Discussion Articles

Biological Effects-based Sediment Quality in Ecological Risk Assessment for European Waters

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Abstract

An overview is given of decision making frameworks for Ecological Risk Assessment (ERA) used for sediment in a number of European countries. These frameworks fall into two categories:
- Biological Effects-Based Assessment of *in situ* risks (referred to as *in situ* BEBA);
- Biological Effects-Based Assessment of the *ex situ* quality of dredged sediments (referred to as *ex situ* BEBA).

The first approach is usually part of an evaluation of whether remediation is needed in order to control or reduce the ecological risks of sediment pollution in a given location. The purpose of the second approach is to evaluate the risks of possible (unconfined) disposal options for dredged sediment (including sediment that is dredged for navigational reasons).
Important aspects are:
- Objectives for sediment management;
- The level of integration of BEBA in legal frameworks;
- The use of chemical (numeric) SQG's in BEBA and their integration with biological information;
- The criteria used to infer effects and to classify sediment quality.

Between EU countries the basis for deriving SQG's as well as the level of implementation of SQGs varies considerably. For use of SQGs in river basins, clearly there is a need for harmonisation of SQGs. Also, there is a large variation between EU countries with regard to the role BEBA plays in decision making frameworks. With respect to the implementation of the EU Water Framework Directive, possibilities arise for harmonization of BEBA on a river basin level, especially for *ex situ* BEBA.

1 Introduction

Since the mid sixties, national governments in a number of countries of Europe have become active in their attempts to monitor and control environmental pollution. Most efforts have focused on the development of environmental quality guidelines and their implementation in policies and regulations. After 35 years, guidelines have been still derived for less than one percent of the compounds known to be present in the environment due to human activities (Van Wezel 1999). Therefore, although much knowledge has been gained concerning the risks of this relatively small group of chemicals, it is clear that there is insufficient information about the risks caused by the majority of the other compounds that are released in the environment.

Special concern has been given to compounds that can bind strongly to particulate matter (especially clays and organic matter) and eventually are deposited in the sediment. Often in depositional areas historical pollution can be found that represents the sometimes unforeseen consequences of human activities. Historically, sediment quality has been assessed by making comparisons between concentrations of contaminants with (numerical) sediment quality guidelines (SQGs). Based on such a comparison, the potential risks, or hazard of (groups of) sediment-bound contaminants can be estimated. A recent overview of the use of SQGs in Europe has been given by Babut et al. (2003). An important aspect in the risks caused by sediment-bound chemicals is the degree of exposure encountered by sediment-dwelling organisms. It is well documented that often only a fraction of the contaminants bound to sediments are biologically available, in part because desorption can be slow. Thus, 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; De March 1987; Von Danwitz 1992), which may not be well addressed by single SQGs. For these reasons, and because of the large number of unknown contaminants as explained above, ecological risk assessment of sediment quality has received much attention in the past decades. The advantages of biological endpoints, such as effect bioassays, over chemical quality assessment are that biological testing integrates the effects of all contaminants present at their actual bioavailability (and detect possible combination or synergistic effects).

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Most so-called ecological risk assessment (ERA) frameworks deploy biological effects-based sediment quality assessments (BEBA). The basic principle of most of ERA is the use of multiple lines of evidence (Burton et al. 2002). Important lines of evidence are: 1) Assessment of the condition of the benthic zoomacroinvertebrate community; 2) Assessment of sediment toxicity by using bioassays (BEBA); and 3) Assessment of the potential effect occurring through foodchain poisoning (evaluation of bioaccumulation and biomagnification). The present paper is intended to give an overview of ecological risk assessment frameworks that are used in Europe, including the use of SQGs within those frameworks. One of the important points that needs to be addressed is that in this paper the term 'decision' will mean different things for the different countries. The information available will vary, as will the 'depth' of environmental policies. Therefore, an important question is what the purpose for ERA is in the different countries.

Two main goals for sediment quality assessment in Europe are distinguished:

- Biological Effects-Based Assessment of *in situ* risks (*in situ* BEBA) at sites where sediment quality and potentially sediment management is to be considered;
- Biological Effects-Based Assessment of the *ex situ* quality of dredged sediments (*ex situ* BEBA) in order to select sediment management options (e.g., free or confined disposal or treatment options).

The two objectives for carrying out a risk assessment procedure are different in nature and therefore are structured differently, as described herein:

1.1 *In situ* BEBA

The biological effects-based assessment of the *in situ* risks in sediment (*in situ* BEBA) focuses on location-specific conditions with respect to the bioavailability of contaminants and the assessment of the damage to the ecosystem. The assessment of damage to the ecosystem can be either predictive or retrospective. *In situ* BEBA can be considered as one of the lines of evidence in an ERA (Burton et al. 2002). ERA can be combined with studies focusing on the risks related to transportation of contaminants to the surface water and to biota, or to deeper sediment layers, and subsequently to the ground water.

With the growing concern for the potential problems caused by sediment pollution, ecological risk assessment approaches have been proposed as decision support tools or instruments for prioritisation. These approaches generally rely upon a tiered process, to allocate limited technical and financial resources.

In general, three main purposes can be identified for which *in situ* BEBA frameworks have been developed (Ingersoll et al. 1997):

- Integration of information from large numbers of parameters that use different lines of evidence (e.g., sediment chemical concentrations, sediment toxicity, benthic community measures, tissue concentrations, etc.);
- Proof of causality between environmental effects and sediment contamination;
- Tiered approach for increasing confidence in a cost-effective manner.

1.2 *Ex situ* BEBA

Ex situ BEBA is a *hazard assessment*, in which biological/toxicological endpoints are used as predictors of possible effects that may occur when the sediment is disposed of in the environment (aquatic or on land). In this BEBA, bioassays are often included in the sediment quality assessment or added as a second Tier. The approach is more prognostic, i.e. based on the outcome of the assessment, predictions are made of the consequences of free disposal of dredged sediments in the environment. In that respect this approach, using sediment toxicity assessment bears resemblance with total effluent risk assessments (see e.g. Grothe et al. 1996, Tonkes et al. 1999).

Comparing *in situ* and *ex situ* BEBA, it is likely that the assessments lie at very different levels in a decision making process (see also Apitz and White in this issue). *In situ* BEBA is usually a front-end investigation necessary to evaluate whether sediments are a risk, before any decision about some action would be needed. *Ex situ* BEBA is something that is carried out after it has already been proposed to dredge (e.g. dredging for nautical reasons), but when disposal options have to be considered.

Apart from the ecological risk assessment approaches explained above, there may be different concepts that use risk information for other questions, such as prioritising. These concepts will also be discussed in the present paper. From a number of countries we were able to obtain information on current practice with regard to sediment quality assessment. The present paper describes the stage of implementation of *in situ* and *ex situ* BEBA in national monitoring programs or in legal frameworks. The focus is on freshwater sediments, but for some countries there is more experience with *in situ* sediment quality assessment in estuarine or marine waters, or the evaluation of *ex situ* quality is more important for marine sediments, with regard to disposal of sediment at sea. Subsequently, BEBA applications for estuarine and marine waters will also be described. Selected BEBA approaches are described in more detail, such as the *in situ* BEBA used in The Netherlands, and a number of *ex situ* BEBA approaches in other countries.

2 Description of Integrative Sediment Quality Assessment (BEBA) Approaches in EU Member Countries

2.1 *In situ* BEBA approaches

2.1.1 Belgium

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 program. 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), sum of the chlorinated pesticides (SOCP), sum of 7 PCBs (PCB7), sum of 6 Borneff PAHs (PAH6), heavy metals Cd, Cr, Cu, Ni, Pb, Hg, Zn and As. The concentrations are normalised to values for sediment with a standard granular composition and organic carbon content (see description of normalisation in 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 to 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* sp (De Deckere et al. 2000).

