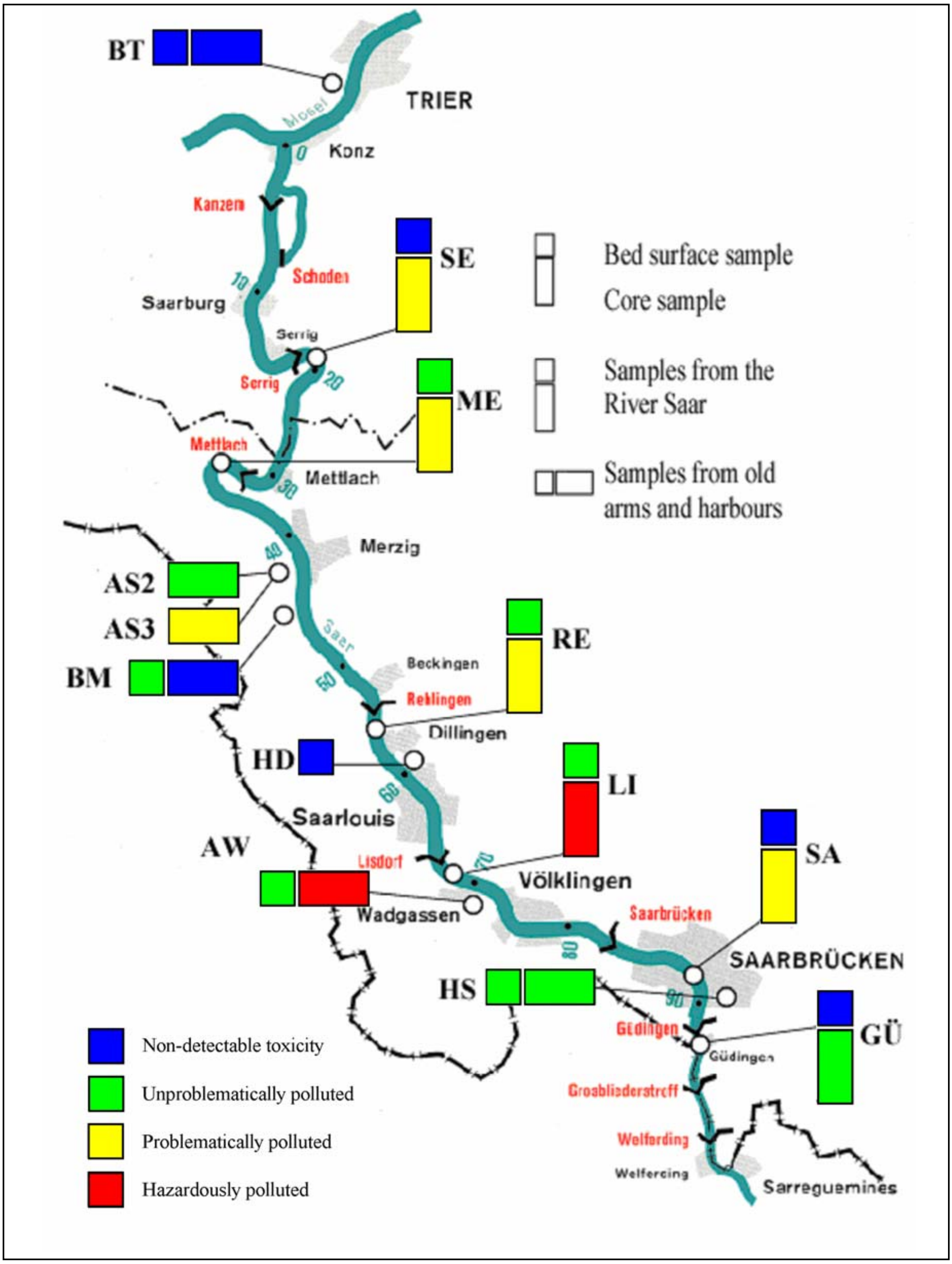


Figure 3a: Sediment mapping of the River Saar based on pT-scale ecotoxicological investigations conducted with algal, bacterial, and microcrustacean bioassays. This classification is derived from results reported in Table 7. For colour coding information of toxicity classes see Table 6. (Krebs, 2005b)

A map of the federal waterways in Germany is given on the next page (Figure 3b).

BUNDESWASSERSTRASSEN

- Wasser- und Schifffahrtsverwaltung des Bundes -

SÜDWEST	Bezeichnung einer Wasser-und Schifffahrtsdirektion		Grenze zwischen Wasser- und Schifffahrtsdirektionen
MAINZ □	Sitz einer Wasser-und Schifffahrtsdirektion		Grenze zwischen Wasser- und Schifffahrtsämtern
Mannheim ◇	Sitz eines Wasser-und Schifffahrtsamtes u. dgl.		WS-Klasse 0 - III
- - - - -	Staatsgrenze		WS-Klasse IV - VI

Bundeswasserstraßen, die eine Länge von unter 5 km aufweisen, sind maßstabsbedingt teilweise nicht dargestellt.

Kartographie: Sonderstelle für Vermessungswesen beim Wasser- und Schifffahrtsamt Regensburg
Vertrieb: Drucksachenstelle der WSV bei der Wasser- und Schifffahrtsdirektion Mitte, Postfach 6307, 30063 Hannover

4.1.2 Statutory order for the Environmental Risk Assessment (ERA) associated with the relocation of dredged material in the planning of waterway new construction and development projects according to the ERA-matrix

(1) Evaluation of the project impacts

Environmental risk assessment (ERA)

In Germany the methodology for an Environmental Risk Assessment (ERA, Umweltrisikoeinschätzung, URE) is regulated in the **Federal Transport Infrastructure Plan** (BMVBW 2002a, b and 2003a, b). The impacts of projected new-construction work or work to upgrade on nature and the landscape, water and soil, human health and well-being have to be identified and evaluated. It has to be pointed out that ERA is assigned exclusively to the decision-making level in the federal transport-infrastructure planning process and is thus clearly delimited in its terms of reference and accuracy of detail from the studies in the subsequent **plan-approval procedure** (*Planfeststellungsverfahren*), the **Environmental Impact Study for project planning (EIS)** (*Umweltverträglichkeitsuntersuchung, UVU*) and the **Environmental Impact Assessment** (**EIA**) (*Umweltverträglichkeitsprüfung, UVP*), see Boxes 4 and 5.

Box 4: Stages of the examination of probable environmental impacts of projected new-construction work of transport infrastructure in Germany. Planning procedures for the development and new construction of federal waterways by the German Federal Waterways and Shipping Administration (WSV).

Planning level: <u>Federal Transport Infrastructure Planning</u>
(waterways, roads, railways) (BMVBW, 2002a, 2003a)
Planungsebene: Bundesverkehrswegeplan (BMVBW, 2002b, 2003b)

Strategic Environmental Review of Plans and Programmes (SUPG, 2005)
Strategische Umweltprüfung (SUP) der Bundesverkehrswegeplanung

Environmental Risk Assessment (ERA) (BMVBW, 2002a, 2003a)
Umweltrisikoeinschätzung (URE) (BMVBW, 2002b, 2003b)

Planning level: <u>Plan-approval procedure</u>
Planungsebene: Planfeststellungsverfahren für ein Vorhaben

Environmental Impact Study for project planning (EIS) by the developer, i.e. the Waterways and Shipping Offices (WSÄ) or
the Offices for Waterway New Construction (*Neubauämter*) (UVPG, 2005)
Umweltverträglichkeitsuntersuchung (UVU) durch den Vorhabensträger

Environmental Impact Assessment (EIA) by the plan-approving authority, i.e. the Waterways and Shipping Directorates (WSD) (UVPG, 2005)
Umweltverträglichkeitsprüfung (*UVP*) *durch die Planfeststellungsbehörde (Genehmigungsbehörde)*

Box 5: Historical background of German guidelines for planning procedures for the development and new construction of federal waterways.

1970	National Environmental Policy Act (NEPA), USA „Environmental Impact Statement (EIS)“
1985	Council Directive on the Assessment of the Effects of Certain Public and Private Projects on the Environment.- European Communities (85/337/EEC) “Environmental Impact Assessment (EIA Directive)”, amended in 1997 and in 2003
1990	German Act on the Implementation of the Council Directive of 27 June 1985 on the Assessment of the Effects of Certain Public and Private Projects on the Environment (EIAA), (UVPG, 1990, amended in 1991, 2005)
1994	German Guidelines for Planning Procedures for the Development and New Construction of Federal Waterways. Federal Ministry of Transport, Berlin. (BMVBW, 1994, 2004)
1995	German Administrative Provision on the Implementation of the Environmental Impact Assessment Act (EIAA), on the basis of §20 EIAA (UVPGVwV). Federal Ministry of the Environment, Berlin (BMU, 1995)
1997	Council Directive 97/11/EC of 3 March 1997 amending Directive 85/337/EEC on the Assessment of the Effects of Certain Public and Private Projects on the Environment.-
2001	Criteria for determining the likely significance of environmental effects referred to Article 3(5) (2001/42/EC: SEA Directive) *[Kriterien für die Bestimmung der voraussichtlichen Erheblichkeit von Umweltauswirkungen* (2001/42/EG)*]*
2002	Methodological Framework for the Environmental Risk Assessment.- Federal Ministry of Transport, Building and Housing, Berlin (BMVBW, 2002, 2003)
2004	**Integration of an Ecotoxicological Hazard Assessment Scheme for the Environmental Risk Assessment (ERA)** *[Umweltrisikoeinschätzung (URE)]* in the context of Federal Transport Infrastructure Planning. Federal Institute of Hydrology and Federal Agency for Nature Conservation on behalf of the Federal Ministry of Transport and the Federal Ministry of the Environment, Berlin. (BfG, 2004)
2005	German Act on the Environmental Impact Assessment (EIA) of 25.06.2005 (UVPG, 2005)
2005	German Act on the Strategic Environmental Assessment (SEA) of 27.06.2005 (SUPG, 2005)

The tasks and objectives of the Environmental Risk Assessment in the planning of new-construction and development projects on German federal waterways can be summarized as follows (BfG, 2004):

The **Environmental Risk Assessment** (**ERA**)

- examines probable environmental impacts of planned construction projects in the context of **Federal Transport Infrastructure Planning** (waterways, roads, railways);
- is a tool for the early, rough assessment of projects for planning purposes;
- has the environment as its subject-matter in the sense of the **Act on Environmental Impact Assessment (UVPG**, 1990, 2005), that means, it considers the protected assets "human beings", "animals", "plants", "soil", "water", "air", "climate", "landscape", "cultural and other material assets", including the interactions between them;
- identifies ecologically sensitive areas and potential conflicting interests early at the federal level of transport infrastructure planning;
- registers and evaluates contaminated soils and sediments;
- registers and evaluates the ecotoxicological potential of contaminated soils and sediments by means of a standardized hazard assessment scheme;
- gives recommendations for risk avoidance and mitigation in detailed planning and thus contributes to the optimization of the projects in ecological terms;
- helps to minimize costs, because options requiring much ecological compensation can be ruled out,
- is based on the rationale of the **Environmental Impact Assessment (EIA)**; however, it does not substitute the formal EIA procedure; there remains the necessity of in-depth examinations pursuant to the "**Guidelines for the plan-approval procedure of development and new-construction projects on federal waterways**" (BMVBW, 1994) in the subsequent planning steps;
- is neither the final decision about the environmental compatibility of projects nor an anticipation of decisions of subsequent planning and administrative procedures,
- includes no statement about the feasibility or (legal) admissibility of the project;
- is oriented at projects on a case-by-case basis and does not evaluate the overall impacts of the implementation of the Federal Transport Infrastructure Plan on the whole;
- considers also requirements of the "**Strategic Environmental Review of Plans and Programmes" (*SUP*** *Strategische Umweltprüfung*) (orientation at protected assets, prediction of potentially "significant impacts", recommendations for planning, etc.);
- has the attention of detail required for the level of the Federal Transport Infrastructure Plan and can thus be compiled with a reasonable expenditure of time and funds;
- uses as data source exclusively available databases. Field surveys and costly model studies are not made.

Following the **Act on Environmental Impact Studies for project planning** (**EIS**) (UVPG 1990, 1991, 2005), the Federal Institute of Hydrology (BfG) developed methodologies for application in waterway development and new-construction projects, including a directive on EISs on federal waterways issued by the Federal Ministry of Transport, Building and Housing. Fundamental studies (Esser, 1996, 1997, 1998, 2000, 2001) and work-aids were published (BfG, 1994, 1995a, 1996, 2004), in which a general section describes the administrative procedures, and several annexes present assessment methodologies for waterways along with test methods and orientation values for chemical contamination.

For the first time, **ecotoxicological impact assessment studies** become mandatory, and an **ecotoxicological assessment method** becomes an integral part of environmental risk assessment procedures (Ackermann et al., 2003; BfG, 2004; Krebs et al., 2007).

Requirements for the appraisal of contaminant loads and ecotoxicological effects and the assessment of the degree of environmental pressure

Pursuant to the Environmental Impact Assessment Act (UVPG, 1990, 2005), the *project-related impacts have to be assessed by the difference between the environmental status <u>before</u> and <u>after</u> project execution at the <u>dredging site</u> and at the <u>site of dredged-material disposal</u>*. At both these sites sediments have to be protected and at the dumping site also the surface water, what refers mainly to suspended solids as the carrier of adsorbed contaminants.

Pursuant to the Plan-approval Directive on Waterway Development and New Construction (BMVBW, 1994), the quality of sediments and suspended solids has to be appraised according to a **five-level scale***, with Level 5 (very high value) being <u>oriented at the natural/pristine status</u>*. The ranking in **value levels** 1 to 4 considers the potentially harmful effects of contaminants in the biosphere (BfG, 1996).

The requirements that result from the Environmental Impact Assessment Act (UVPG, 1990, 2005) and from the Administrative Ordinance of the Federal Waterways and Shipping Administration (BMVBW, 1994) for the assessment of the status are met only partially by the HABAB-WSV (2000) and HABAK-WSV (1999) guidelines and the TBT Concept (BfG, 2001a). Their assessment scales have only three levels instead of five, and they are *not oriented at the naturalness but at the existing previous contamination load of the water body* (here represented by suspended solids) and of the sediment. Surface water as an asset to be protected is omitted in the HABAK-WSV coastal guideline. The protected asset according to the EIA Act is here suspended matter.

In the first Federal Plan of Transportation Infrastructure there was no generally accepted assessment scheme for contamination loads in the context of waterway-development projects for which an EIA is mandatory. The determination of the "degree of significance" of construction projects with view to the contaminant load as demanded in the EIA Act (UVPG) was therefore often inconsistent and lacked transparency.

Since the revision of the Federal Transport Infrastructure Plan (BMVBW, 2002a, b; 2003a, b) new construction and development projects on waterways have required an evaluation by the environmental risk assessment scheme (ERA / *URE*). Following this predefined assessment matrix (Figure 4), the German Federal Institute of Hydrology (BfG) formulated criteria for dredging and disposal of contaminated sediment, taking existing regulations of the Federal Waterways and Shipping Administration as far as possible into account (Esser, 2001, Ackermann et al., 2003; BfG, 2004). The ERA-environmental risk assessment scheme for sediments/dredged material described below is applied in all mandatory EIAs. The above-mentioned regulations, HABAB-WSV and HABAK-WSV, find continued application in maintenance works on federal waterways in Germany.