Finally, the results are integrated based on the three classifications (**Table 1**). 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.

Table 1: Overall class assignments for sediments in Belgium using the TRIAD approach[a]

Chemical class	Ecotoxicological class	Biological class	Overall class
3 or 4	2, 3 or 4	2, 3 or 4	4
1 or 2	2, 3 or 4	2, 3 or 4	3
3 or 4	2, 3 or 4	1	3
3 or 4	1	2, 3 or 4	3
1 or 2	1	2, 3 or 4	2
1 or 2	2, 3 or 4	1	2
3 or 4	1	1	2
1 or 2	1	1	1

[a] Class 1 is good quality, class 2 is moderate quality, class 3 is poor quality and class 4 is very poor quality. See for other criteria (De Deckere et al. 2000)

For prioritisation of sediment locations, a method is proposed in Belgium that was previously developed for the prioritisation of pesticides in Flanders (Callebaut and Vanhaecke 1999). Apart from the information described above, some additional factors were added from the methodology for the risk assessment of existing chemicals (Van der Zandt and Van Leeuwen 1992). Each chemical parameter that assesses the exposure and the risk is categorised, resulting in a score for both the exposure and the risk. The scores are multiplied, which gives a final score that can be used for ranking the chemicals. However, some changes had to be made for sediments. The exposure of organisms to chemicals in sediments will also be assessed by the sediment-water exchange. Both the equilibrium between the solid phase and the interstitial water as well as the release of contaminants to the surface water will affect the sediment-water exchange. The human risk of chemicals in sediments is less important, because the direct exposure to sediments is negligible for humans. The most important pathway of human exposure is the risk of bioaccumulation throughout the food chain. These evaluations have yet to be incorporated in a legal framework.

2.1.2 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 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 / biological responses 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[1]. 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

[1] The standard method for invertebrates communities assessment is based on sampling of various habitats; the above mentioned approach looking for relationship between richness or abundance and sediment contamination did not discriminate between the habitats.

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 authorisation 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 urbanised areas, or in the vicinity of dams. Again, the dredging project will be subject to authorisation 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 one considers various management options, not only dredging (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-place' sediments risk assessment is currently under development on behalf of the French Ministry of Transportation.

2.1.3 Germany

Sediment quality assessment to determine *in situ* risks in Germany is mainly based on chemical quality criteria. Three main systems are used: The assessment systems of the Joint Water Commission of the Federal States 'LAWA' (Länderarbeitsgemeinschaft Wasser), of the ElbeRiver Water Quality Board 'ARGE ELBE' (Arbeitsgemeinschaft für die Reinhaltung der Elbe) and the "index of geoaccumulation" (Müller 1979). Whereby the LAWA and ARGE ELBE classifications are based on ecotoxicological effect levels of heavy metals, organic substances (industrial chemicals) and biocides, the index of geoaccumulation exclusively considers geochemical data, and does not take into account, that heavy metals have different effects on organisms. Nevertheless it is still widely used.

The LAWA-System: The classification of LAWA for sediment quality consists of 4 main and 3 sub classes based on data for 7 heavy metals, 28 organic chemicals, nutrients, salts and 11 sum parameters (ATV-DVWK 2000). Quality class I reflects a natural or potentially natural environment with no xenobiotic substances measured in the sediment and with average geogenic (natural) background levels of heavy metals. Class II includes target values that are expected to guarantee a high ecological protection. Target values for the water phase are based on ecotoxicological No Observed Effect Concentrations (NOEC), measured with four different test systems (**Table 2**) whereby the lowest measured concentration of the most sensitive species used is multiplied with a compensation factor of 0.1. According to the Federal Environment Agency, it is not yet possible to designate target criteria for the protection of sediment-dwelling organisms, due to the lack of generally acknowledged methods. Therefore soil limit values in force under the Sewage Sludge Ordinance are adopted as water quality targets for the asset of "Suspended solids and sediments" (Federal Environment Agency 2001). These quality targets, which make up class II of the LAWA system are considered to correspond to the "good environmental quality" of the European Water Framework Directive. The limits between the higher classes (II – IV) are derived from a multiplication of the target values by a factor of 2.

The ARGE-ELBE-System: the classification according to the ARGE ELBE is structured similarly to the LAWA system but uses target values that have been decided upon by the International Commission for the Protection of the Elbe. It uses 27 priority substances and includes arsenic because of its special importance for the Elbe (ATV-DVWK 2000).

The 'Index of Geoaccumulation' consists of a 7 tiered classification system, whereby the different classes derive from continued doubling of a background level. No biological considerations are involved.

Constructive criticism of the existing guidelines has increased in recent years, main points being the chemical based nature of the classifications, the doubtful transfer from sewage sludge to sediments for quality target levels, the high uncertainty due to the limited number of chemical substances measured, the resulting high potential for either underestimating or overestimating risk (Heise and Ahlf 2002). Several research groups have recommended application of a Triad approach, integrating chemical measurements, biological investigations and ecotoxicological measurements (Neumann-Hensel et al. 2000, Ahlf and Förstner 2001, Ahlf et al. 2002a, Hollert et al. 2002). An investigation of suitability of bioassays and biological classification methods has been prepared in Germany for the Federal Environment Agency in order to form a basis for a revision of Sediment Quality Criteria (Ahlf and Gratzer 1999). However, discussions are still ongoing.

Recently, recommendations were made for the use of an integrated stepwise approach combining toxicological, chemical and ecological information to assess and evaluate the quality of sediments (Ahlf et al. 2002b). A difference from approaches followed in most other countries is that bioassays are used as a trigger for further research steps, instead of the

Table 2: Data basis for the protection level 'Aquatic communities' used in Germany (Calmano 2001)

Primary producer: Green algae, e.g. *Scenedesmus subspicatus*	Growth within 72 hours
Primary consumer: Crustaceans, e.g. *Daphnia magna*	Reproduction within 21 days
Secondary consumer: Fish, e.g. *Brachydanio rerio*	Toxicity within 28 days (or 14 days), can be replaced by toxicity in early life stages
Degraders: Bacteria, e.g. *Pseudomonas putida*	Growth within 16 hours

chemical data that are more commonly used. In Henschel et al. (2002) a stepwise approach is described for an integrated assessment of ecosystem health effects and the consequences of sediment contamination for human health.

2.1.4 Italy

Presently, an *in situ* BEBA is not regulated in Italy and only recently some recent research and monitoring programmes have combined toxicological, chemical and ecological information. To assess the risks to biota, the Italian Dec. Leg. 152/99 (to implement 91/271/CEE and 91/676/CEE Directives) fixes Water Quality Objectives, and sediment quality assessment is considered supplementary. In freshwaters, two procedures are foreseen:

1. Basic analyses of water quality: anthropogenic impacts on biota are evaluated using the Extended Biotic Index (IBE), a biological species diversity index;
2. Additional analyses: not obliged by law, but suggested to investigate short or long term effects in particular cases. Toxicity test of concentrated water samples in *Daphnia magna*; mutagenicity and teratogenicity tests of concentrated water samples; algal development test; tests of concentrated water samples in bioluminescent bacteria.

In addition, bioaccumulation analyses of priority pollutants (PCBs, DDTs and Cd) on muscle tissues of fish or on macrobenthos are suggested.

As for sediments, analysis of a number of metals and organic micropollutants are considered as supplementary and performed in particular cases to determine causes of environmental degradation of the water body. In case it is necessary to highlight short or long-term toxic effects the Decree indicates to perform the following types of bioassays:

- Bioassay on sediment extracts;
- Bioassay on sediment as it is;
- Bioassay on interstitial water.

The following organisms are suggested for the bioassay tests: *Oncorhynchus mykiss, Daphnia magna, Ceriodaphnia dubia, Chironomus tentans* and *C. riparius, Selenastrum capricornutum* and luminescent bacteria.

As for coastal and marine environments, the Decree recommends a preliminary classification of the water quality based on dissolved pollutant concentrations. To finally define the quality of the coastal/marine environment, analyses on sediments and biota should also be performed. To asses the sediment quality the following analyses should be performed:

- Granulometric analyses;
- Heavy metals;
- Bioassay in different taxonomic group and with standardised protocols;
- Organic Carbon;
- PCBs and pesticides;
- TBTs (in the proximity of harbours).

To assess the state of the coastal/marine environment, bioaccumulation analyses of metals and organic pollutants (PAHs, PCBs, pesticides) in Mytilidae (*Mytilus galloprovincialis*) and Ostreoidea (*Ostrea edulis*, *Crassostrea gigas*) are of importance. If the indicator species are not present in the environment the following should be alternatively used: Tellinoidea (*Donax trunculus*) and Veneroidea (*Tapes decussates, Tapes philippinarum*). Additional analysis could be performed a) on key communities (phanerogame, reefs) to more completely characterise the ecological state of the environments, b) by the use of bioassay to test short and long term effects of pollutants in different taxonomic groups (favoring autochthonous species for which standardised protocols exist).

To classify the coastal/estuarine environment there are no existing integrated approaches assigning ranks. The environmental classification will therefore be based on existing trophic indices of species diversity.

2.1.5 Netherlands

In the Netherlands, sediment quality assessment has become part of routine monitoring programs, both in fresh- and marine waters. Different sets of SQGs have been implemented for a chemical classification of sediment quality on a scale from 0 to 4. These SQGs were to some extent based on ecotoxicological effect data (Van der Gaag et al. 1991; Van der Kooij et al. 1991). More recently, SQGs have been developed for sediment quality assessment using species sensitivity distributions (SSDs) for specific chemicals (Van de Guchte et al. 2000). Chemicals below their SQG value reflecting protection of ≥ 95% of the theoretically present species are considered to cause a tolerable degree of risk; chemicals higher than their SQG value reflecting protection of ≤ 50% of the species indicate potential high risk. The latter SQGs form, together with SQGs that have been derived specifically for assessment of potential human risk, the basis for the Dutch intervention value (see below).

Location-specific assessments of the *in situ* risks of sediment pollution in the Netherlands are carried out mainly in fresh water systems. The *in situ* BEBA is then part of a broader evaluation of the risks caused by sediment pollution, aimed at the question whether the risks make sediment remediation necessary. For this evaluation, a tiered approach is followed:

1. *1st tier assessment: Comparison of levels of priority pollutants with national standards/guidelines*. Chemicals measured routinely are mineral oil, chlorobenzenes, organochlorine pesticides, PCBs (standard group of 7 congeners), PAHs (16 of EPA) and the heavy metals Cd, Cr, Cu, Ni, Pb, Hg, Zn and As. Contaminant levels are normalised according to the approach described by CUWVO (1990), in order to compensate for differences in sorption characteristics between sediments[2]. Normalised contaminant levels are then compared with the Dutch sediment quality criteria (developed for first tier assessment of risks for human health and ecosystems). According to the resulting classification, most polluted sediments (exceeding intervention value(s): class 4 on a scale from 0 to 4) require a risk assessment (2nd tier).

[2] Standard sediment is defined as having a 25% particle fraction < 2 µm and 10% organic matter on a dry weight basis.

2. *2nd tier assessment:* The primary statement for this second line assessment is as follows: if a priority pollutant exceeds the intervention value (indicating potential high risks, e.g. in the case of ecological risks: species-sensitivity distribution for the pollutant indicates that > 50% of the theoretically present species might be affected), the site needs to be remediated urgently, unless it is shown that there are actually no high risks at that particular site. Thus, there is an assumption of risk, until it is disproved. If the data supplied from the second tier show that there is actually no high risk at a site where a priority pollutant exceeds the intervention value, the need for remediation is no longer considered as urgent. Conversely, if actual (high) risk were confirmed, the next step would review different remedial options, that are to be compared for the expected risk reduction. Three main pathways are considered within this tier for achieving a complete risk assessment, the third being an *in situ* BEBA approach:
   1. Human exposure: 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.
   2. Investigation of the risk of 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.
   3. *In situ* BEBA (see **Fig. 1**): 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 using indigenous organisms (Den Besten et al. 1995). Based on the *most sensitive* parameter, sediments are classified for the categories 'field observations' and 'bioassays' (**Table 3**) 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 normalised concentration and the lowest NOEC[3] 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 (**Table 4**).
3. *Prioritisation*: 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 no longer considered urgent. In the case where 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 urgently need to be remediated. In the Netherlands, some experience exists with the use of multi criteria analysis (MCA; also called Analytic Hierarchy Process – AHP; Saaty 1980) for this purpose. MCA enables a ranking of sites based on risks for the ecosystem.

[3] If sufficient data are available, the median value would be preferable.

Table 3: Classification of effects in bioassays in the Triad approach used in The Netherlands[a]

Daphnia	**Parameters (equal, take most sensitive)**			
	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%
Chironomus	**Parameters (equal, take most sensitive)**			
	Mortality eggs, prior to start sediment bioassay (incubation of eggs in elutriate)	Mortality larvae	Inhibition of development	Effect on weight (negative effects are scored only)
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%
Microtox	**Parameter: 1/EC20 (5, 15, 30 min: equal, take most sensitive)**			
Criterion 1	1/EC20 > 2			
Criterion 2	1/EC20 ≥ 10			

[a] NOECs are expressed as the % pore water dilution showing no effect; 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).

Classification of sediment can be done on each of the bioassays independently (i.e., most sensitive bioassay determines the toxicity score).