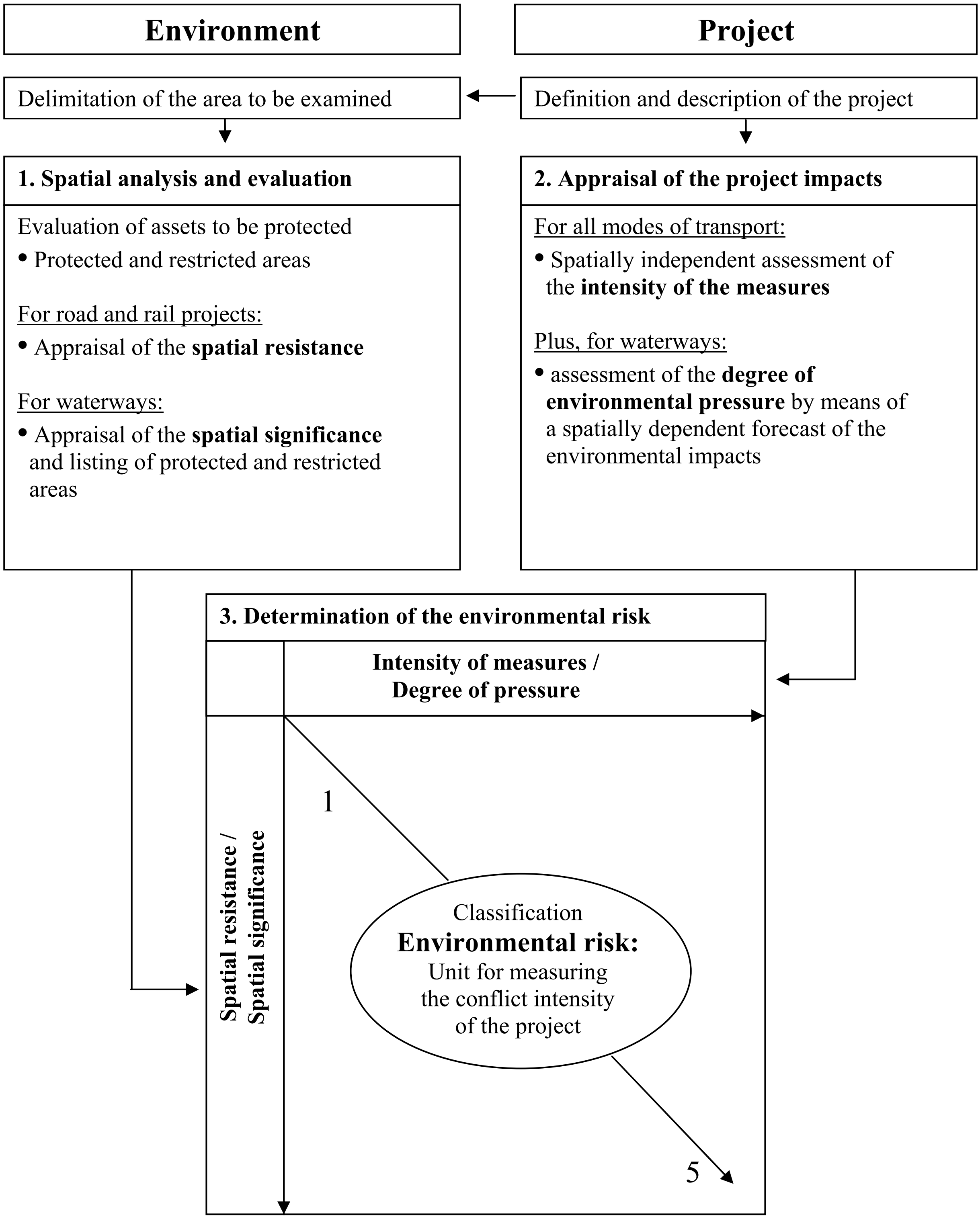


Figure 4: Methodological framework of environmental risk assessment (ERA), matrix for the classification of the conflict intensity of a project into 5 classes (blue, green, yellow, amber and red in Table 8 and 13). (BMVBW, 2002a)

The environmental-risk assessment (ERA) scheme for contaminant loads and ecotoxicological effects: the basic ERA-matrix

The risk assessment of planned new construction and development projects of federal waterways applies a uniform assessment matrix to all environmental assets to be protected (the so-called **basic ERA matrix**) that is a combination of the five-level appraisal of the present state (letters A - E = spatial significance / *Raumbedeutung*) with a three-level assessment of the degree of pressure / *Grad der Belastung* (Roman numerals I to III). This is a highly formalized procedure. The terms "spatial significance" and "degree of pressure " from the ERA terminology are "translated" in this specific context of chemical loads and ecotoxicological effects into "contaminant loads" and "ecotoxicological effects" (*Schadstoffbelastung und ökotoxikologische Wirkung*) and "degree of environmental pressure" (*Grad der Umweltbelastung*) .

Because in the case of contamination of sediments/suspended solids it may happen that dredging does not cause any significant change or may even result in an improvement of the situation, the Level "0" (*Wertstufe 0*)was added to the Levels I (low), II (intermediate), and III (high) in the ranking of the "degrees of pressures" of the ERA scheme according to BMVBW (2002a) (Ackermann et al., 2003; BfG, 2004) (Table 8).

A sediment contamination with heavy-metal compounds or organic substances can have damaging effects on the aquatic environment. By using chemical analysis, only a small fraction of organic contaminants in sediments can be determined. For one thing, their number is very large, and for another, the analytical effort required for their determination would be rather great. Chemical testing is, therefore, limited to relatively few substances of particular importance for sediments and suspended matter. Impacts of contaminants – also of those that are not determined by chemical analysis – can be detected by carrying out ecotoxicological tests (bioassays). Thus, chemical analyses and ecotoxicological tests are complementary to each other in an environmental risk assessment.

Notes concerning Table 8:

The boxes describing the generally applicable basic ERA matrix are uncoloured; the specific application to dredging is shown in the yellow boxes.

The three axis of the basic ERA matrix are described in the following sections:

Y-axis	cf. Section (2) and (4):	<u>Appraisal</u> levels of the contaminant loads and the ecotoxicological effects
X-axis	cf. Section (3) and (5):	<u>Assessment</u> of the degree of environmental pressure
Z-axis	cf. Section (6):	<u>Classification</u> of the project-induced environmental risk

Table 8: Basic ERA matrix to determine the environmental risk of waterway new-construction or development projects for the protected asset “bottom sediment” at the dredging and the placement sites, and for “surface water / suspended matter” at the placement site according to BMVBW (2002a,b), modified for contaminant loads and ecotoxicological effects by Ackermann et al., 2003; Krebs et al., 2007.

Spatial analysis and evaluation

Y-axis (letters): Appraisal of the present state

Spatial resistance ↓ | Spatial significance ↓ | Pollution ↑ | Contaminant loads ↑ | Ecotoxicological effects ↑

Toxicity classes	Value level	0 No	I Low	II Inter-mediate	III High
VI	**A** Very low	1	1	1	3
V	**B** Low	1	1	2	3
III - IV	**C** Inter-mediate	1	2	3	4
I - II	**D** High	1	3	4	5
0	**E** Very high	1	3	5	5
		0	1	2	3

Difference in sediment quality / toxicity classes before and after dredging in the dredging area or at the placement site

Z-axis (Arabic numerals):
Classification of the project impacts in environmental risk levels
(see Table 13)

1	very low environmental risk
2	low environmental risk
3	intermediate environmental risk
4	high environmental risk
5	very high environmental risk

X-axis (Roman numerals):
Assessment of environmental pressure
Measure intensity

Degree of pressure

(2) Appraisal of the contaminant loads

Value levels (A to E) for the appraisal of contaminants in dredged material

The appraisal method of the contamination load does not make any distinction between marine and limnic conditions.

Regarding the bottoms of rivers, lakes, canals, or seas (Baltic Sea and North Sea) the contaminant load in the uppermost sediment strata (top decimetres) at the dredging site and at the placement site of the dredged material before the intervention/project have to be assigned to the value levels A to E according to the basic ERA matrix, cf. Table 8.

The value level “E” refers to natural background values and the value level “D” to quality objectives which are considered as ecologically safe.

Guidance values for the appraisal of contaminants in dredged material

For appraising contaminant loads in dredged material, the guidance values listed in Table 9 are applied. The values are to be considered and used as guidance values (*Richtwerte*) only.

Currently, there is neither in Germany nor elsewhere any objective, purely scientifically based ecotoxicological assessment of the chemical data of dredged material. Despite this fact, in order to be able to carry out the required appraisal of the contaminant content in dredged material, two levels of guidance values corresponding to international directives for dredged material have been introduced.

Regarding **heavy metals**, the five appraisal levels for the contamination load of sediments/ suspended solids in the ERA methodology are oriented, on the one hand, for value level E, at **natural background values** (e.g. clay standard, *Tongesteinstandard*) and, on the other hand, for value level D, at **quality objectives** defined for the protection of inland surface waters by the joint working group of the German Federal States on Water Issues [LAWA (1997a; 1998a), cf. Table 9. The concentrations that were defined as quality objectives (ZV *Zielvorgaben*) are considered as ecologically safe for waters (here: suspended solids and sediments).

For **organic contaminants** too, the value level E refers to natural background values, cf. Table 9. Because no quality objectives for sediments have been defined by LAWA so far, and no other scientifically substantiated orientation values exist for the assessment of organic contaminants in sediments and suspended solids, the value levels E, D and C for organic contaminants use the **guidance values** GV 1 and GV 2 from HABAK-WSV (1999) as reference.
So far, a lower threshold GV1 was defined on the basis of ecotoxicological tests only for tributyl-tin (TBT) (orientation value of the TBT management concept of 20 µg TBT/kg) (BfG, 2001a).

Table 9: Appraisal of contaminant concentrations in bottom sediments, dredged material, and suspended matter. Five-level appraisal of the present state (letters A to E) according to the ERA matrix. Guidance values GV 1 and GV 2 and quality objectives (ZV *Zielvorgaben*) are related to the grain-size fraction < 20µm. Definition of value levels A-E according to Ackermann et al. (2003) and BfG (2004).

Organic contaminant		**nat. BG**	**GV 1**	**GV 2**	**3 x GV 2** (rounded)	**> 3 x GV 2**
PCB 28	µg/kg	0	2	6	20	> 20
PCB 52	µg/kg	0	1	3	10	> 10
PCB 101	µg/kg	0	2	6	20	> 20
PCB 118	µg/kg	0	3	10	30	> 30
PCB 138	µg/kg	0	4	12	40	> 40
PCB 153	µg/kg	0	5	15	45	> 45
PCB 180	µg/kg	0	2	6	20	> 20
Σ 7 PCB	µg/kg	0	20	60	200	> 200
α-Hexachlorocyclohexane	µg/kg	0	0,4	1	3	> 3
γ -Hexachlorocyclohexane	µg/kg	0	0,2	0,6	2	> 2
Hexachlorobenzene	µg/kg	0	2	6	20	> 20
Pentachlorobenzene	µg/kg	0	1	3	10	> 10
p,p‘ – DDT	µg/kg	0	1	3	10	> 10
p,p‘ – DDE	µg/kg	0	1	3	10	> 10
p,p‘ – DDD	µg/kg	0	3	10	30	> 30
Σ 6 PAH (TVO)	µg/kg	50	1000	3000	9000	> 9000
Σ 16 PAH (EPA)	µg/kg	100	2000	6000	18000	> 18000
Hydrocarbons, total	mg/kg	50	300	1000	3000	> 3000
Tributyl tin (TBT)	µg/kg	0	20	60	200	> 200

Heavy metal		**nat. BG**	**ZV**	**2 x ZV**	**4 x ZV**	**>4 x ZV**
Arsenic	mg/kg	10	20	40	80	> 80
Cadmium	mg/kg	0,3	1,2	2,4	4,8	> 4,8
Chromium	mg/kg	80	100	200	400	> 400
Copper	mg/kg	20	60	120	240	> 240
Mercury	mg/kg	0,2	0,8	1,6	3,2	> 3,2
Nickel	mg/kg	30	50	100	200	> 200
Lead	mg/kg	25	100	200	400	> 400
Zinc	mg/kg	100	200	400	800	> 800

Value level	**E**	**D**	**C**	**B**	**A**

Legend:

nat. BG: natural background

GV1, GV2: Guidance values GV1 and GV2 for the chemical assessment of organic contaminants according to HABAK (1999), TBT according to BfG (2001a)

ZV: Quality objectives (*Zielvorgaben*) for the chemical assessment of heavy-metal contaminants according to LAWA (1997a, 1998a)

The lower guidance value GV1 for organic substances is derived from the currently existing regional contaminant concentrations in coastal sediments of the North Sea in Germany. These thresholds are no ecotoxicologically derived quality objectives (*Zielvorgaben or Qualitätsziele*) (LAWA, 1997, 1998), and their observation does not guarantee that the project is ecotoxicologically safe. The guidance value GV2 is calculated from the guidance value GV 1 by multiplication with the factor 3.

The observation of the orientation threshold for contaminants and of the ecotoxicological criteria for the handling of dredged material should prevent a deterioration of the contamination status of sediments and suspended solids through dredged-material relocation projects (principle of no-deterioration).

The evaluation of the contaminant load in the water column (protected environmental asset: surface water, here represented by suspended solids) is based on the three-year mean of contaminant levels in suspended solids (optionally also from recent, freshly settled sediments at the placement site).

If the occurrence of contaminants that are not listed in Table 9 is suspected, case-by-case decisions should be taken and, if necessary, additional contaminants have to be analysed.

The worst ranking of any single substance in each sample of sediment/suspended solid determines the ranking of the whole sample. This assessment comprises heavy metals, organic contaminants, and TBT. The quoted contaminant concentrations always refer to the grain fraction < 20 µm.

Correction for the grain-size effect

For that purpose measurements have to be performed in the separated fraction <20 µm. If the chemical analysis is made differently, the measured data must be corrected (normalized) in relation to the grain-size fraction < 20µm, because contaminants accumulate preferentially in this grain-size fraction. Contaminant concentrations that were measured in whole-sediment samples may vary strongly, simply because of different contents of sandy material that is hardly contaminated. Only the contaminant concentrations in the contamination-carrying fraction < 20 µm are directly comparable.

The TBT levels, too, are usually considered in the fraction < 20 µm, in contrast to the approach chosen in the TBT Concept (BfG, 2001a). In certain cases, such as shipyards, where sediments may contain coarse-grained particles of anti-fouling paint with very high TBT concentrations, the TBT levels of the whole sample must be taken into account.

(3) Assessment of the "degree of environmental pressure" by means of a contamination-dependent forecast of environmental impacts

Assessment of the environmental pressure in terms of the contaminant load

The project-induced changes in contaminant levels in the sediments at the dredging and placement sites or in the suspended solids at the placement site are ranked in the Levels 0 to III. Like in the differentiation of cases according to HABAB-WSV (2000), the factor by which the contaminant levels in the new sediment surface after dredging and the new river bottom resulting from the placement of dredged material (c_{new}) exceed the original concentrations (c_{old}) assigns the sites to the Levels 0 to III (Table 10).

Accordingly, regarding surface water, the contaminant concentrations measured in the sediment to be dredged and relocated (c_{DM}) are compared with the concentrations in suspended matter (c_{SM}) (3-year mean) at the placement site (Table 10).

The worst ranking of every individual substance is decisive for the overall ranking.

Table 10: Assessment of the environmental pressure in terms of the contaminant load according to Ackermann et al. (2003) and BfG (2004).

Asset to be protected: "Aquatic Sediment"	**Asset to be protected: "Surface Water"**	**Assessment of the Degree of Environmental Pressure**	**HABAB-WSV (2000)**
Dredging Site and Placement Site	**Placement Site**		
$\mathbf{c_{new}} \leq c_{old}$	$\mathbf{c_{DM}} \leq c_{SM}$	**0** **no effect**	**Case 1**
$c_{old} < \mathbf{c_{new}} \leq 1,5\ c_{old}$	$c_{SM} < \mathbf{c_{DM}} \leq 1,5\ c_{SM}$	**I** **low effect**	**Case 1**
$1,5\ c_{old} < \mathbf{c_{new}} \leq 3\ c_{old}$	$1,5\ c_{SM} < \mathbf{c_{DM}} \leq 3\ c_{SM}$	**II** **intermediate effect**	**Case 2**
$\mathbf{c_{new}} > 3\ c_{old}$	$\mathbf{c_{DM}} > 3\ c_{SM}$	**III** **high effect**	**Case 3**

Legend:

DM Dredged Material,
SM Suspended Matter
c Concentration, grain-size corrected (related to the grain-size fraction < 20 µm)
$\mathbf{c_{old}}$ Concentration in the original (old) sediment surface
$\mathbf{c_{new}}$ Concentration in the new sediment surface
$\mathbf{c_{DM}}$ Concentration in the dredged material
$\mathbf{c_{SM}}$ Concentration in the suspended matter

Moreover, the following special regulations apply:

If the silt and clay content of dredged material assigned to Levels II or III and II is < 10 %, it can be ranked in Level I (low impact) or Level II (intermediate impact), respectively.

In addition to contaminant levels, also particle-bound contaminant loads are taken into consideration. If the dredging and relocation of the material produce a very low additional load of suspended solids in comparison with the annual suspended-solids load and thus add relatively little to the annual contaminant load, the impact of such a project may be assessed more favourably, i.e. instead of Level III, Level II may be assigned.

Conversely, when the volumes of material to be relocated are likely to produce suspended-solid loads in the order of magnitude of the annual load, each case should be considered separately, what might result in a more unfavourable assessment.

Explanation of terms and concepts:

Protected asset "<u>aquatic sediments</u>": a) at the <u>dredging site</u>

The ranking of the degree of environmental pressures in the Levels 0 to III takes account of the project-induced changes in the contamination of the bottom of the water bodies at the dredging-project site and uses HABAB-WSV (2000) as a guideline (Table 10).

Level 0 "no pressure"
This zero-level that was additionally introduced characterizes *projects which do not deteriorate the contamination in the sediment at the dredging site or even improve it.*
► This means that *"no decision-relevant environmental risk"* was identified.

Level I "low pressure"
Dredging does not increase the concentration of any contaminant in the newly created sediment surface <u>beyond the 1.5-fold</u> of the concentration in the original sediment surface
► *no release of more heavily contaminated old deposits.*

Level II "intermediate pressure"
Dredging increases the concentration of at least one contaminant in the newly created sediment surface to the level <u>between the 1.5-fold and the 3-fold</u> of the concentration in the original sediment surface
► *release of more heavily contaminated old deposits.*

Level III "strong pressure"
Dredging increases the concentration of at least one contaminant in the newly created sediment surface <u>beyond the 3-fold</u> of the concentration in the original sediment surface
► *release of significantly more heavily contaminated old deposits.*

Additional assessment criterion
Old contaminated deposits to be dredged or uncovered by dredging, i.e. sediment areas that have against their surroundings significantly higher and untypical contaminant concentrations are assessed on a case-by-case basis. This applies even to very small amounts e.g. oil spills on the shore, barrels containing chemicals.

Protected asset "aquatic sediments": b) at the dredged-material placement site

The project pressure at the placement site is assessed through a comparison of the contamination of the dredged sediments with the contamination of the bottom of the water body at the place-ment site. The assessment levels 0 to III are applied according to the above Section (Table 10).

Protected asset "surface water" at the dredged-material placement site

Here represented by suspended solids.

The ranking of the project pressures on the protected asset "surface water" in the Levels 0 to III takes account of the project-induced changes in the contamination of the suspended solids at the dredging-project site and uses HABAB-WSV (2000) as a guideline (Table 10). If no updated data on the contamination level in suspended solids are available, fine-grained sediment consisting of freshly settled suspended solids may be used as reference.

Level 0 "no pressure"
Dredged-material placement does not increase the concentration of any contaminant beyond the 3-year average of the contaminant concentration in suspended solids. This means that "no decision-relevant environmental risk" was identified.

Level I "low pressure"
Dredged-material placement increases the concentration of at least one contaminant in the dredged material above the mean level of the contaminant concentration at the placement site, but not beyond the 1.5-fold of this value (HABAB-WSV principle).

Level II "intermediate pressure"
Dredged-material placement increases the concentration of at least one contaminant in the dredged material above the 1.5-fold of the mean contaminant concentration at the placement site, but not beyond the 3-fold of this value.

If the amount of dredged material to be relocated is less than 2 % of the annual suspended-solids load (NOTE: not applicable in tidal waters) and/or the sand content is > 90 %, the ranking may be in Level I “low pressure".

Level III "strong pressure"
Dredged-material placement increases the concentration of at least one contaminant in the dredged material beyond the 3-fold of the mean concentration in suspended solids at the placement site.

If the amount of dredged material to be relocated is less than 2 % of the annual suspended-solids load (NOTE: not applicable in tidal waters) and/or the sand content is > 90 %, the ranking may be in Level II "intermediate pressure".

(4) Appraisal of the ecotoxicological effects

Appraisal method for ecotoxicological effects

The pT-method is used for appraisal and comparison of the toxic potential of sediments and dredged material. Both objective and scope of the method and the determination of the pT-values and the pT-indices are described in Chapter 2.1.1.

Value levels (A to E) for the appraisal of the ecotoxicological effects

The ecotoxicological assessment in the framework of environmental risk assessment ERA (*URE*) follows the definition of categories of the directives on inland and coastal waterway dredging (HABAB-WSV, 2000, HABAK-WSV, 1999).

These directives use the *pT-index as a management tool to incorporate bioassay data into the decision-making process for assessing and comparing the relative toxic hazards of sediments and dredged material* according to Krebs (2000, 2001, 2005a,b).

The ecotoxicity of sediments and dredged material is defined and determined operationally by specified bioassays for the test batteries pursuant to the directives on inland and coastal waterway dredging. The ranking is described in Table 11.

Table 11: Appraisal of ecotoxicological effects measured in bottom sediments, dredged material, and suspended matter. Five-level appraisal of the present state (letters A to E) according to the ERA matrix. Definition of value levels A to E according to Ackermann et al. (2003) and BfG (2004).

Test result of a battery of bioassays pT_{max} value	Toxicity class pT-index	Value level
all pT values = 0	**Toxicity class 0**	**E Very high**
pT_{max} value = 1 or 2	**Toxicity class I or II**	**D High**
pT_{max} value = 3 or 4	**Toxicity class III or IV**	**C Intermediate**
pT_{max} value = 5	**Toxicity class V**	**B Low**
pT_{max} value = 6 or > 6	**Toxicity class VI**	**A Very low**

(5) Assessment of the "degree of environmental pressure" by means of an ecotoxicity-dependent forecast of environmental impacts

Assessment of the environmental pressure in terms of ecotoxicological effects

Parallel to the assessment of the chemical data, an ecotoxicological assessment regarding the pressure of the project on the assets to be protected must be made. This assessment needs toxicity data from the dredged material itself and from the uncovered sediment layers at the site of dredging, as well as from the surface sediment and from the suspended matter at the site of dredged material placement. The steps of the procedure are analogue to those of the chemical assessment.

The assessment of the pressure at the dredging site in ecotoxicological terms results from the comparison of the toxicity of the bottom sediments with the toxicity of the uncovered sediment strata and at the placement site from the comparison of the toxicity of the bottom sediments with the toxicity of the dredged material. For the protected asset "surface water", the toxicity of the dredged material is compared with the toxicity of suspended solids at the site of placement (cf. Section on the chemical load). This comparison considers individually for each compartment the toxicity classes, which are derived from the pT_{max} values that were determined by standardized bioassay test batteries (cf. Chapter 2.1.1; Boxes 1 and 2; Krebs, 2005a, b). The assessment method is listed in Table 12.

Table 12: Assessment of the "environmental pressure of the project" for ecotoxicological effects according to the ERA methodology: The degree of environmental pressure is calculated from the difference between the sediment quality before and after dredging in the dredging area and at the placement site. The quality is expressed in toxicity-classes which are derived from pT_{max}- values. Definition of the degrees according to Ackermann et al. (2003) and BfG (2004).

Determination of the assessment classes	Assessment of the Degree of Environmental Pressure
No difference between the toxicity classes	0 no effect
Difference between toxicity classes / pT_{max} values by 1	I low effect
Difference between toxicity classes / pT_{max} values by 2	II intermediate effect
Difference between toxicity classes / pT_{max} values by 3	III high effect

Explanation of terms and concepts:

Level 0 "no" or "positive effect"
Level 0 is the ranking for projects when the toxicity classes of the examined compartments do not differ; i.e. the project does not cause any deterioration. If the project should even cause an improvement of the environmental situation, it is also ranked in this level. Level 0 means that there is no observable decision-relevant environmental risk.

Level I "low effect"
If the project leads to an increase in toxicity class by 1, the project impact is ranked at Level I. At the dredging site, Level I signifies that the uncovered underlying sediment layer is one class more toxic than the dredged material itself.
At the placement site, Level I indicates that the dredged material is one class more toxic than the bottom sediment or the suspended solids at the placement site.

Level II "intermediate effect"
If the project leads to an increase in toxicity class by 2, the project impact is ranked at Level II. At the dredging site, Level II signifies that the uncovered underlying sediment layer is two classes more toxic than the dredged material itself.
At the placement site, Level II indicates that the dredged material is two classes more toxic than the bottom sediment or the suspended solids at the placement site.

Level III "high effect"
If the project leads to an increase in toxicity class by 3, the project impact is ranked at Level III.
At the dredging site, Level III signifies that the uncovered underlying sediment layer is three classes more toxic than the dredged material itself.
At the placement site, Level III indicates that the dredged material is three classes more toxic than the bottom sediment or the suspended matter at the placement site.

In the case of highly polluted sediments, the open-end pT-scale has to be used. If the pT_{max}-value of the dredged material is 6 (toxicity class VI), then the uncovered sediment layer must have had a pT_{max}- value of 9 to get the “degree of environmental pressure” of III (Tables 8 and 12). The highest pT-value so far measured in bottom sediments was 11 (Krebs, 2005b).

The Environmental Hazard Assessment Scheme described above for ecotoxicological effects is likewise applicable to sediments and suspended solids. However, to date very few studies on suspended matter have been made, because it is difficult to collect suspended solids in such amounts as are needed for ecotoxicological tests. In the future, research in this field should be intensified.

(6) Classification of the project-induced environmental risk

Classification of the environmental risk

The classification of the project-induced environmental risk according to the ERA methodology “very low”, “low”, “intermediate”, “high”, and “very high” is based on the highly formalized procedure of the **basic ERA matrix.**

The matrix comprises a co-ordinate system with 3 axes (**X-, Y-, and Z-axes**):

The value levels A to E for the **appraisal of the present state** are put on the “**Y- axis**” and the assessment levels 0 to III for the **project effects – the environmental pressure** - on the “**X-axis**” (Table 8).

The method of assessment of the degree of environmental pressure by means of a pollution-dependent forecast of the **environmental impacts** is specified on the “**Z-axis**” pursuant to the directive of the Federal Transport Infrastructure Plan (BMVBW 2002a, b and 2003a, b)

The five-level **classification of the environmental risk** for measuring the conflict intensity of a waterway new-construction or development project is listed in Table 13.

Definition of "significant impairment" in the sense of the Environmental Impact Study (EIS) pursuant to the German Act UVPG (2005)

In contrast to the ERA / *URE* procedure, the Environmental Impact Study for project planning (EIS / *UVU*) has also to check or to predict whether the construction project entails “significant and / or lasting impairments”.

Future environmental impact studies in the Federal Waterways and Shipping Administration (WSV) should predict a significant impairment / *erhebliche Beeinträchtigung* whenever the above-described assessment scheme identifies a high or very high environmental risk. This assessment scheme ranks the environmental risk due to a project-induced intensification of the contaminant load or of ecotoxicological effects lower in areas, where the ecological status is assessed as poor, than in such areas, where the ecological value is rated as good.

Table 13: Classification of the environmental risk: Five units for measuring the conflict intensity of a waterway new-construction or development project (BfG, 2004).

Classification of the environmental risk pursuant to the ERA matrix ("Z-axis")	
Units of the conflict intensity of the project	**Appraisal of the project impacts**
1 **Very low**	No decision-relevant environmental risk identified No compensation and substitution measures to be expected.
2 **Low**	No increased environmental risk. Most existing environmental risks may be avoided or minimized. Compensation and substitution measures to be expected.
3 **Intermediate**	Significant environmental risks exist. They can be avoided or minimized only partially. Comprehensive compensation and substitution measures to be expected.
4 **High**	Significant environmental risks to be expected in essential parts of the study area or for several assets to be protected/sub-complexes. These risks can be avoided, minimized or compensated only with significant expenses. Comprehensive substitution measures to be expected.
5 **Very high**	Significant environmental risks to be expected in large parts of the study area or for most assets to be protected/sub-complexes. These risks cannot always be avoided, minimized or compensated. Very comprehensive substitution measures to be expected.

(7) Examples of environmental risk assessment and compatibility assessment pursuant to the Habitats Directive of projects on federal waterways by the Federal Institute of Hydrology (BfG)

The **environmental risk assessment (ERA)** and the **compatibility assessment (CA) of the habitats directive (Habitats-CA)** (Council Directive 92/43/EEC, *Flora-Fauna-Habitat-* (*FFH*)*-Richtlinie, FFH-CA*) are procedures to examine the expected regional environmental consequences of projects that are scheduled for inclusion into the Federal Plan of Transport Infrastructure of the German government (BMVBW 2002a, b and 2003a, b). They also provide information about ecologically sensitive areas and potential conflicts of interests. Moreover, they identify possibilities for avoiding or mitigating impairments of the natural balance. Thus, they contribute to the ecological optimization and to cost reductions in waterway-construction projects.

ERA and Habitats-CA studies use exclusively data that are already available. Accordingly, these assessments are less detailed than the Environmental Impact Studies (EIS) or the Habitats Compatibility Studies (Habitats-CS) that have to be made in the next planning phase of such projects. The final decisions about the environmental compatibility of a project, its feasibility or its acceptability will be taken only at the later planning level of the EIS and the Habitats-CS.