Effects on which the toxicity score depends should be significant at $p < 0.05$

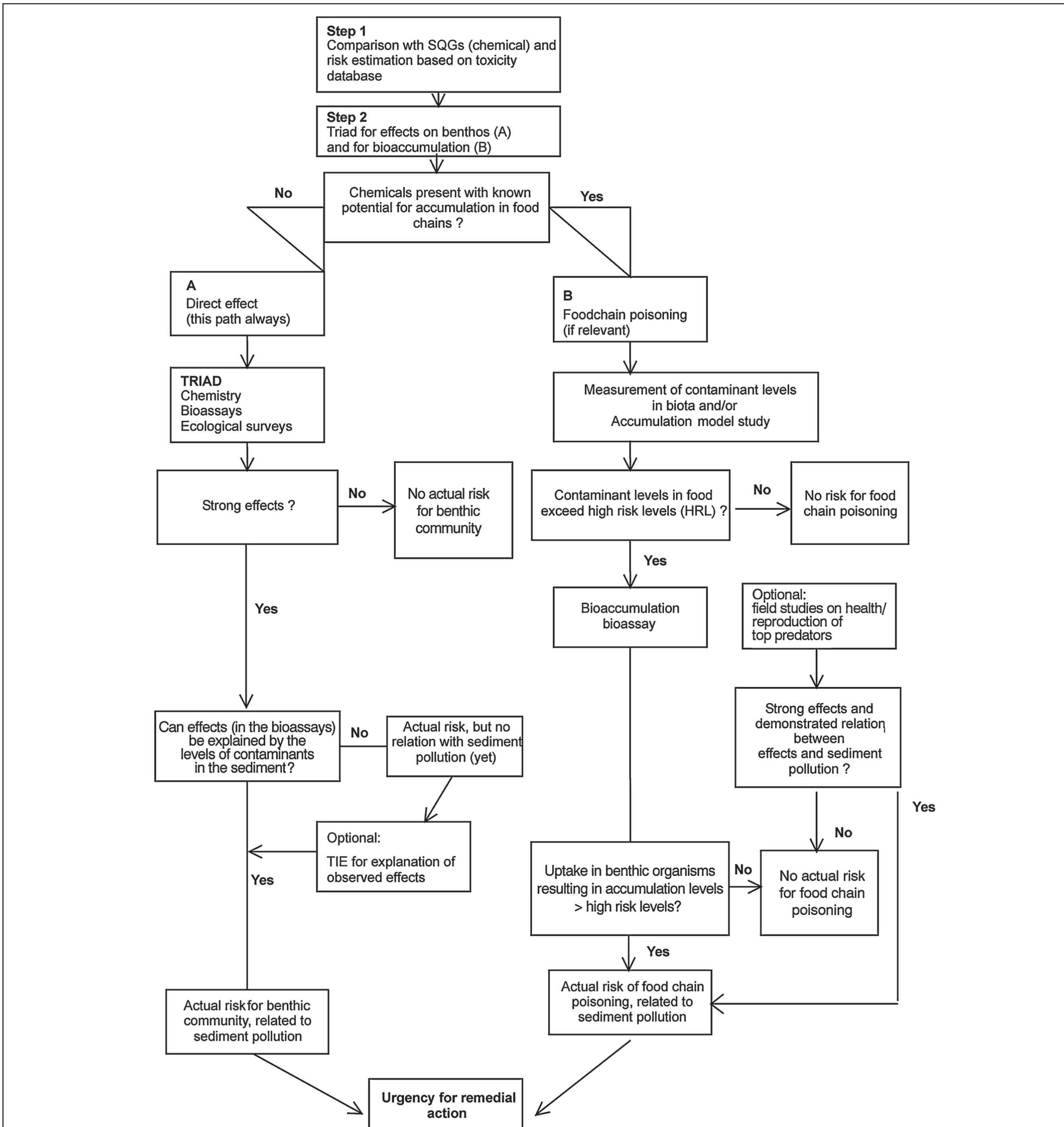

Fig. 1: Ecological risk assessment in NL for the decision yes/no urgency for remedial action

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 totalised 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

Table 4: TRIAD: Interpretation and Consequence for Remedial Action (after van Elswijk et al. 2001)

Effects on ecology (field studies) or Bioaccumulation levels in biota *	Effects measured in bio-assays (toxicity) or Bioaccumulation measured in bioassays *	Explained toxicity: $TU_{bioassay}$ ** or Comparison of levels of bio-accumulating contaminants in the sediment with 10xMPC or the PEL	Triad conclusion: actual ecological risk ?	Additional research needed ?	Decision based on Triad research with respect to urgency*** of remedial action and if applicable, recommendations for additional research
+	+	+	Yes	No	Actual risk demonstrated, urgency for remedial action.
+	±	+	Yes	No	Actual risk demonstrated, urgency for remedial action.
±	+	+	Yes	No	Actual risk demonstrated, urgency for remedial action.
–	+	+	Yes	Yes	Actual risk demonstrated, but urgency cannot be concluded yet. Additional research should be aimed at field surveys (other season, more parameters).
+	–	+	Yes	Yes	The serious effect(s) observed in the field were not confirmed by the outcome of bioassays. Additional research is recommended before remedial action is considered urgent. Possibly a low bioavailability of contaminants, or too short exposure in the bioassays.
+	+	–	Yes	Yes	Actual risk demonstrated, but urgency cannot be concluded yet (no relation between contaminants and observed effects). Additional research needed to identify cause of the effects.
+	±	–	Yes	Yes	Actual risk demonstrated, but urgency cannot be concluded yet (no relation between contaminants and observed effects). Additional research needed to identify cause of the effects.
±	+	–	Yes	Yes	Actual risk demonstrated, but urgency cannot be concluded yet (no relation between contaminants and observed effects). Additional research needed to identify cause of the effects.
±	±	+	indications only	Yes	Moderate effects, but they can be explained. Additional research necessary to confirm that risks are acceptable (repeat bioassays and/or use more parameters). No urgency
±	±	–	No	optional	Moderate effects, and no causal relationships between effects and sediment contamination. No urgency
+	–	–	No	optional	Effect seems not to be caused by contaminants. Possibly other factors have caused effects in field. No urgency
–	+	–	No	optional	Research needed in order to explain the results of bioassay(s). No urgency
±	–	+	No	optional	Moderate effects in field, but not in bioassays, although they would have been explainable on the basis of sediment contamination. Possibly low availability of contaminants. Additional research could focus on other bioassays (possibly more representative of benthic community in the field) or on other contaminants. No urgency
–	±	+	No	optional	Moderate effect, explainable. Additional research should be aimed at field surveys (other season, more parameters). No urgency
±	–	–	No	no	No urgency
–	±	–	No	no	No urgency
–	–	+	No	no	Contaminants cause no effects and seem to have low availability. No urgency
–	–	–	No	no	No urgency

* + = strong effect/risk; ± = moderate effect/risk; – = no effect/risk
** $TU_{bioassay}$: calculation of toxic units for explanation of effects observed in bioassays; + = TU for groups of contaminants or individual contaminants > 1
*** Criteria for urgency:
1. Category field or bioassays showing a strong effect (+); the other category at minimum moderate effect (±)
2. Effect in at least 1 bioassay should be explainable (either by calculating TU for a group of contaminants or for individual contaminants: TU > 1)
3. If criterion 1 is not met because a strong effect is observed in one category and no effect in the other, and effects could be explained, then actual ecological risk still is indicated, but no urgency for remedial action until additional research has been carried out to provide more effect data

studies. At a higher level of a decision 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 prioritisation of dredging locations.

2.1.6 Norway

Fresh and marine waters. In Norway a classification system for sediment quality has been developed to evaluate the environmental quality in freshwater and fjord/coastal water systems (SFT 1997a; 1997b). The classification of sediment quality is based on chemical concentrations and ranges from class I (insignificantly polluted) to class V (extremely polluted). The chemical parameters that are included in the assessment are TBT, hexachlorobenzene, sum of DTT derivates, sum of 7 PCBs, sum of toxic equivalents of dioxins and dibenzofurans, sum of 16 PAHs and the heavy metals Cd, Cr, Cu, Ni, Pb, Hg, Zn and As for fjord/coastal sediments and only heavy metals for freshwater sediments. The specific values used in each classification range are based on diffuse background levels for pollutants from a large data set and statistical methods. The criterion that distinguishes class I and II is this diffuse background level, with the criteria for the remaining classes based on a statistic scaling up of the background level value. The difference between class IV and V takes the pollutants' bioavailability into consideration.

There is currently no Norwegian national framework for conducting a biological effects based sediment quality assessment for *in situ* risks of sediment contamination. However, studies have been carried out in order to facilitate the development of such a system (Källqvist 1993). Test organisms included were algae (*Skeletonema costatum*) and crustaceans (*Acartia tonsa*, *Nitocra spinipes* and *Tisbe furcata*). A classification system was proposed for each test system independently, but not for an overall classification of sediment quality. Recent attempts to carry out risk assessment of contaminated sediments focus on bioavailability and bioaccumulation (utilising a test with a polychaete, *Nereis diversicolor* and a test with the snail *Hinia (Nassarius) reticulate*) (Hylland 1996). Furthermore, ecological effects on soft bottom fauna, as well as fluxes of contaminants and the relative importance of sediments as a source of contamination are considered (Skei et al. 2002).

The Norwegian Pollution Authorities (SFT) in cooperation with the Norwegian Institute for Water Research (NIVA) and the Norwegian Geotechnical Institute (NGI) are currently developing a national framework for conducting risk assessments of contaminated sediments. The experience and knowledge that has been gained through national projects as well as internationally are integral in the development of this framework. The structure and main components of the framework are to be completed by the end of 2003.

2.1.7 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. Under this provision, 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 the situation and is re-examining their policies related to sediment assessment and management. They commissioned two reviews to advise them. Environment Agency (2002a) reviewed the use of sediment quality guidelines and Environment Agency (2002b) reviewed the nature and extent of sediment issues, encompassing both ARPS and BEBA.

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 minimise '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. The Environment Agency (Harris et al. 2002) has identified the need for an increased focus on sediment assessment and management issues (in partnership with other stakeholders) and intends to develop an overall strategy, within which ARPS would sit.

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 organisations 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[4]. 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 three years from 2002) of these variables and their inter-relationships.

2.1.8 Spain

Freshwaters. No sediment quality assessment other than the use of chemical measurements is recommended for the management of sediment polluted in fresh water from Spain. However, no SQGs are recommended to evaluate chemical measurements in sediments. Most of the assessments carried out in these areas are based on the geochemistry of sediments. Some research includes the use of biological assays such as the commercial Microtox®, but only as complementary information. Most of the studies that have used both chemical and biological data were forced by the Spanish legislation related to hazardous material, but were not part of an integrated approach for sediment quality monitoring in these environments. The biological assays used for the assessment of hazardous material (mammalian tests, no fresh water organisms) are inappropriate for determining the quality of sediments.

Estuarine and marine waters. Most of the regulatory agencies both from the Central and Autonomous Governments follow a typical Triad schema to determine sediment quality in these areas. It is important to note that it is not a weight of evidence approach (e.g. the Triad as described by Chapman 2000), but a general monitoring using the main idea from the Triad: quantification of pollution based on the assessments of the contamination (enrichment of anthropogenic substances) and the effects associated with those polluted sediments. It is not mandated by any law but has been adapted by most regulatory agencies. The Triad approach has been used used in Spain since about 1992–1993 (DelValls et al. 1998). From the results obtained, different sediment quality values (guidelines) were derived by comparing them to those obtained in a similar Triad application carried out in San Francisco Bay (USA) and reported by DelValls and Chapman (1998). These Sediment quality guidelines haven been used following a tiered approach for the monitoring of the impact provoked by a mining spill (Aznalcóllar, April 1998) in some coastal areas located in the Gulf of Cádiz (Riba et al. 2002, Riba 2003).

The SQGs reported by DelValls & Chapman (1998) were used to evaluate the concentrations of certain heavy metals from the mining spill in different areas of an impacted estuarine and coastal environment (DelValls et al. 2002). Briefly, an ecological risk factor was derived by calculating the ratio of the measured concentration of the metals to their respective SQGs (Tier I). Those areas with ratios higher than 1 in some metals from the mining spill were considered as potentially impacted (polluted) and then tested using a battery of toxicity tests (both sediments and waters) and compared to other clean stations in the area with ratios clearly below 1 (Tier II). Based on the results of Tier II, stations were selected for application of TIER III, the sediment quality Triad. The Triad results were compared to those of an area known for its contamination by mining activities (e.g. Huelva). Such use of SQGs and integrated BEBA type of approaches is under consideration by different regulatory agencies and is being discussed by expert commissions for the monitoring of the impact in the Galician coasts after the oil spill provoked by the tanker 'Prestige'.

2.2 *Ex situ* BEBA

2.2.1 Belgium

No *ex situ* BEBA approaches exist in Belgium.

2.2.2 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 pieces of guidance, e.g. for the overall management process, has been introduced (Imbert 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 2001).

Tier 1: Chemistry based screening risk assessment[5]. In this proposed framework (**Fig. 2**), SQGs are used in Tier 1 in order to determine a 'risk score', which may trigger three decisions: (a) below a given value, the materials can be disposed of without specific requirements; (b) above another given value, the intended disposal should be further assessed through a detailed ERA; (c) between these values, complementary tests should be done.

[4] 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.

[5] Note that although this step was designed as a part of the risk assessment framework for dredged materials, i.e. for *ex situ* BEBA, it could also, and would probably, be applied as part of any sediment quality assessment, including *in situ* BEBA.

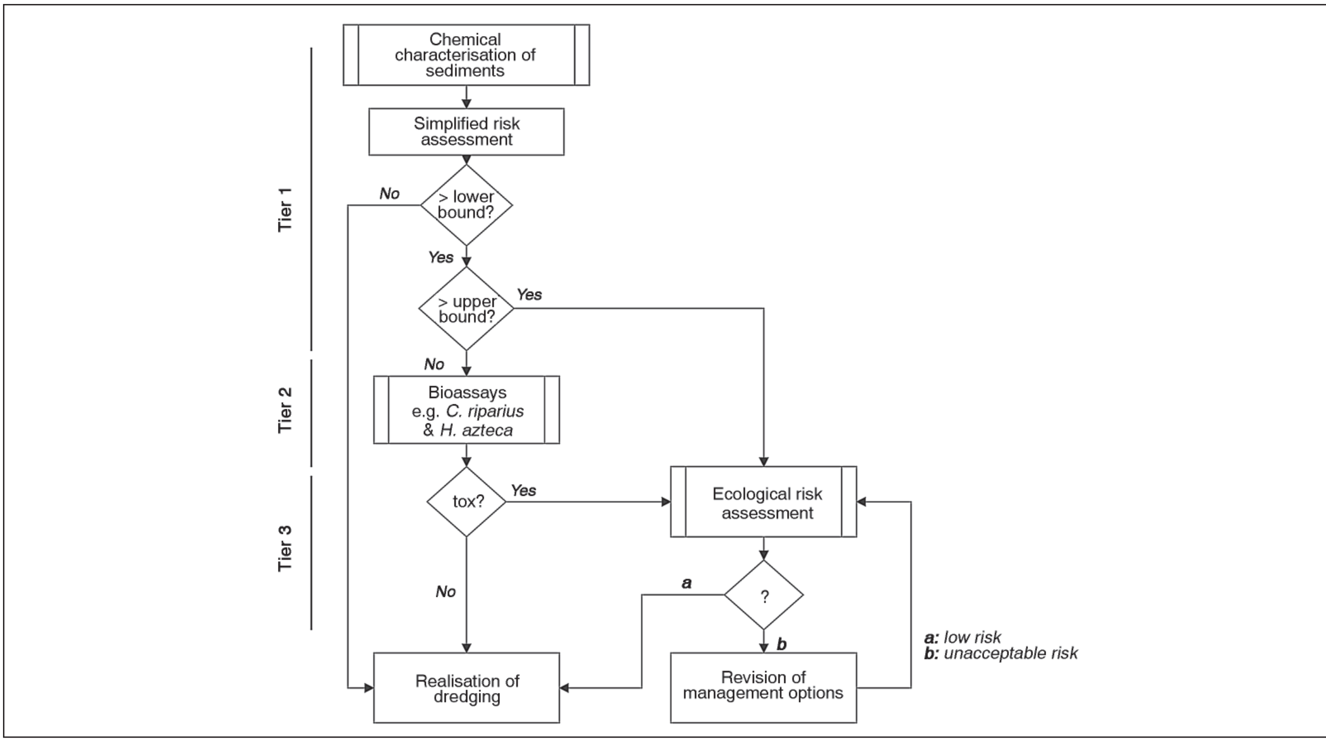

Fig. 2: Tiered framework for dredged materials *ex situ* BEBA in France

The initial proposal was based on consensus-based probable effect concentrations (PEC; MacDonald et al. 2000). The risk score was calculated as a mean quotient, following the same authors; trigger values were 0.1 and 0.5 for options (a) and (b) respectively. The former value, according to MacDonald et al., should correspond to a toxicity incidence of less than 10%, and the latter to an incidence of more than 80%.

As either the proposed SQGs or the trigger values were obtained from other countries, their relevance for French watercourses should be checked prior to their formal adoption. Other SQGs, e.g. those proposed by other French institutions, or SQGs used in other European countries, should also be tested. This work has been planned.

Tier 2: Refinement / hazard assessment. In case of doubt, i.e. if the hazard quotient value calculated at step 1 is in the intermediate range, a battery of two toxicity biotests would be applied. Two versions of the battery are currently being discussed: one is based on two whole-sediment tests, that is *C. riparius* and *H. azteca*, the second is based on a combination of a whole-sediment test (*C. riparius*) and a pore-water test (*B. calyciflorus*).