Besides developing the methodology, the Federal Institute of Hydrology (BfG) is also performing assessments of waterway projects of the Federal Waterways and Shipping Administration (*WSV*) regarding the environmental risks they pose and their compatibility assessment pursuant to the habitats directive (Habitats-CA). In the years 2004/05, such assessments were made for projects of fairway extensions in the Lower and Outer Elbe River, in the Outer Weser River, in the Kiel Canal, and in the northern Peenestrom (Figure 5).

The assessments of the fairway extensions in the Lower and Outer Elbe River, in the Outer Weser River, and in the northern Peenestrom rated the environmental risk as medium, while the ERA on the development of the eastern reach of the Kiel Canal found a high environmental risk (Schmitt & Fiedler, 2006).

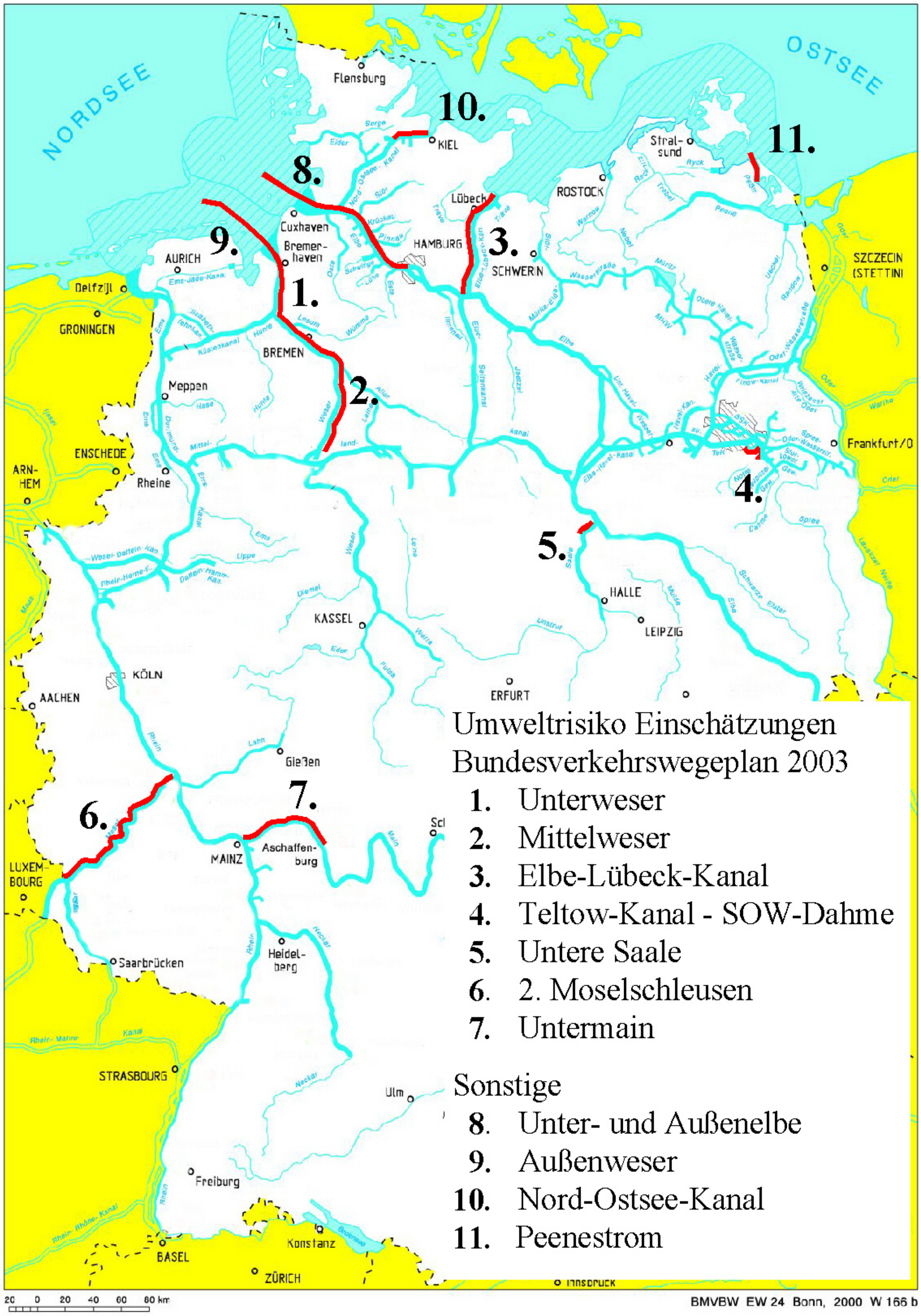


Figure 5: Environmental risk assessments (ERA) of projects on federal waterways for the Federal Transport Infrastructure Planning 2003 performed by the Federal Institute of Hydrology (Figure BMVBW EW 24).

4.2 Statutory order in The Netherlands for disposal of dredged material at sea

4.2.1 Management of disposal of dredged material at sea: The Chemistry-Toxicity Test (CTT) - Approach

The CTT is used to assess whether the relocation of dredged material is acceptable. If it does not meet the quality criteria in the CTT, it may not be relocated in the marine environment. The CTT consists of a combination of chemical and biological assessment criteria. Qualifying and disqualifying standards are associated with the chemical assessment criteria. For hazardous substances the criteria are directly disqualifying (one out all out). If more than two non-hazardous substances exceed the relevant standards by more than 50%, the dredged material in question is 'disqualified'. In other words, it may not be relocated in the marine environment. Since analyzing and setting standards for all potential problematic substances in dredged material is a time-consuming business, the CTT includes three bioassays. They are used to determine the toxicity (combined or otherwise) of the dredged material. At the time the CTT was published in the *Staatscourant*, no representative dataset of sufficient size was yet available for the bioassays. Furthermore, more detailed protocols and certification were needed to ensure uniform and reliable analysis with the bioassays. The qualifying and disqualifying standards have not, therefore, been set for the bioassays as of yet. For the time being, therefore, only a monitoring obligation and a signal function apply. If the signal value is exceeded, the causing factor must be further investigated.

Tributyltin

Tributyltin (TBT) is the most problematic among all the chemicals tested in assessing whether dredged material may be relocated in the marine environment. TBT is highly harmful to the environment and is used, among other compounds, in antifouling paint for ships. In view of its harmfulness, a treaty has been drawn up under the auspices of the International Maritime Organization (IMO) banning the use of TBT on seagoing vessels. The treaty will come into effect once the required number of member states has ratified it. In anticipation of this, the EU issued a regulation in 2003 banning the use of TBT on ocean-going vessels in EU member states. These international developments will reduce the TBT burden on the marine environment.

Evaluation of the CTT

The Staatscourant text publishing the CTT announced the development of an integrated future vision on dealing with sediment in marine water systems and, partly with a view to the development of this vision, an evaluation of the CTT. The deadline for publication of the evaluation was July 2006. The results from the evaluation of the CTT covered (Schipper and Klamer, 2006) the following points:

1. *Developments associated with the IMO Convention and scope for tightening up TBT norms:*

The international effort to bring to an end the environmental burden posed by TBT has not produced a substantial improvement in the quality of the marine environment. This is partly because programmes of measures still have to be established under the WFD. Furthermore, the IMO convention banning the use of TBT on ocean-going vessels has not yet been ratified. In the Netherlands in 2007, it is proposed that an unambiguous TBT norm of 250 μg Sn/kg d.s. be introduced for the assessment of dredged material in marine areas covered by the Pollution of Surface Waters Act (the Zeeland Delta and Wadden Sea). This would be enough to preclude the possibility of a sharp reversal in the amount of dredged material being dispersed in the marine environment. This upper limit is in line with the TBT contents in the

Wadden Sea and Zeeland waters, and with the TBT norm applied in Germany. A TBT norm of 115 µg Sn/kg d.s. is proposed for the North Sea coast (the area covered by the Pollution of Marine Waters Act). This would avoid any unnecessary extra dispersal up to the level of a higher norm, as a more generous TBT norm is not needed to avoid any sharp trend reversal. These TBT norms would be tightened up step by step as the quality of the marine environment improved under national, European and global measures to end TBT pollution.

2. *The Water Framework Directive, to curb the burden on aquatic systems caused by priority hazardous substances and other relevant substances:*

The Water Framework Directive and European Marine Strategy have not yet impacted on dispersal policy and the assessment system used for the purpose.

3. *The development of a sustainable, integrated future vision of how to deal with marine sediment, which might have implications for assessment systems like the CTT:*

The evaluation has revealed that the CTT bioassays examined are not suitable for disqualifying dredged material. They will therefore no longer constitute part of the assessment system. This included an assessment of the suitability of bioassays for disqualifying dredged material or performing an alert function. The performance characteristics of both the *Corophium volutator* and the Microtox Solid Phase test were not adjudged to be adequate for a disqualifying role in an assessment system such as the CTT. This also applies to any alert function in a monitoring system. The DR-CALUX test is however sufficiently robust to be used in a monitoring system for persistent, bioaccumulating and toxic substances.

4.2.2 Management of disposal of dredged material at sea: the 'Saline-Dredged-Material-Test' ('Zoute-Bagger-Toets' =ZBT) - Approach

Since this will remove the biological component of the CTT, it has been proposed that the name be changed to '**Saline-Dredged-Material-Test**' in 2007 ('**Zoute-Bagger-Toets**' =**ZBT**) (Figure 6). The ZBT will be introduced as the successor to the CTT for assessing whether dredged material can be dispersed in the marine environment. Bioassays could still be used as a safety net, particularly for new problem substances in monitoring systems designed to safeguard water quality in a broader sense than simply aquatic sediments.

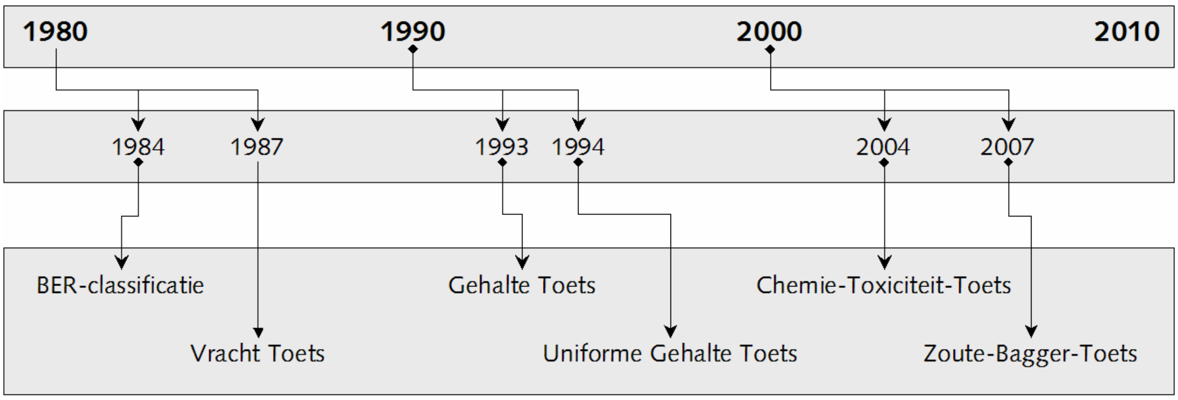


Figure 6: Historical development of norms for sediment management (Schipper and Klamer, 2006).
BER-classificatie = BER classification; Vracht Toets/VT = Freight Test; Gehalte Toets = Content Test; Uniforme Gehalte Toets/UGT = Uniform Content Test; Chemie-Toxiciteit-Toets/CTT = Chemical Toxicity Test; Zoute-Bagger-Toets/ZBT = Saline Dredged Material Test.

Under the new ZBT policy (see Table 14), regional water management plans will be devised, providing for a monitoring system that measures the quality of the water system in a broader sense, including its ecological functioning (Van Zundert, 2006). The water management agency and port authorities will reach agreement as to the latter's input into the monitoring system. This input will involve monitoring the impact of dredging activities and the relocation of dredged material in the marine environment on the functions, quality objectives and ecological objectives assigned to the water body in question.

Table 14: Proposed chemical criteria for Saline-Dredged-Material-Test/ZBT of dredged marine sediment.

Chemical name	**Group**	**Units**	**Test value**
Tributyltin	Organometal	µg Sn/kg dw	115-250
Copper	Metal	mg/kg dw	60
Arsenic	Metal	mg/kg dw	29
Cadmium	Metal	mg/kg dw	4
Mercury	Metal	mg/kg dw	1.2
Chromium	Metal	mg/kg dw	120
Zinc	Metal	mg/kg dw	365
Nickel	Metal	mg/kg dw	45
Lead	Metal	mg/kg dw	110
Sum 10 PAH	PAH	mg/kg dw	8
Hexachlorobenzene	OCP	µg/kg dw	20
Sum DDT/DDD/DDE	OCP	µg/kg dw	20
Mineral oil C10-40	Oil	mg/kg dw	1250
Sum 7 PCB	PCB	µg/kg dw	100

4.2.3 Future vision and the new Saline Dredged Material Test/ZBT

The TBT norms would be tightened up step by step as the quality of the marine environment improved under national, European and global measures to end TBT pollution. This is in line with the German approach. The evaluation gives no grounds for adjusting the test values for other substances in the CTT. The possibility of giving bioassays an alert function is currently being investigated.

In the future vision on disposal of dredged material (Schipper, Klamer, Hin and Bel, 2007) the preferred option would be to embed them in a monitoring system for the quality of the water system in a broad sense, based on integrated water management. Water management and port authorities are looking into the possibility of drawing up a voluntary agreement for the purpose, perhaps covering the following areas:

- the use of bioassays to signal the presence of certain groups of known or unknown substances and mixtures of substances;
- the identification and possible tackling of local sources of pollution, including aquatic sediments that were severely polluted in the past;
- the dredging and disposal of material from the perspective of the ecological objective of the WFD and the EMS and, where useful and necessary, the further optimisation of its ecological compatibility.

In the future vision, disposal of dredged material will no longer be regulated on the basis of the idea that it is waste; sediment will be regarded as a natural and valuable part of the water system. If the target vision is achieved, the ZBT will no longer be used to assess each individual batch of dredged material to see whether it can be disposed of in the marine environment. Instead, a monitoring system would be introduced to safeguard water quality in a broader sense, taking into account other uses and quality objectives.

The findings of the evaluation of the CTT will be included in the integrated future vision for dredged material in the marine environment, and might also serve as a basis for drawing up a covenant between central government and port authorities.

4.2.4 Statutory order for Environmental Impact Assessment (EIA) in The Netherlands

An **Environmental Impact Assessment** (**E.I.A.**) provides the information needed to fully weigh up the environmental impact before decisions are taken on plans and projects with major environmental consequences. The assessment states the environmental consequences of a plan or project and gives any possibly more environmentally-friendly alternatives. An E.I.A. is mandatory when building confined disposal sites and often for projects where beneficial reuse of dredged material is planned. Besides the E.I.A. there is also the EIR. E.I.A. stands for environmental impact assessment, while the EIR is the environmental impact report. The EIR is part of the E.I.A. procedure followed to arrive at a decision or activity.

The legal provisions for E.I.A.