Tier 3: Detailed risk assessment. The tier 3 was primarily developed according to 2 different management scenarios, i.e. disposal on soil and disposal under water (in open gravel quarries) (Babut and Perrodin 2001, Babut et al. 2002).

2.2.2.1 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 kinds 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.

2.2.2.2 Disposal in water

The quarry is assimilated to a cross section of the alluvial groundwater. Therefore, the water will flow through the dredged material deposit. Possible contaminants will be eluted in 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 by various ways, in particular when they will colonise the deposit. The following criteria for allowing disposal in water have thus been proposed:

(1) the deposit should have no effect on the structure and abundance of benthic invertebrates in the quarry;
(2) it should have no long term effect on pelagic species;
(3) it should not cause groundwater pollution, as such quarries are in fact cross sections of shallow alluvial groundwater.

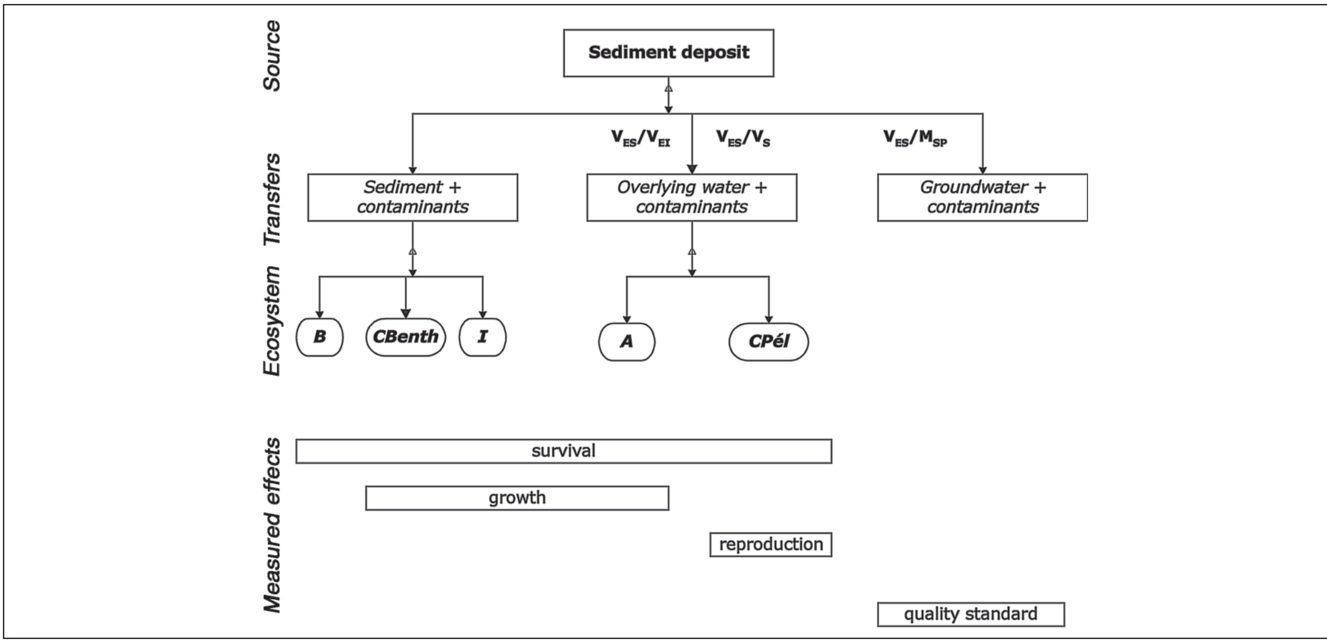

Fig. 3: Conceptual framework for disposal of dredged sediment in water. Abbreviations: B: bacteria; Cbenth: benthic crustaceans; I: insects; A: algae; CPél: pelagic crustaceans; V_{ES}/V_{EI} etc. designate dilution ratios

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™, algae, microcrustaceans *Ceriodaphnia dubia*, rotifers *Brachionus calyciflorus*), and leaching assays in columns under ascendant flow. The corresponding conceptual model is summarised in **Fig. 3**.

The first aspect (risk to benthic invertebrates) is assessed qualitatively, as it is difficult to draw dose-response curves with natural sediments. Therefore, lines of evidence are accounted for, to infer risk to the ecosystem. A more quantitative approach would be possible if a large database including chemicals' concentration levels and biological observations was made available.

The approach for the second aspect (risk to aquatic species) is more traditional: it consists of determining a PNEC (*probable no effect concentration* – here the pore water dilution corresponding to the lowest EC10), which is then compared to an exposure concentration, obtained from the pollutant concentration within the pore water and the ratio between the pore water volume to the total volume of the quarry. If the quotient 'PNEC to exposure concentration' is higher than 1, it will be considered that there is a risk to aquatic species.

The third aspect (risk to groundwater quality) is examined in a way similar to the second aspect, the PNEC being replaced by a drinking water standard, and the exposure concentration being the pollutant concentration in the elutriate. As in the previous case, if the quotient is higher than one, there is a risk of groundwater quality degradation.

The fourth aspect (risk to humans) has not yet been developed. Each tier of this framework is currently being tested through several application studies. An improved framework, with more developed guidance, is expected by the end of 2003.

2.2.3 Germany

There is no common, coherent policy in Germany on the assessment of dredged material. For the Federal waterways, the Federal Ministry of Transport (Bundesministerium für Verkehr BMV) and its subordinate authorities are responsible. Conceptual guidance and project monitoring with regard to environmental aspects are covered by the Federal Institute of Hydrology (Bundesanstalt für Gewässerkunde BfG). All other inland waterways are under the responsibility of the Länder (Federal states) which have their own guidelines and recommendations (see also: Hagner and Peters 2001).

For the Federal waterways two directives apply: the Directive for the Handling of Dredged Material on Federal Inland Waterways (Bundesanstalt für Gewässerkunde 2000) for the freshwater area, and the Directive for the Handling of Dredged Material on Federal Coastal Waterways (Bundesanstalt für Gewässerkunde 1999) for coastal waters up to the freshwater limit.

Federal waterways in the freshwater zone. The evaluation of dredged material is based on physical (e.g. grain size, water and organic content) chemical (7 heavy metals and arsenic, 28 organic compounds, 4 organic tin-compounds), biochemical (oxygen consumption, nutrients) and ecotoxicogical (test with green algae, luminescent bacteria, dapnia magna) characterisation.

Criteria for the relocation of dredged material include a comparison of the characteristics of the sediment with the characteristics of the proposed deployment area, where chemical and biological parameters are generally determined, with biochemical and ecotoxicological measurements being carried out only as an exception.

The decision system about the fate of the dredged material foresees three possible outcomes based on chemical criteria:

Table 5: Toxicity classes for sediment assessment and action levels for dredged material in Germany

Highest dilution level without effect	Dilution factor	pT-Value	Toxicity Class	Action category	Decision
Original sample	2^0	0	0	Not polluted	1
1:2	2^{-1}	1	I	Of no concern	
1:4	2^{-2}	2	II		
1:8	2^{-3}	3	III	Critically polluted	2
1:16	2^{-4}	4	IV		
1:32	2^{-5}	5	V	Dangerously polluted	3
1:64 or more	2^{-6} or more	More than 6	VI		

Decision 1 No limitations to relocation
Decision 2 Relocation possible based on a further evaluation of risks
Decision 3 Should not be relocated; decision based on ecotoxicological risk observations

1. no measured contaminant shows a concentration higher than 1.5 times the average concentration at the relocation site: Relocation is possible;
2. no measured contaminant shows a concentration higher than 3 times the average concentration at the relocation site: Relocation is possible, if no alternative measures can be applied and if no adverse effects are expected;
3. at least one measured concentration exceeds the average concentration of that substance at the relocation site by a factor of 3: Relocation is not possible. Other measures than relocation must be applied.