The E.I.A. is regulated in chapter 7 of the **Dutch Environmental Management Act** (**Wm**) and in the **Environmental Impact Assessment Decree** 1994 (Besluit M.e.r.1994). This Act is a framework act describing the basic principles of environmental policy. The details are provided for in orders in council (AMvB). The Environmental Impact Assessment Decree 1994 is such an order in council. Other important passages in the Environmental Protection Act on the E.I.A. besides Chapter 7 can be found in Chapter 2.2 (about the E.I.A. Commission), Chapter 14.2 (about the coordination required when drawing up an environmental impact report) and chapter 20 (appeal).

The Environmental Impact Assessment Decree 1994 states when an E.I.A. should be carried out. The decree contains appendices, which include the C and D lists. The C list indicates which activities and decisions require a mandatory environmental impact report. The D list sums up the activities and decisions for which a so-called 'article 7.8a/7.8d procedure' is required. These activities and decisions are evaluated on an individual basis to see whether an E.I.A. is necessary. (The lists can be found in the Besluit m.e.r. 1994 (in Dutch)).
The Environmental Impact Assessment Decree 1994 resulted from a European Directive for E.I.A. (officially known as Directive 97/11). It also incorporates the United Nations Economic Commission for Europe (Unece) treaty on E.I.A. for transboundary environmental impacts (Espoo treaty).

An E.I.A. procedure comprises 10 steps:

1. Pre-starting note: the initiator writes the pre-starting note. This document contains the basic data for the project. The procedure can start when the competent authority publishes the pre-starting note.

2. Public participation and recommendations: there are usually 4 weeks set aside for public participation. Participation is open to everyone. This participation and the recommendations focus on the guidelines for the desired content of the environmental impact report. An important element of this is the recommendations for the Commission's guidelines for the environmental impact assessment.

3. Guidelines: within 13 weeks of the publication of the pre-starting note the competent authority will set the guidelines. These indicate which alternatives and which environmental impacts have to be dealt with in the environmental impact report.

4. Environmental impact report (EIR): the initiator is responsible for writing the report. This is not subject to any time limit. Good interaction with project development is recommended in this step. When the environmental impact report is ready, the initiator sends it to the competent authority together with the request for a decision.

5. Acceptability assessment: within 6 weeks of the environmental impact report being submitted, the competent authority assesses whether the environmental impact report meets the guidelines (the desired content) and legal requirements. The competent authority also checks whether the application can be considered.

6. Publication of environmental impact report and application or draft decision: the competent authority publishes the report together with the application for the decision within 8 weeks so that participation and recommendations can take place. If the decision does not require an application for a decision to be submitted, the environmental impact report will be published with the draft decision or preliminary draft decision.

7. Participation, recommendations and hearing: anyone may comment on the environmental impact report and raise objections to the application or the draft decision. The deadline is at least 4 weeks but follows the period for objections to the procedure for the decision.

8. Testing by the environmental impact assessment Commission: once the period for public participation has ended, the environmental impact assessment Commission publishes its report on the completeness and the quality of the environmental impact report within 5 weeks. The comments and recommendations that have been received will be taken into account when compiling the report.

9. Decision: the competent authority takes the decision on the project. In so doing, it takes account of the environmental impacts and the reactions and recommendations that have been received. In the decision it explains what has been done with the result of the environmental impact report. It also specifies what is to be assessed and when. The regulations for making objections and appeals result from the regulations in the decision.

10. Assessment: with the cooperation of the initiator, the competent authority assesses the environmental impacts that actually occur, as laid down in the assessment section of the decision. Where necessary, it takes extra measures to limit the impact on the environment. An objection or an appeal must be submitted within six weeks.

Strategic environmental assessment

The EU Strategic Environmental Assessment Directive (Directive 2001/42/EC) was adopted in July 2001. This Directive lays down the rules for a mandatory environmental impact assessment for strategic decisions. This means, for example, that plans for spatial planning or waste management must be checked for any impact they may have on the environment.

5. Risk assessment of the in situ quality of polluted sediments for remediation

5.1 Statutory order of risk assessment of polluted sediments in Germany

Special procedures have been proposed for a risk assessment step, but none have yet been implemented in a legal framework. Risk assessment is based on a combination of exposure characterization and effect measurements, a general scheme proposed is given by the USEPA (USEPA, 1998).
An inventory of bioassays and biological classification methods is made in Germany to establish a biological sediment guideline for the rivers Elbe and Rhine (Zimmer & Ahlf, 1994, Ahlf & Gratzer, 1999). Several other research groups also recommend the triad approach according to Chapman (2000) for Germany. Ahlf (1995) proposed a sediment classification scheme for screening of sediment quality.
Recommendations were also made for the use of an integrated stepwise approach combining toxicological, chemical and ecological information to assess and evaluate the quality of sediments. Henschel et al. used a stepwise approach for an integrated assessment of ecosystem health effects and the consequences of sediment contamination for human health. A difference with the approaches followed in most other countries is that bioassays are used as a trigger for further research steps, instead of chemical data that is more commonly used, see Henschel et al. (2001c,d; 2003a,b).

5.2 Statutory order of risk assessment of polluted sediments in The Netherlands

For freshwater systems in The Netherlands, assessment of in situ sediment quality is required within the legal framework of the **Soil Protection Act** (Ministry of Housing, Spatial Planning and the Environment, 1994). This act requires **tiered risk assessment** as a second tier for sediments with contaminant concentrations above the **intervention value**, in order to determine the urgency of remedial action.

In order to evaluate human risks, model calculations are carried out in order to quantify the extent to which humans (adults/children) can be exposed to contaminant via food consumption or via recreation activities in water. When the exposure exceeds maximum permissible risk criteria, actual risk is concluded. The model is based on general assumptions with regard to behaviour and diet of human populations.

Investigation of the risk for transport of contaminants from the sediment to groundwater, or to surface water. Model calculations are carried out in order to quantify the extent to which these processes occur. When contaminant fluxes (preferably calculated from field data) exceed high risk criteria, actual risk is concluded. For the ecological risks the assessment is outlined in Figure 7.

1. *1st tier assessment*: In tier 1 the toxic pressure on aquatic organisms is calculated using the model OMEGA, and with bioavailable concentrations of contaminants in the sediment as the model input. The model will calculate the potentially affected fraction of species (PAF; Posthuma et al., 2002). The same methodology has been developed for the assessment of risks of soil contamination (Mesman et al., 2003). With this model, direct effects and effects as a result of foodchain poisoning can be distinguished. In the Netherlands, mild extraction techniques with $CaCl_2$ or Tenax are used for measurement of the contaminant concentrations considered to be bioavailable (Cornelissen et al., 2001; see also Van Elswijk et al., 2001).

2. *2nd tier assessment:* Assessment of ecological risks. The evaluation of risks for the ecosystem is done by using the TRIAD assessment. In the Dutch version of the TRIAD, bioaccumulation measurements are also considered, using the results of laboratory tests, or preferably by measurements in indigenous organisms (Den Besten et al., 1995). Based on the most sensitive parameter, sediments are classified for the categories "field observations" and "bioassays" as either "-" (no effect/risk), "±" (moderate effect/risk) or "+" (strong effect/high risk). The goal is to elucidate the relationship between effects on macrozoobenthos and responses of bioassays which, in turn, can be related to levels of chemical pollution. For that purpose, chemical concentrations are converted into "toxic units" (TU): these are the ratio between the chemical's normalized concentration and the lowest NOEC reported in the literature, among the bioassays included in the battery (Den Besten, 1995). High risk is inferred when strong effects are observed in field surveys and/or bioassays that can be related to chemicals present in the sediment (see Van Elswijk et al., 2001).

3. *Prioritization.* When the supplied data from the second tier show that there are actually no high risks at a site where a priority pollutant exceeds the intervention value, the need for remediation is not urgent anymore. In case actual high risks were confirmed, a next step is possible in which different remedial options are considered for the risk reduction that can be achieved. The information from the sediment quality assessment can be used again in setting priorities within the group of locations that need to be remediated urgently.

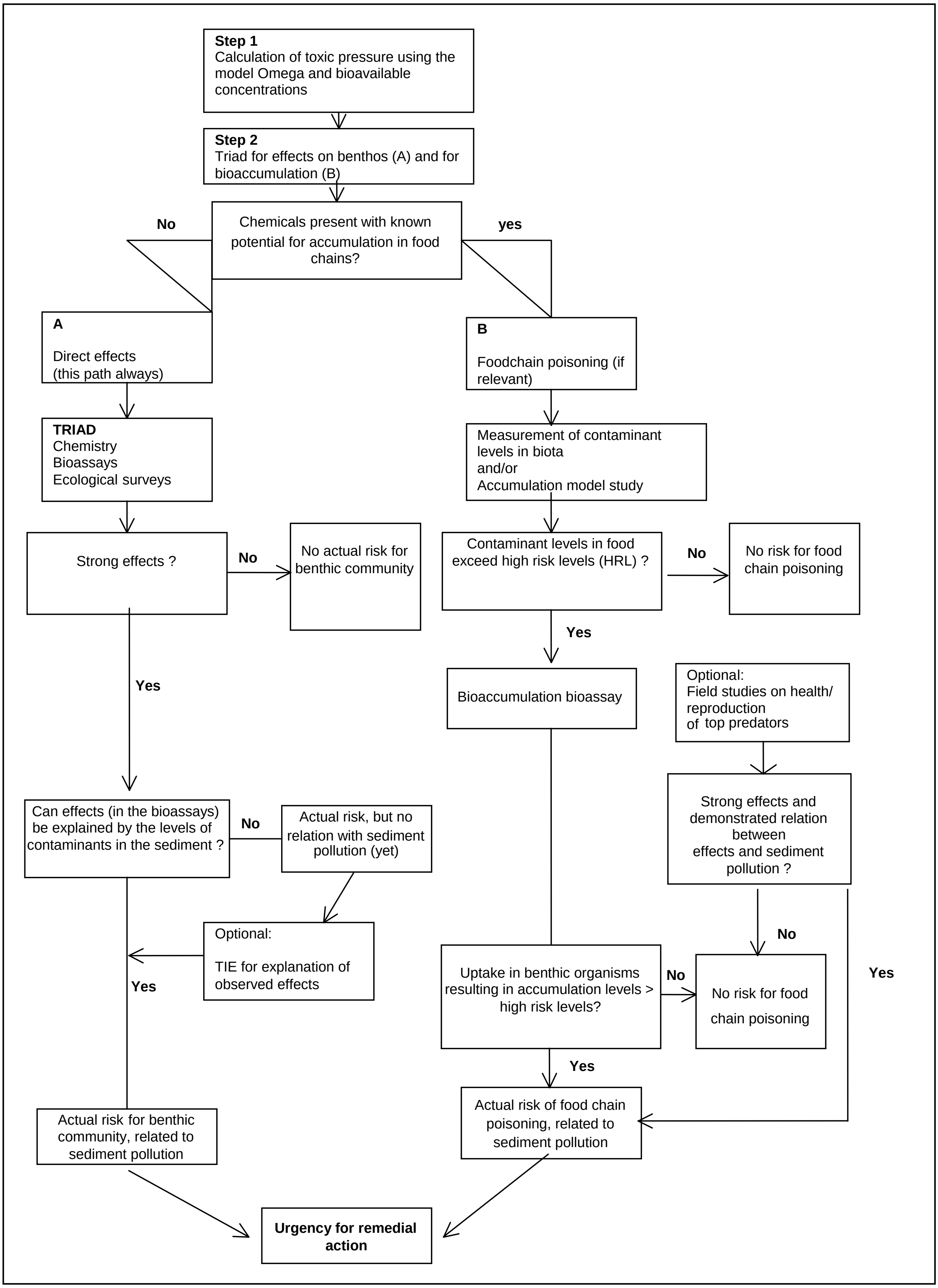


Figure 7: Ecological risk assessment in NL for the decision yes/no urgency for remedial action (van Elswijk et al., 2001).

In the Netherlands some experience exists with the use of multi criteria analysis (MCA; also called Analytic Hierarchy Process (Saaty, 1980) for this purpose. MCA enables a ranking of sites based on risks for the ecosystem. This method (described by Den Besten et al., 1995) is based on the same classification of results as described above. For each criterion (= parameter), standard numerical values (scores) were assigned to the effect/risk classes, from the value 1 for the class representing the strongest effect or highest risk, to for example 0.5 and 0.25 for the classes representing moderate risk and no risk, respectively. Then the criteria are given a specific place and weight in a hierarchy. The scores are multiplied by the weight of the corresponding criterion and subsequently totalized bottom-up using a computer program, resulting in a final score between 0 and 1. The difference between the final score and the theoretical score 1 (the score for a site with strong effects / high risk for all parameters) gives an indication of the risks for ecosystem health at each of the sites. For this method all available information from the field surveys can be used, including site-specific information from bioaccumulation studies.

At a higher level of hierarchy, information from human risk studies, ecological risk assessment, and estimates of contaminant mobility (transport) can be integrated. In the MCA, specific weights can be attributed to the different criteria (=parameters) and higher in the hierarchy, at branch points. This makes the method useful for decision makers, who have to deal with all these aspects at the same time and therefore need integrated information. In the near future, estimates of the expected beneficial effects of remedial action will also be integrated in the step of prioritization of dredging locations.

6. Newly emerging pollutants causing specific ecotoxicological effects

6.1 Developments in Germany: in vitro and in vivo techniques

6.1.1 Genotoxicity and mutagenicity

Besides acute toxicity, specific endpoints displaying chronic and sublethal potentials such as genotoxicity, mutagenicity and immuno-modulation are of increasing interest. These sublethal effects are mainly caused by priority pollutants, by integral parts of mass convenience goods that generate a new kind of diffuse pollution source by their widely daily use, and last but not least by a significant fraction of analytically not yet specified chemicals.
Toxicants related to the above mentioned endpoints normally react in a specific manner with their biological targets. As it is the case in classical ecotoxicological bioassays, specifically acting potentially hazardous chemicals are defined by their intrinsic properties like persistence, potential for bioaccumulation and toxicity which characterise them as particularly relevant. Toxicity thereby is the capacity of a substance to cause adverse effects like (i) reduction of life span, growth and reproduction, (ii) mutagenicity, carcinogenicity, teratogenicity immunotoxicity, and (iii) adverse effects as result of endocrine disruption.
This definition demonstrates the need for application of both classical ecotoxicological methods as well as newly developed methods for the assessment of specific toxicity. Although specific endpoints are clearly distinguishable from each other, the biological consequences for organisms or even populations can culminate in related dysfunctions. That means genotoxicity and mutagenicity may result in primary DNA-damage, reduction of biological fitness, hereditary diseases and even cancer. Immunotoxic reagents may lead to diseases as result of insufficient immune defence, but also to cancer as consequence of the reduced ability to eliminate pre-cancerous cells. Finally, endocrine disruptors possibly interfere with the regulation of endocrinologically dependent genes which (besides the well known possible interferences with individual development, reproductive function, sex ratio, imposex) also may have impacts on the development of malignancies.