The ecotoxicological criteria used in Germany for the assessment of dredged material are listed in **Table 5**. The toxicity class is evaluated according to the pT-value of the most sensitive organism (pT = negative logarithm of the dilution factor necessary to reduce the effect below threshold). So far, for biochemical parameters no action limits are available.

Federal waterways in the coastal zone. The concept for the coastal zone area is in principle similar. The main modifications are a stronger emphasis on nutrient concentrations and the use of two interim action levels for phosphorus, nitrogen and the chemical substances that are to be measured.

Current discussion about necessary revisions of this risk assessment scheme focuses on the following aspects:

a) Case by case decisions by the risk managers reduce the transparency of decision making.
b) Basing decisions on the outcome of only one (the most sensitive) out of three ecotoxicological tests results in high uncertainty of the final conclusion on one side and in a low comparability of data for different sediments on the other because results from different test systems may be used for the final decision making.
c) Despite the fact that relocation is the favoured action with regard to dredged material, assessment of sediment bound toxicity is not included in the decision framework.
d) Using the pT-Value, no attention is paid to the toxicity in the undiluted sample within the risk assessment scheme. A high toxicity resulting from particle-bound contaminants may not be detected in the elutriates but may still pose a danger during relocation.

Local assessment schemes. The port authority of Hamburg carries out sediment tests prior to dredging. Although ecotoxicological measurements have accompanied the sediment testing for ten years now, decisions about the fate of dredging material are made solely according to chemical data.

2.2.4 Italy

For the assessment of sediment quality and management of disposal of coastal/marine dredged material, ICRAM is defining National Sediment Quality Guidelines. The use of bioassays and bioaccumulation tests is foreseen in the process of quantifying the environmental risk. Pilot projects on dredged sediment treatments, focused on sustainable re-use in the environment, are ongoing.

For some specific local areas, quality criteria for sediment management exist. In the Venice Lagoon a set of local quality criteria exists for evaluation of the concentrations of metals, PCBs, PAHs, HC and pesticides (Ministry of the Environment et al. 1993). Basically there are three classes of sediments that can be used inside the lagoon with increasing cautions related with the levels of pollutants. Sediments of Class A (good quality), can be reused for morphological restoration of the lagoon. Sediments of Class B (medium quality; easily manageable), can be reused in the lagoon islands but need to be permanently confined in order to avoid the release of pollutants in the water. Class C (poor quality; careful handling required), can be reused only in parts of islands that are permanently dry (no risk for flooding) and are not subject to erosion. Above the levels of the Class C is necessary to send the dredged sediments in a landfill on the mainland.

No bioassays are required for the evaluation of the quality of dredged sediments. However, due to the fact that the total pollutant concentrations do not give information on the availability and toxicity of contaminants, much emphasis has recently been put on the ecotoxicological evaluation of sediment quality. Therefore the use of bioassays and the comparison of the levels of contaminants in surficial sediments with reference values (ERL: Effects-Range-Low, and ERM: Effects Range-Medium) such as those proposed by US-NOAA (National Oceanic and Atmospheric Administration), is encouraged.

2.2.5 Netherlands

Freshwaters. Moderately and heavily contaminated sediments (Class 3 and 4) are to be transported to confined disposal sites, of which several have been built over the past decades. The management of only slightly polluted dredged material from freshwaters (Class 1 or 2) in the Netherlands is still under debate. Until now, Class 1 or 2 dredged sediments have mostly been put ashore or have been used in construction works (subject to certain criteria for the chemical composition). A risk assessment framework is being developed that predicts the risks of prolonged free disposal of dredged sediment, considering the specific function of the land where the sediment is to be disposed.

Table 6: Chemical and ecotoxicological criteria for the CTT test for evaluation of dredged marine sediment in the Netherlands

Test/compound	Group	Units	Criterion[a]
Amphipod *C. volutator*	combination toxicity	Mortality (%)	50
Microtox SP, bacteria *V. fisheri*	combination toxicity	Decrease bioluminicence ($1/EC_{50,corr}$)[b]	100
DR-CALUX, cell-line	dioxine-type	ng TEQ/kg dw	100
Tributyltin (TBT)	organometal	µg Sn/kg dw	100
Copper (Cu)	metal	mg/kg dw	60
Arsenic (As)	metal	mg/kg dw	29
Cadmium (Cd)	metal	mg/kg dw	4
Mercury (Hg)	metal	mg/kg dw	1.2
Chromium (Cr)	metal	mg/kg dw	120
Zinc (Zn)	metal	mg/kg dw	365
Nickel (Ni)	metal	mg/kg dw	45
Lead (Pb)	metal	mg/kg dw	110
Sum 10-PAHs	PAH	mg/kg dw	8
Hexachlorobenzene	OCP	µg /kg dw	20
Sum DDT's	OCP	µg /kg dw	20
Mineral oil (C10-40)	Oil	mg/kg dw	1250
Sum 7-PCB s	PCB	µg /kg dw	100

[a] Concentrations without standard correction
[b] EC_{50} corrected for fraction of fine silt

Marine waters. For the assessment of the possibilities of disposal of marine dredged material in the coastal waters of The Netherlands, a new sediment quality approach was recently developed, the chemistry toxicity test (CTT; **Table 6**; Stronkhorst et al. 2001). Three bioassays have been selected for routine application in the CTT approach, viz a mud shrimp toxicity test, a bacterial test (Microtox solid phase) and the DR-Calux assay, which reacts specifically to dioxin-type compounds. In the CTT approach, in order to allow free disposal of sediments, sediment quality guidelines must be met both for the concentrations of a list of chemicals and for the degree of effect observed in the bioassays.

2.2.6 Norway

There is currently no Norwegian national framework for conducting a biological effects based sediment quality assessment for the *ex situ* risks of dredged sediments. Norway is in a unique situation where the need to carry out dredging activities is limited. Deep fjord systems and the absence of large rivers limit the accumulation of sediment in most harbour areas. When dredging or dumping activities do need to be carried out, a permit must first be obtained. Dredging and dumping activities are specifically administered by the regional commissioner according to the regulation which controls dredging and dumping operations in the sea and watercourses (MD, 1997). The permit application must include "all information that is necessary to assess permit approval..., including characterisation of the material and site conditions...." However, the sediment characterisation requirements vary from region to region. Normally the sediment grain size distribution and chemical analysis of heavy metals, PCB, PAH and TBT are used to evaluate the application for dredging or dumping. When assessing the analytical results, it is common practice to use the upper limit of sediment quality class II to determine whether free aquatic disposal is acceptable.