As surface waters are widely used for the preparation of drinking water, appropriate test systems are required for the monitoring of possible genotoxic contaminations. In the course of a collaborative BMBF funded project (1997-1999) several methods (Ames test, *umu* test, alkaline elution, DNA unwinding assay, Comet assay and unscheduled DNA-synthesis test) were examined for their ability to measure genotoxicity particularly in natural surface waters. Appropriate test versions were developed (sensitized Ames test, luminometric *umu* test, alkaline elution using clams, comet assay with fish cells or aquatic plants), adapted to the test subject and validated regarding their sensitivity towards standard genotoxins. In addition to the bacterial test systems the Comet assay played an important part in the project. The assay was validated with different fish cell lines from rainbow trout (*Oncorhyhnchus mykiss*), the fibroblast-like RTG-2 cell line established from gonad tissue, and the epitheloid RTL-W1 cell line established from liver tissue. Both natural and concentrated samples of German rivers and a drinking water resource were tested. Among the non-concentrated samples several statistically positive test results were obtained both for the rivers Elbe and Rhine especially with the Ames test and the Comet assay. The river Wupper showed significant genotoxicity in the bacterial *umu* test. No genotoxicity could be found in the drinking water resource even after concentration. The findings demonstrated that the comet assay with fish cell lines is suitable as in vitro screening assay in environmental genotoxicity testing, but the choice of test cell line may be critical. As a conclusion of the study a graduated testing battery has been

proposed consisting of a bacterial (*umu* or Ames test) and an eucaryotic test like the Comet assay followed by an additional eucaryotic test (UDS test or micronucleus test) in a decisive function (Erbes et al., 1997; Schmid et al., 1997; Strmac and Braunbeck, 2000; Schnurstein and Braunbeck, 2001; Nehls and Segner, 2001).
Two bacterial short term test systems for the detection of genotoxic and mutagenic potentials in the freshwater field have been standardised according to DIN and ISO. The *umu*-test (Reifferscheid et al., 1991a,b; 1996; Schmid et al, 1997) which is based on primary DNA-damage dependent induction of the DNA-repair gene *umuC* in bacteria (DEV T3-DIN 38415-3 (1996); ISO 13829). In this system genotoxicity is determined in a liquid culture approach by colorimetric measurement of the activity of a galactosidase gene fusion product.
The second internationally standardised test system is the *Salmonella*/mammalian-microsome mutagenicity test, better known as Ames test (Ames et al., 1975; DEV T4-DIN 38415-4 (1999); ISO/DIS 16240). In its conventional form as an agar plate incorporation assay it is the most used mutagenicity test system world-wide with a background data base of several thousands of chemicals. Albeit the Ames test is very sensitive and detects a broad range of genotoxic substances, several disadvantages should be considered: As the standardised approach uses agar plates, exact concentrations of the test material cannot be specified which makes intercomparisons of test systems difficult. Furthermore high amounts of sample and consumables supplies are necessary which leads to increased costs. A very promising alternative of the standard method is an Ames fluctuation approach which is based on the DIN- and ISO-methods regarding the applied tester strains TA98 (for frameshift mutations) and TA100 (for base substitutions) and test conditions in respect of S9-incubation. A mixture of S9 liver homogenate and cofactors is essential for the detection of genotoxins which are metabolically activated to their DNA-reactive form (Reifferscheid and v. Oepen, 2002; Reifferscheid et al. 2005b). Gee et al. (1994) developed a set of base specific strains which cover all six possible base substitutions as complementation of the TA100 strain. Extensive studies carried out at the University of Mainz, Germany, and the BfG show that this system is especially applicable to waste water, sediment pore water and sediment extracts and detects a broader range of sediment associated genotoxins compared to the *umu*-test (Reifferscheid et al. 2005b). Nevertheless the *umu*-test has been incorporated into the German federal water regulation (appendix 22), but only for the testing of newly-constructed waste water plants.
Because of a multitude of scientific reasons a standard test battery for tier I genotoxicity assessment should consist of both a bacterial and a eucaryotic approach.

Though standardisation attempts on the procaryotic level have been successful, the eukaryotic level is still under discussion. Promising test systems are the Comet assay and the Micronucleus test. The Comet assay measures DNA integrity by the detection of single strand DNA breaks (alkaline unwinding method) in individual cells (McKelvey-Martin et al., 1993). The application of the comet assay in field-collected mussels (*C. fluminea*) creates new possibilities for risk assessment studies (Waldmann et al., 1995). Recently, a protocol was developed for generating a suspension of single cells from sediment exposed zebra fish embryos suitable for detecting particle-bound genotoxicity (Kosmehl et al., 2006a). The authors could demonstrate that this newly developed sediment contact assay is suitable to detect the bioavailable fraction of the total hazard potential of sediments and dredged materials. This test version can be regarded as an ecotoxicologically relevant improvement of the widely-used cell culture approaches.

At present, particularly for the Comet assay a multitude of laboratory protocols using different *in vitro* cell systems or even *in vivo* approaches exists. The selection the most adequate method for environmental purposes and standardisation efforts is still a subject of discussion. Besides that, the discussion concerning the validation of primary (and repairable) DNA-

damages as measured with the Comet assay is not finished yet. This led to the decision to use the micronucleus test for first standardisation measures of a eukaryotic test system. The measurement of micronuclei is an important parameter for the detection of cytogenetic damage. The *in vitro* micronucleus test detects non-repaired and thus manifested genetic damage. Consequently, this test can be regarded as the more significant test system, as compared to the Comet assay. A more frequent occurrence of micronuclei in treated cells suggests a risk of severe genetic damage for subsequent cell generations. In the interest of a precautionary environmental protection and health protection no samples should show any significant induction of micronuclei in the treated cell populations.
Draft standards for in vitro and in vivo testing had been submitted by German (*in vitro*) and French *(in vivo)* delegations for international votings. To meet the standardisation requirements for the method, encoded wastewater samples, some of them spiked with known genotoxins, were tested in a collaborative study organized by the German Federal Institute of Hydrology (*BfG*) (Reifferscheid et al., 2007). The study demonstrated practicability of the *in vitro* micronucleus test for (waste)water testing and provided validity data. By December 2006 both micronucleus test versions have been published as international standards (ISO 21427-1 and 21427-2)

6.1.2 Endocrine effects

There is still a considerable lack of information about concentrations and quality of hormonal active substances in environmental matrices like (waste) water, suspended particulate matter, sediments and dredged material. The existence of estrogenic, anti-estrogenic, androgenic and anti-androgenic mechanisms for endocrine disruption demonstrates the difficulty to establish one general test system to cover the broad range of suspected adverse substances. Nevertheless a working group under the German Institute for Standardisation (DIN) currently elicits possible *in vitro* test systems for standardisation.
Test systems on different biological levels have been developed in the last decade. A very basic mechanism of endocrine disruption is represented by the Enzyme-Linked Receptor Assay (ELRA) (Seifert et al., 1998; Seifert, 2004). This assay employs the same principles as competitive immunoassays based on ligand-protein interactions. However, receptor binding implicates a biological effect, either agonistic or antagonistic. For the ELRA the human estrogen receptor alpha was used for binding estrogenic substances in environmental samples. The test can be used for cost-effective and high throughput screening of estrogens and xenoestrogens in environmental samples.
A yeast-based estrogen screen test is currently in discussion with regard to possible standardisation. Genetically modified *Saccharomyces cerevisiae* cells are able to express estrogen and androgen receptor proteins whereby receptor mediated endocrine disruption is colorimetrically measured by induction of a reporter gene (*lacZ*) (McDonnell et al., 1991; Breithofer et al., 1998; Jungbauer et al., 2002; Reifferscheid et al., 2005a). The yeast cells bear an expression plasmid for the human estrogen receptor alpha (and the androgen receptor resp.) and a reporter plasmid containing the *lacZ* gene under the control of the vitellogenin hormone response element. In 2005 the first inter-laboratory round-robin test with a recombinant yeast assay (RYA) has been successfully performed under German contribution as part of a project titled "Screening Methods for Water Data Information in Support of the Implementation of the Water Framework Directive (SWIFT-WFD)" supported by the European Union. The study resulted in a very low frequency of false positives and detected a number of false negatives, depending on the limit of detection (LOD). As it revealed larger variations than expected, along with important pitfalls it should be considered how to improve the robustness, repeatability and transferability of the assay. A very promising and impressively incomplex *in vitro* system is also the so-called E-screen assay which uses the

MCF-7 cell line (Soto et al., 1995; Payne et al., 2000). The test is based on the proliferation of human estrogen-receptor-positive breast cancer cells and can be performed in microplates. At a first glance the relatively long incubation time of about 6 days can be regarded as a disadvantage. This is compensated by the fact that the system obviously is the most sensitive *in vitro* tool available so far and alarm scenarios concerning endocrine disruptors requiring immediate reaction are rather improbable. The test has been successfully introduced not only for testing of pure chemicals but also for sediment pore waters and elutriates (Reifferscheid et al., 2005a).

Among considerable activities concerning endocrine disruptors on the *in vivo* side, several sediment relevant findings demonstrate the need for further follow up of this matter. Oetken et al. (2004) showed that the issue of endocrine disruption (ED) in invertebrates - which represent 95% of all known species in the animal kingdom - has generated rather little interest in the past compared to research with aquatic vertebrates in this field. As invertebrates account for an important part of the global biodiversity, key species for the structure and function of aquatic (and terrestrial) ecosystems could reasonably be used to monitor endocrine disruption. The principal susceptibility of invertebrates to endocrine-active compounds has been demonstrated with case studies of tributyltin effects in molluscs (Bauer et al., 1997; Schulte-Oehlmann, et al., 2000) and of insect growth regulators (Oetken et al., 2004) , the latter designed for endocrine disruption. The effects of three suspected endocrine disrupting chemicals, the xeno-estrogens bisphenol A (BPA), 4-tert-octylphenol (OP) and 4-n-nonylphenol (NP), were investigated by Duft et al. (2003b) in a whole-sediment biotest with the freshwater mudsnail *Potamopyrgus antipodarum* (Gastropoda, Prosobranchia). The results indicated that *P. antipodarum* is highly sensitive to the tested endocrine disruptors at environmentally relevant concentrations.

6.1.3 Immunotoxicity

The third work package on specific toxic effects deals with immunotoxicity. With respect to available standardized methods this issue is least processed. For a short time a DIN-working group is engaged in this field. Similar to the endocrine disruptor problem, immuntotoxicity can be modulated by a variety of mechanisms in consequence of the complexity of the immune system. Further on exempt from fish species only limited information exists about the immuno-competence of species affected by sediment quality and dredging processes.

As a matter of fact a basic immunological mechanism is phagocytosis of body-foreign particles by certain blood cells. In aquatic animals, cell mediated, unspecific immune defence plays a major role compared to mammalia. Hence, a phagocytosis assay could be used as functional biomarker for immunotoxic influences of sediment bound substances. Phagocytosis theoretically could link the *in vitro*/*in vivo* relationship by using hematocytes of mussels. In this context, Blaise and colleagues developed a rapid, cost-effective, and miniaturized immunocompetence assay to evaluate phagocytic activity, viability, and concentration of haemocytes in freshwater and marine bivalves (Blaise et al., 2002a). They already used this system as a part of a comprehensive battery of biomarkers to assess possible effects of anthropogenic contaminant input on soft-shell clam (*Mya arenaria*) populations (Blaise et al., 2002b).

Besides all proceedings concerning the detection of risks for specific toxicological impacts on the *in vitro* level, future approaches should take into account not only tier I aspects for screening but also tier II attempts which include whole organismic endpoints. Thereby concepts in consonance with national animal welfare acts should be discussed.

6.2 Developments in The Netherlands: improving bioassays, application of biomarkers, effect-directed analysis (EDA), toxicity identification evaluation (TIE), cell lines and genomics

6.2.1 In-situ bioassays

In-situ bioassays have over the past decade gained increased attention and acceptance as ways to improve the ability to link cause and effect in aquatic ecotoxicological studies. One of the main advantages provided by *in-situ* tests compared to more conventional approaches is better control over "stressor" exposure to a defined population of test animals under natural or near-natural field conditions. By partly controlling what environmental compartment(s) a known or standardized number of test animals are in contact with, the researcher can have an improved ability to describe and link cause and effect. In short, when conducted properly, *in situ* tests can provide improved diagnostic ability and high ecological relevance. *In-situ* approaches also allow for some level of "control" and replication within natural systems.

In-situ tests with caged organisms can also serve an important function in the "Problem Formulation" phase of larger ecological risk assessments where there is a need to better identify which environmental variables or exposure pathways require further assessment or consideration (Box 6).

Box 6: The *in-situ* bioassays with midge larvae.

The *in-situ* bioassays with midge larvae, *Chironomus riparius*.

The survival, rate development and increase in biomass of *Chironomus* larvae was compared between sediment bioassays performed in the laboratory and bioassays in field cages. The incidence of mentum deformities was compared not only between laboratory and field bioassays, but also with observations on field populations of *Chironomus* larvae. Survival in the field bioassays was slightly higher than in the laboratory bioassays, except at locations with known contamination of the surface water. The influence of surface water quality in field bioassays was demonstrated in translocation experiments, in which clean sediment was placed in a polluted site, and vice versa. In a field bioassay carried out in the autumn, an inverse relation between the rate of development and the initial larval density in the field cages was demonstrated. In addition, in field bioassays with *C. riparius* considerable seasonal variation in the survival and incidence of mentum deformities was found. Field bioassays performed during the winter season indicate that low temperatures can interact with or add to the effects of sediment contamination on chironomid populations (Den Besten et al., 2003b).

6.2.2 Improving bioassay techniques by using optimized approaches for exposure tests and sediment assays

The use of biological sensors, rather than chemical measurements, appears promising for risk assessments of contaminated environments provided that the assay is able to mimic natural conditions. As an alternative to the standard bioassay protocol, a new technique was developed that meets this requirement, leaving sample and geochemical conditions in tact. Exposure tests were conducted with two aquatic species that occur in sediment and water, respectively. Comparison between the two methods showed that the standard protocol tends to overestimate risks for PAHs, and underestimates the risks for heavy metals, in terms of accumulated amounts. Sample handling largely affected chemical speciation, and exposure concentrations deviated from the ones observed in the undisturbed setting. This new approach may contribute to better-founded quality criteria for sediments.

6.2.3 Effect-directed analysis (EDA) and toxicity identification (TIE)

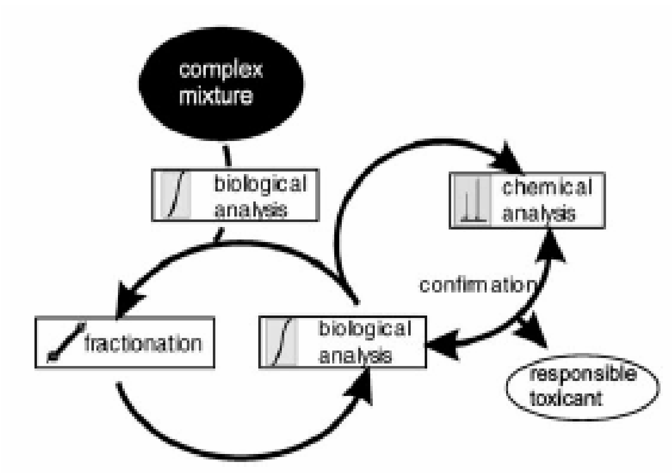
Biological testing does not provide information on the compounds causing the measured effect. A severe increase of toxicity measured with bio-analyses should initiate an investigation to identify the causes of toxicity. This information is crucial for effective emission control of the measured toxicity. Therefore a tool is needed for successful identification of the toxic compounds causing the measured effect. Successful identification of toxicants may provide a first step to include this compound on the list of other relevant compounds.

Effect-Directed Analysis (EDA) is a promising tool for the identification of organic toxicants in complex mixtures. EDA aims at the identification of chemical causes of toxic effects (see intermezzo: Effect-directed Analyses [EDA]).
The toxicological endpoint used for EDA is crucial and determines not only the toxicants that can be identified, but also the chances for success. Assays that are able to detect toxic responses on (sub)cellular level are easy to use and react in many cases on specific toxic compounds. With these assays, the chances for success in EDA studies are high. In the near future more rapid screening tools with specific end-points will be available. This makes EDA more successful and decreases the cost (Box 7).

Box 7: Scheme of effect-directed analysis of complex mixtures (figure W. Brack, 2003)

Effect-directed Analyses (EDA)

EDA is a tool for the identification of organic toxicants in complex mixtures. Chemical causes of toxic effects could be identified. This is done in an EDA procedure called bioassay-directed chemical fractionation. First, complex mixtures are tested for toxicity and subjected to fractionation procedures. After each fractionation step, all fractions are examined for toxicity. The toxic fractions are subjected to chemical identification and quantification. Because of the decreased complexity of the environmental sample mixture, chances of positive identification of toxicants are increased. The method requires a final step aimed at the confirmation of identified toxicants as the cause of the measured effects. This conformation is a crucial, however often overlooked step for the establishment of reliable cause-effect relationships. The figure shows a scheme of the EDA method.



6.2.4 Practical experience with effect-directed analysis (EDA) on a surface water extract

The EDA method described above has been used to explain the toxicity of a XAD-extract from the river Meuse at Eijsden. The bio-analysis used to give a measure of the toxicity was the algae growth test. The toxic extract was fractionated and toxicity was measured in every fraction. Three of the thirty fractions appeared to be toxic. Chemical analysis was used to identify the compounds present in the toxic fractions. The herbicides, alachlor, metholachlor and terbutryn were identified as suspicious toxicants.

Suspended matter extract: ER-Calux

Estrogenic activity was tested in three extracts of suspended particulate matter. All extracts showed a significant estrogenic activity, compared to a blank extract. Extracts were split in fractions, based upon aqueous solubility of the compounds present. The highest activity (up to 95%) was found in the fraction that in other studies contained human estrogens (chemical analysis of these compounds was not feasible in this project). Only 5% of the total activity was retained in the fraction of compounds with highest aqueous solubility.

Suspended matter extract: DR-Calux
Dioxin-type toxicity was tested in two extracts of suspended matter (Eijsden and Schaar van Ouden Doel) and an extract of sediment (Veerse Meer), both in the total extract and in sub-fractions. Fractions were prepared such that different compound groups ended up in different fractions. PAHs, heterocyclic PAHs (e.g., PAHs with sulfur or oxygen in the aromatic ring) and brominated flame retardants were present in the most active fractions. WFD priority compounds were responsible for less than 10 % of the total activity. While all sample extracts gave a response in the test, the summed activities of the individual fractions were always higher than the activity of the total sample. This indicates that compounds are present with an antagonistic activity in the DR-CALUX test.

6.2.5 Toxicity Identification and Evaluation research in the CTT

For the disposal of dredged material in the North Sea, an exemption is required under the Dutch Pollution of Marine Waters Act, and for the other coastal waters a permit is needed under the Dutch Pollution of Surface Waters Act. Recent ecotoxicological research has shown that, alongside the classic chemicals of the 1970s and 1980s, it is necessary to focus on more specific pollutants in estuarine dredged materials. In particular, it has been shown that harbour mud contains 'new' pollutants, such as tributyltin from marine paint, substances with a dioxin-like mode of action that affect hormones and have directly harmful effects on marine organisms.

Not only environmental risks played a role in the final setup of the CTT. It also took into account the consequences for the volumes of dredged material that could be dispersed in saline waters. A significant break in trend in these volumes was not considered acceptable from a societal and political perspective. Not only because of the additional costs for sea port managers, but also because the depot capacity is a rare commodity, that can best be used for storing far more critical contaminated freshwater dredged material than for the generally cleaner saline variety. To ensure that the various bioassays give effective warning of problems, signal values have been included for them in the assessment systems. Any exceedance of these signal values means that the permit or exemption holder must investigate the cause of the exceedance, generally in consultation with the competent authority at central or local government level. But which substances were actually responsible for the effect? If this is known, then specific measures can be taken to reduce the source or to clean the material. Activities must perform specific test the CTT, are aimed at disclosure of the causes of found excess responses of harbour sludge samples in bio-assays. If chemical substances are thought to be involved, the identity of these substances is determined in so-called Toxicity Identification and Evaluation research, TIE.

6.2.6 Biomarkers

An important, ongoing advancement in ecotoxicology is the shift from broad-spectrum bioassays to receptor based assays with high specificity (see Van der Oost et al., 2005). This will result in the development of diagnostic approaches where toxicity is only one of the stressors present in the field. Biomarkers are useful tools in this respect. There are different concepts for the use of biomarkers (Depledge and Fossi, 1994; Den Besten, 1998)**:**

- Biomarkers in combination with bioassays as parameters in water or sediment quality monitoring (trend analysis)
- Biomarkers that lead the investigations from screening to detailed assessment (tiered approaches or weight of evidence approaches)
- Biomarkers linked with chemical analysis (hyphenated approaches/ toxicity identification evaluation [TIE])
- Biomarkers as diagnostic tools.

For many environmental quality assessments, bioassays and biomarkers can be used together. The main reason for having a battery of bioassays and biomarkers enables coverage of a broad spectrum of chemicals or make a better representation of the species present in the field. On the other hand, concepts can be chosen in which biomarkers clearly give additional information. For example, bioassays are selected for their ability to detect adverse toxic effects on ecosystem components whereas biomarkers are included as measures of health and fitness of selected species (from bioassays or from the field). Biomarkers often provide an avenue to study combination effects and enable in-depth analysis of toxic mechanisms on molecular and cellular levels, thus allowing insight in causal and adaptive responses.
In some cases, biomarkers are integrated in bioassays, as is the case for the bioluminescent bacterium *Vibrio fischeri* (bioluminescence is the biomarker for energy metabolism). Standard bioassays are widely used because they are designed to fulfill regulatory purposes in a reliable way. Practical demand comes to the fore compared to scientific demand. However, the European Framework Directive may require 'good ecological quality' far beyond established trigger values which call or increased scientific demand. Therefore, more sensitive and more specific approaches have to be used. Biomarker responses integrate toxicokinetics and toxic interactions if exposed to mixtures. The rapid responses provided by biomarkers allow an early warning system of longer-term effects. Biomarker approaches also overcome the problem of extrapolation of *in vitro* measurements to *in vivo* responses by the potential application in laboratory tests as well as in field monitoring. *In vitro* tests provide insights in toxicological mechanisms, a thorough balance of protection and susceptibility factors, comparisons of organ and species sensitivity, and links to chemical analysis and causative agents. On the other hand, biomarker measurements in the field integrate exposure of different routes over time and ideally over a range of species.

6.2.7 Cell lines in bioassays

There is growing criticism on the use of animals in toxicity tests. This has lead to an enhanced attention for alternative test systems, with plants or bacteria as test species, or with immortalized cell lines derived from animals. With regard to the detection of chemicals, these in vitro bioassays may either be "*broad spectrum*" or so called "*toxic mechanism based*" assays. Examples of the first type of bioassays are the Microtox assay and biochemical endpoints in fish cell lines. Examples of toxic mechanism based assays are the Mutatox and Umu-C (genotoxicity), DR-CALUX (dioxin-like or Ah-mediated toxicity), ER-CALUX (estrogenic toxicity) and AR-CALUX (androgenic toxicity). In The Netherlands and elsewhere, cytoxicity measurements with fish cell lines have been studied for their value as a screening parameters for the detection of effluent toxicity (Babich & Borenfreund, 1991; Gagné & Ahna, 1997; Tuk & Den Besten, 2001). Apart from broad spectrum endpoints such as crystal violett, MTT and neutral red uptake, also markers for genotoxicity have been used in fish cell lines (Hollert et al., 2000a; Nehls & Segner, 2001). In The Netherlands, the in vitro bioassay with the fish cell line RTG-2 has been used in the BECPELAG project (Den Besten et al., 2006).

6.2.8 Toxicogenomics

In 2006 the Dutch Ministry of Transport, Public Works and Water Management has started an initiative to explore the possibilities of integrating "state-of-the-art" genomics, proteomics and metabolomics techniques with methods currently used for environmental risk management purposes. The basic idea behind this is that in time the developments in this field of research could lead to highly specific, highly sensitive instruments for the detection of chemical risks in various environmental compartments, including sediments. In combination with advanced miniaturization (lab-on-a-chip) and automation they may evolve into potentially very fast and cheap instruments that, when effectively interlinked with effect directed analysis (EDA) and Toxicity Identification and Evaluation (TIE) could facilitate a future transition to a true effect based risk assessment and management. Currently a research programme is being started that includes applications of toxicogenomics for surface water and sediment quality assessment.

7. Situation in the United Kingdom, France and Belgium

Information about the use of ecotoxicological methods for sediment and dredged material quality assessment in other European countries was described earlier within the framework of SedNet (Den Besten et al., 2003a; Den Besten, 2007). An update of this information is given below.

7.1 Assessment of dredged material for relocation

7.1.1 The United Kingdom

Sea disposal

In England and Wales, sea disposal is regulated nationally by the **Department of Environment, Food and Rural Affairs (Defra)**, but many of the decisions are driven by policy decisions made within OSPAR. Defra controls these activities relating to sea disposal through a system of licences under the **Food and Environment Protection Act (FEPA)** 1985. This Act provides a licensing system for the deposit of substances and articles from vehicles and vessels, etc. in tidal waters below the level of mean high-water springs.

Sea disposal licences are only issued after detailed scientific assessment [with the support of the **Centre for Environment, Fisheries and Aquaculture Science (CEFAS)** who advise Defra of the potential environmental impact, with particular regard to the need to safeguard marine conservation sites, fisheries and other uses of the sea. Prior to this year, the assessment procedure focused on (1) review of sediment data (physical quality and chemical quality relative to action levels) from the area proposed for dredging and (2) information about the sea disposal site and its ability to assimilate the materials proposed for disposal. As of this year, bioassay data are being collected in parallel to sediment chemistry data (i.e. *in situ* sediment quality assessment). In addition, CEFAS are testing a new dredged material disposal assessment decision tree (Murray, pers. comm.), which is both rule- and risk-based, providing a tiered assessment procedure that considers not only environmental risks but also beneficial uses for dredged materials proposed for disposal.

In summary, to assess the potential effects of contaminants, firstly the physical properties of the sediment are assessed. Secondly, the sediment chemistry of materials proposed for disposal at sea are assessed using action levels (applied by CEFAS) to give an indication of the potential for impacts. A standard suite of chemicals is used in the first instance and augmented as needed for site-specific conditions. CEFAS has an assessment procedure that involves two action levels **(Action Level 1 and Action Level 2)**. Below Action Level 1 the material is usually suitable chemically for beneficial use or for sea disposal, while below and above Action Level 2, further assessment will be required before a licence for either sea disposal or beneficial use is issued. Action level figures are not pass or fail criteria, however, as the approach used by CEFAS is one of weight of evidence. Using the physical, chemical and bioassay data in parallel to make decisions about the suitability of dredged materials for sea disposal will permit CEFAS to collect enough data to evaluate this new approach, and the decision tree will be modified in light of CEFAS's findings.

A recent report for the Environment Agency for England and Wales, (Apitz et al., 2004) reviewed the ways in which SQVs have been implemented within sediment assessment and management frameworks (including DM disposal options) in other countries so that an implementation strategy appropriate to England and Wales can be developed for the Environment Agency. It was recommended that the Agency should begin to roll out use of SQVs within separate, tiered risk-assessment frameworks for assessing ecological quality and dredging activities.

Disposal on land (spreading and in landfills)

Maintenance dredging in inland waterways is subject to limited environmental legislation, as reviewed in Bates and Hooper (1997). Capital dredging is subject to the same controls as maintenance dredging, but in some cases requires a full environmental assessment (which has the scope to include ecological risk assessment, but does not usually do so).
Under the **Waste Management Licensing Regulations (WMLR)** 1994, all disposal of dredged material not qualifying for an exemption must be licensed. Management of sediment through spreading on land under exemptions is regulated by DEFRA, through the WMLR. Generally, only site history and sediment chemistry data are used, but this is under review. For dredged materials that are very heavily contaminated, the Special Waste Regulations (1996) might come into play, and these are chemically driven assessment procedures.

7.1.2 France

The political framework for dredged materials is still under discussion for freshwaters. From a legal point of view, these materials are classified as wastes, but not necessarily as hazardous. Another confusing issue is the destination of the materials – a deposit on soils is subject to a different set of regulations than those applying to a deposit in waters. There is thus a need for guidance at various levels of the management process; some aspects of guidance, e.g. for the overall management process, have been introduced (Imbert et al., 1998) and should now be completed by more specific frameworks for the evaluation of the dredged materials. On behalf of the Ministry of Equipment and Transportation, such a framework was recently proposed for the ecological side of the assessment (Babut & Perrodin, 2001).

Disposal on soil

If the deposit is located close to a river or a canal, contaminant transfers to the surface water may occur, or to the surrounding soils, and to the groundwater. Organisms of concern include plants and aquatic species. The following assessment endpoints have thus been proposed:
The deposit should not disrupt the germination or growth of plants, in particular those of agricultural value; Run-off waters should not affect aquatic species;
Finally, it should not degrade the groundwater quality, i.e. for drinking water purposes.

In this scenario, stressors are represented by two types of water samples: excess water (mixture of overlying and pore water) collected on the deposit, which will support the transfers to the surrounding soils or the surface waters, and water obtained from elutriate assays in unsaturated packed columns. The tested assumptions are assessed with bioassays on bacteria (Metplate®), unicellular algae, a pelagic crustacean (*Ceriodaphnia dubia*), amphibians, and vegetables (lettuce, maize, etc.). The soil macrofauna and microflora have not yet been considered, but should be in the future versions of this protocol.

Disposal in water

In case a disposal site is constructed as a cross-section of the alluvial groundwater, the water will flow through the dredged material deposit and contaminants may be eluted over time. Aquatic species can also be affected at the time of deposition, by direct exposure to pollutants dissolved in sediment pore water. Benthic species may be affected in various ways, in particular when they colonize the deposit. The following criteria for allowing disposal in water have thus been proposed:
The deposit should have no effect on the structure and abundance of benthic invertebrates in the location;
It should have no long-term effect on pelagic species;
It should not cause groundwater pollution, as such disposal sites are in fact cross-sections of shallow alluvial groundwater.
A fourth assessment endpoint should be introduced, regarding health risks for recreational uses, including fishing, but this endpoint was not implemented in the current version of the approach. The analysis phase includes aquatic bioassays (bacteria-Metplate^{TM}, algae, microcrustaceans *Ceriodaphnia dubia*, rotifers *Brachionus calyciflorus*), and leaching assays in columns under ascendant flow.

7.2 Assessment of the in situ sediment quality for remediation

7.2.1 The United Kingdom

There has been considerable research and development in the field of sediment risk assessment in the UK, but not much uptake in a regulatory sense, particularly for freshwater sediments. The UK had a very active period of sediment research in the mid-1990s, resulting in broad reviews of approaches to risk assessment (e.g., NRA, 1995; SNIFFER, 1995) as well as establishment of the National Marine Monitoring Programme (MPMMG, 1998).

In the UK, *in situ* freshwater sediments are only routinely assessed for environmental quality within the framework of the EC Dangerous Substances Directive (76/464/EEC), together with the Water Resources Act 1991, both of which require control over inputs of dangerous substances into water. Specifically for sediments, List 1 Dangerous Substances are monitored in sediment or biota at sites proximate to dischargers that discharge those substances, under a standstill provision, which states that it is necessary to demonstrate that the levels of a particular substance, present in either sediments and/or biota, do not increase significantly with time (Environment Agency, 1997). In addition, *ad hoc* investigations of freshwater sediment quality are conducted, related to site-specific issues such as contaminated land, navigable waterways management, water quality problems and academic interest. Aside from draft sediment quality standards for dioxins and furans in England and Wales (Environment Agency, 2000), there are no freshwater standards for sediment assessment at this point in any part of the UK. In the past few years, the Environment Agency has been reviewing their policies related to sediment assessment and management. They commissioned two reviews to advise them. The Environment Agency (2002a) reviewed the derivation of sediment quality guidelines and the Environment Agency (2002b) reviewed the nature and extent of sediment issues, including in situ sediment risk assessment. A more recent report, (Apitz et al., 2004) reviewed the ways in which SQVs have been implemented within sediment assessment and management frameworks in other countries so that an implementation strategy appropriate to England and Wales can be developed for the Environment Agency.

The authors of a recent document by the Environment Agency (2002a) stress that SQGs are insufficiently reliable to support automatic regulatory action should guideline concentrations in sediments be exceeded. Exceedance of SQGs should always trigger investigatory actions that seek to confirm or deny the predicted risk. It was argued that SQGs could be useful in the UK, provided (i) they minimize false negatives (type II error) and (ii) their exceedance must not be the sole reason for regulatory action (Environment Agency, 2002a). They should rather be used as a first screening, along with considerations of background concentrations, which is consistent with the conclusions drawn in another document by the Environment Agency (2002b).

Although the Environment Agency (2002a, b) supports tiered approaches to sediment assessment, they do not propose a definite framework at the moment, but recommend developing and validating an approach for the UK. It was recommended later (Apitz et al., 2004) that the Agency should begin to roll out use of SQVs within separate, tiered risk assessment frameworks for assessing ecological quality and dredging activities. They further recommended that other lines of evidence such as chemical speciation, bioavailability and bioassay studies, plus ecological surveys, should also be used in these frameworks if they can add value to management decisions.

In marine and estuarine waters, the UK's National Marine Monitoring Programme (NMMP) is a well-developed programme that monitors sediment quality, essentially using a triad approach. The Marine Pollution Monitoring Management Group (MPMMG) is a management group for the programme, with representation from all government organizations with statutory marine environmental protection monitoring obligations.

The NMMP was developed in response to OSPAR as well as several EC Directives. The NMMP Phase 2 focuses on stable depositional sediment sites (approximately 110 sites) and evaluates: sediment chemistry, benthic communities, bioaccumulation, and ecological effects methods. It is also anticipated that NMMP data will be used to fulfil some of the monitoring requirements of the Water Framework Directive[2]. Initially, the main objective of the programme was to describe marine quality around the UK through spatial surveys (phase 1), but it has now shifted to detecting with appropriate accuracy long-term trends in physical, biological and chemical variables at selected estuarine and coastal sites (phase 2). Other objectives include support for consistent standards in national and international monitoring programmes for marine environmental quality (for example, EC Directives, OSPAR) and making recommendations on how new analyses and techniques are best implemented in the United Kingdom. Overall, the aim is to produce reports providing overviews of the spatial (NMP holistic report 1998) and temporal distributions (every 3 years from 2002) of these variables and their inter-relationships.

[2] Full details of all sites together with methodologies, sampling schedules and frequencies are provided in the NMMP2 monitoring manual – 'The Green Book' at www.marlab.ac.uk/nmpr/nmp.htm.

7.2.2 France

There is currently no framework applied in France which could be definitely attributed to this kind of risk assessment. *In situ* assessments are mainly related to (i) monitoring, (ii) ecosystem restoration or (iii) flood management.

(i) *Monitoring*: Until now, sediment quality monitoring for the protection of water bodies was done on the basis of analyses of priority pollutants and comparison to numerical SQGs. The use of toxicity bioassays is now seriously envisaged, following several demonstrative studies (Garric et al., 1998). Two bioassays have been selected, i.e. *Chironomus riparius* (10 days, survival and growth) and *Hyalella azteca* (14 days, survival and growth) and will be applied after the formal adoption of a standard. Observations of invertebrate communities are also carried out at almost all the stations of the monitoring network, but their results are usually not matched with measurements of sediment contaminants. An attempt was made in 2001 to identify sensitive benthic organisms and reliable descriptive variables, and possibly underline contamination and biological response patterns (Garric et al., 2002). This first study appears interesting, but should be extended with more powerful multivariate methods, and a more selective approach, as it appears that the impacts of contaminants are stronger in low current sections[3]. Ultimately, either the use of bioassays or matching benthic observations with sediment chemistry should help to consolidate or refine the existing SQGs.

(ii) Ecosystem restoration: there may be many reasons leading a local institution (municipality or group of municipalities) or a water manager to envisage a restoration of a degraded ecosystem, or some functionalities of that ecosystem. Dredging in this case appears as a technical solution among others, or as a part of the overall restoration process. In any case, the dredging project will be subject to authorization by the relevant authority or, depending of the volume, to a simple declaration. In the latter case, the project manager will have to describe all aspects of the project, while in the former he will have to provide a so-called impact study encompassing a broad range of issues.

(iii) Flood management: dredging may be proposed as a solution for managing floods in urbanized areas, or in the vicinity of dams. Again, the dredging project will be subject to authorization by the relevant authority or, depending of the volume, to a simple declaration.

The current guidance for these impact studies is rather open if not vague, and does not require the inclusion of ecotoxicological aspects. It is recommended that various management options, not only dredging, be considered (Imbert et al., 1998). The relevant authority would generally ask for a focus on toxicological or ecological impacts if it knows beforehand, or suspects, that chemicals are present at the site of concern. A specific guidance for *in situ* sediment risk assessment is currently under development on behalf of the French Ministry of Transportation.

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[3] The standard method for invertebrate community assessment is based on sampling of various habitats; the above-mentioned approach looking for relationships between richness or abundance and sediment contamination did not discriminate between the habitats

7.2.3 Belgium (Flanders)

Sediment quality assessment in Flanders has been incorporated in a monitoring network by the Flemish Environment Agency since 2000. The focus is on freshwaters. Every year 150 locations are sampled, with specific locations resampled every four years, so in total 600 locations are included in the monitoring programme.

The assessment is based on a triad approach (Long and Chapman, 1985; Chapman, 1996). Physical-chemical, biological and ecotoxicological assessment methodologies are used, and an identical weight is assigned to each of the three assessments. The principle behind the classification of the watercourse sediments rests on an evaluation of the abnormality compared to a reference condition, so for each methodology a reference condition must be defined. This creates the possibility of classifying watercourse sediments in the absence of existing biological standards.

- **Physical-chemical assessment**
  The chemical parameters that are included in the assessment are nonpolar hydrocarbons (NPHCs), extractable organohalogens (EOX), the sum of the chlorinated pesticides (SOCP), the sum of seven PCBs (PCB7), the sum of six Borneff PAHs (PAH6), and heavy metals Cd, Cr, Cu, Ni, Pb, Hg, Zn and As. The concentrations are normalized to values for sediment with a standard granular composition and organic carbon content (see description of normalization in the section on The Netherlands). The site is classified based on the ratio to reference values. The sediments are ranked in classes based on the concentrations of the various contaminants. The sediment then receives an overall ranking based on the highest contaminant class ranking.

- **Ecotoxicological assessment**
  A battery of three tests is used for the ecotoxicological assessment. The battery consists of two pore water tests, namely a growth inhibition test with *Raphidocelis subcapitata* and an acute mortality test with *Thamnocephalus platyurus*, and one solid-phase test, namely an acute test with *Hyalella azteca*. The results are compared with results obtained with a reference sediment (with similar characteristics for grain size distribution, etc.). Based on the ratio, a classification is assigned. The ultimate ecotoxicological class is determined by the highest class of the two assessments (interstitial water and bulk sediment). The result is used as an estimate of the acute impact determined on aquatic life forms.

- **Biological assessment**
  Two indexes are used for the biological quality of watercourse sediments, namely a Biotic Sediment Index (De Pauw and Heylen, 2001) and the percentage of mouth deformities of *Chironomus* spp. (De Deckere et al., 2000).

Finally, the results are integrated based on the three classifications. This assessment method results in a rough indication of the sediment quality. To date, the results of this approach have not been used directly in sediment management. However, a method was proposed to use the information from risk assessment studies for the prioritization of remediation sites (Van der Zandt and Van Leeuwen, 1992).

8. Future perspectives and recommendations

8.1 Sediment assessment and management within the framework of WFD

Today the European Water Framework Directive (EU-WFD) provides the basis for transnational EU water legislation. Under the WFD, the focus on contaminated sediments will change considerably due to the holistic approach integrated in the framework directive and the subsequent need for an improved understanding of the ecological and ecotoxicological impact of sediments and dredged material on the aquatic environmental quality. When in a given water system the chemical and ecological objectives (defined as a good ecological status) are not met, specific measures need to be included in the river basin management plan. While the focus is on water quality and therefore on the management of (upstream) pollution sources, one of the possible measures is sediment remediation. In order to evaluate whether sediment remediation would be an effective measure for the improvement of the chemical and ecological status, risk assessment is required with special attention to the relation between sediment and water quality.

A first proposal for the implementation of sediment environmental quality standards has been suggested recently by the German Fraunhofer Institute (2002). However, it may very well be possible that sediment quality standards will not be used for compliance checking, but for diagnostic purposes, e.g. to evaluate the role of contaminated sediments in waters where WFD objectives can not be met. Most likely the use of sediment quality standards in diagnostic frameworks will only be the first step in tiered approaches that are currently under development. The focus in these frameworks is on *in situ* sediment quality assessment, although the risks of aquatic disposal of dredged material also need to be included in a river basin management plan. In Germany a first integral approach for the risk assessment of contaminated sediments and the derived suggested management options at the River catchment scale has been performed by Heise et al. (2004).

In the Netherlands, the sediment quality assessment that so far has been part of the legal framework of the soil protection act, now is restructured in order to prioritise sediment remediation locations from the 'WFD perspective'. This means that the main question of the risk assessment will shift from "Are there unacceptable risks for the ecosystem?" to "Is sediment quality the main limiting factor for reaching ecological objectives?". The risk assessment approach for *in* situ sediment quality in The Netherlands already has been changed to be more in line with WFD requirements. It is expected that sediment remediation projects will be closely linked to other measures related to WFD objectives. Ecotoxicological techniques might be used more as diagnostic tools, to distinguish toxic pressure on aquatic organisms from other stress factors like eutrophication, habitat destruction etc. Also biomarkers may become more important as indicators of wildlife health. Another opportunity is to use ecotoxicological techniques as replacement for ecological observations. For instance, instead of macrofauna surveys one could use *in situ* bioassays, in order to follow the survival and reproduction of caged organisms. Of course the species used should bear relevance to the ecosystem being studied.

But perhaps the most challenging opportunity the WFD offers is the development and implementation of screening bioassays that are used before chemical analysis is done. This could become the answer to the discussion about the thousands of chemicals present in the environment and impossible to analyse all. Effect-directed chemical characterization is expected to become a powerful and cost-effective approach when the mixture of chemicals is unknown. The development of simple and sensitive tests that allow for a high throughput capacity is a very promising research field and will certainly gain interest in the future (Den Besten & Munawar, 2005). Screening bioassays may also be applied to obtain a first indication of the risks for the ecosystem.

8.2 Recommendations

During the last decade, much effort has been made both in the Netherlands and Germany towards the development and evaluation of bioassays for the assessment of ecotoxicological effects in sediment and dredged material. In parallel with the implementation of the EU-WFD as the legislative driving force, the future developments should now be focused on

- The harmonization of bioassays with regard to practical use and environmental protection of the methods in use,
- The implementation of biotest batteries for the risk assessment of sediments and dredged material,
- The development and harmonization of whole sediment bioassays and test systems for sublethal and chronic effects,
- The development of ecotoxicological risk analysis schemes for the description of the impact and pressure of dredged material to the environmental conditions at the river catchment scale,
- The development of screening bioassays for effect-directed analysis of water quality,
- The transfer and proliferation of gained knowledge on the field of ecotoxicological assessment of sediments and dredged material within the European Union by formation of demand driven networks and co-operation in transnational field studies.

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10. Abbreviations

ARGE Elbe	Working Group for the Advancement of Water Quality of the River Elbe
BfG	Bundesanstalt für Gewässerkunde
CTT	Chemistry-Toxicity Test
DEV	German Standard Methods for Examination of Water, Wastewater and Sludge
DGE	Dutch-German Exchange on Dredged Material
DIN	German Organization for Standardization
DM	Dredged Material
EIA	Environmental Impact Assessment (UVP)
EIAA	Administrative Provision on the Implementation of the Environmental Impact Assessment Act (EIAA)
EIS	Environmental Impact Study (UVU)
EN	European Organization for Standardization
ERA	Environmental Risk Assessment (URE)
GV	Guidance value / Richtwerte (RW)
HABAB	Directive for the Management of Dredged Material in Inland Waters
HABAK	Directive for the Management of Dredged Material in Coastal Waters
HAS	Hazard Assessment Scheme
ISO	International Organization for Standardization
OECD	Organization for Economic Cooperation and Development
pT	potentia Toxicologiae
RIKZ	Rijksinstituut voor Kust en Zee
RIZA	Rijksinstituut voor Integraal Zoetwaterbeheer en Afvalwaterbehandeling
SEA	Strategic Environmental Assessment
SM	Suspended Matter
SUP	Strategische Umweltprüfung (SEA)
SUPG	Gesetz zur Strategischen Umweltprüfung (SEAA)
SQV	Sediment Quality Values
URE	Umweltrisikoeinschätzung (ERA)
UVP	Umweltverträglichkeitsprüfung (EIA)
UVPG	Gesetz über die Umweltverträglichkeitsprüfung (EIAA)
UVPVwV	Verwaltungsvorschrift zum Gesetzes über die Umweltverträglichkeitsprüfung
UVU	Umweltverträglichkeitsuntersuchung (EIS)
WFD	Water Framework Directive
WSV	Federal Waterways and Shipping Administration
ZBT	Saline-Dredged-Material-Test / Zoute-Bagger-Toets
ZV	Zielvorgaben / Quality Objectives

Dutch-German Exchange
on dredged material