2.2.7 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 sedi-

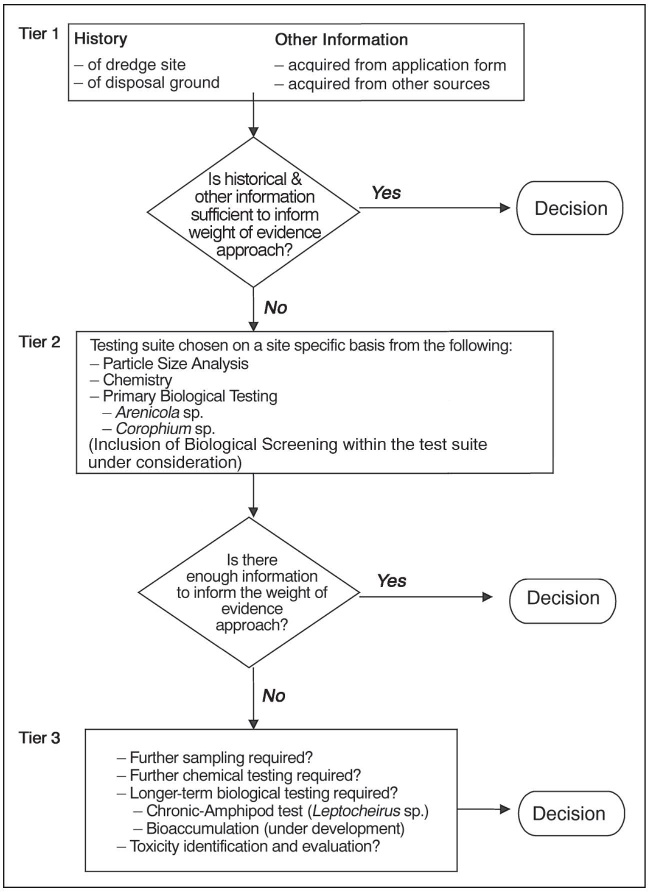

Fig. 4: Generic flowchart demonstrating the potential use of physical, chemical and biological tests as part of the weight of evidence approach being used by CEFAS (London Convention Scientific Group Meeting 2002) to assessing dredged material acceptability for disposal at sea in the UK

ment chemistry data (i.e., *in situ* BEBA). In addition, CEFAS are trialling a new dredged material disposal assessment decision tree (**Fig. 4**; 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' findings.

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 specify chemically-driven assessment procedures.

2.2.8 Spain

Freshwaters. No information is available related to the management of dredged material from freshwaters. Only limited information is available on chemical concentrations in dredged sediments. Only grain size and some other physical characteristics are measured. Based on the erroneous idea that the quality of fresh water sediments to be dredged is comparable to that of the soil/sediment along the shores of the rivers, most dredging operations consist of disposal of the dredged material on these shores.

Estuarine and marine waters. The use of SQGs to addresses the different possibilities of estuarine and marine dredged material in coastal waters of Spain has been recommended in recent years. Action levels for priority contaminants have been defined that can be used to evaluate the possibilities for the disposal of dredged material. Category I (free disposal to sea, only mechanical effects) is established for those dredged materials in which contaminant concentrations are equal to or lower than the action level 1. If the concentrations are higher than this action level but lower than the action level 2, the dredged material is considered category II (disposal to sea under controlled conditions followed by an integrated approach of environmental risk assessment study such as described for the *in situ* BEBA used in Spain). The dredged material with concentrations higher than action level 2 are considered category III (free disposal not authorized; confined disposal required). In case of remediation (treatment) of the material it could be re-assessed and then disposed under the recommendations of category II).

Recently, research has been carried out with the aim of incorporating the use of sediment toxicity tests under a tiered approach for the characterisation of dredged material in Spanish Ports. The battery tested included different benthic and pelagic taxas reflecting different exposure routes, including interstitial water, elutriate and whole sediment. Some screening (commercial) tests were included as well.

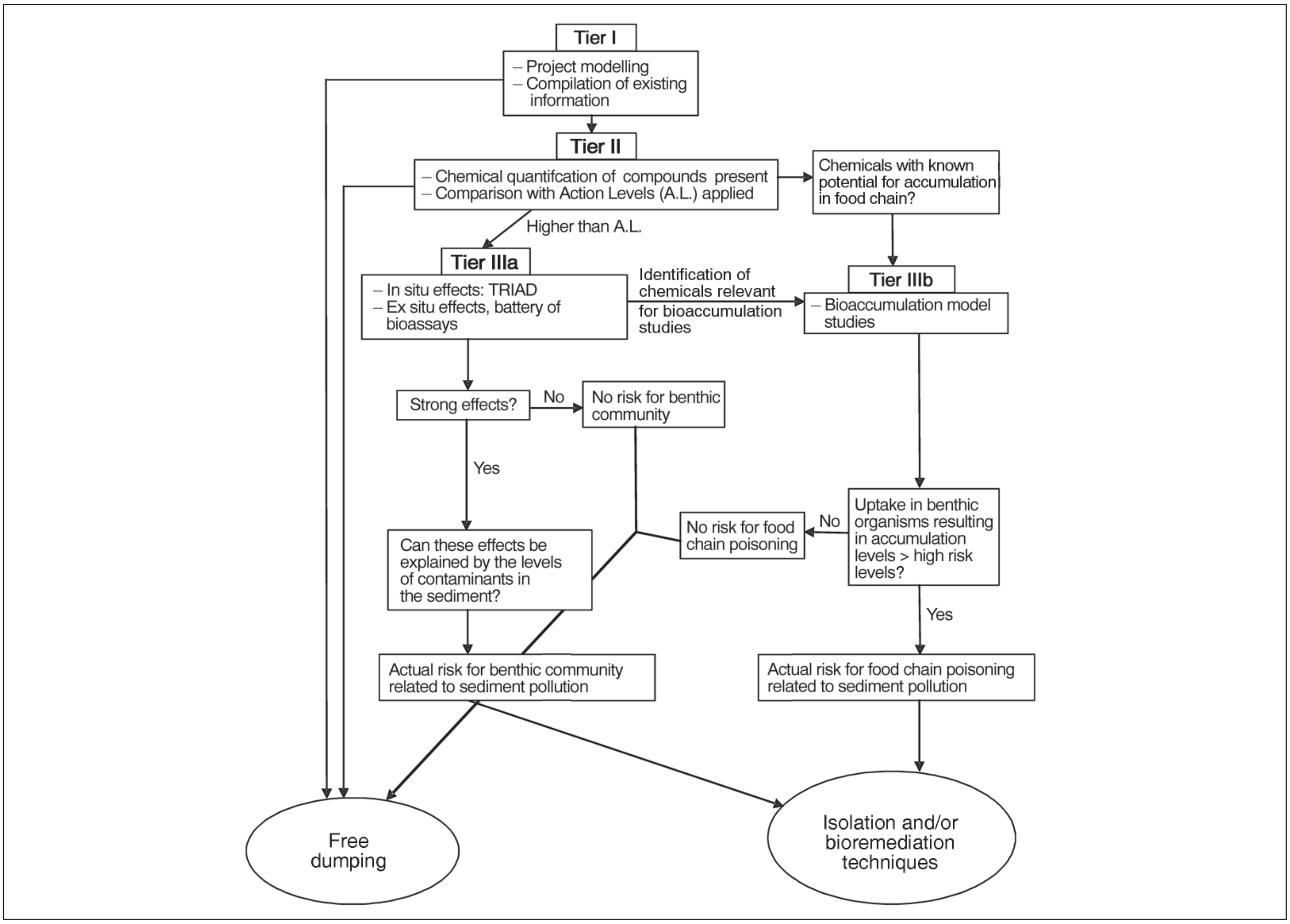

Fig. 5: Proposed assessment scheme for dredged sediment in Spain

The final objective is to propose a tiered scheme for the management and disposal of dredged material which could include the use of both chemical parameters (for comparison with action levels) and new and complementary tools such as bioassays. A proposal for such an approach is shown in **Fig. 5**. It is important to note that this approach is not included in the Spanish recommendations for the management of dredged material at this point.

3 Discussion and Conclusion

In **Table 7**, a comparison is made of to what degree chemical SQGs are implemented in legal framework. Between EU countries the level of implementation of SQGs varies considerably. Also the basis for deriving SQGs differs greatly (see also Babut et al. 2002). Three main sources can be identified for deriving SQGs: firstly the more or less arbitrarily chosen SQGs, secondly based on toxicity incidence among

Table 7: Status of development of SQGs in Europe

Country	SQGs not based on risk information (used for prioritisation) [a]	Empirical SQGs (e.g., based on laboratory tests on field samples)	SQGs based on toxicity data (*and* application of E-P theory)
Belgium	X	–	–
France	–	X	X
Germany	X	–	X (TBT only)
Italy	–	–	X (in progress)
Netherlands	–	–	X
Norway	X	X (in planning stage)	X (in planning stage)
United Kingdom	X	X (in planning stage) [b]	X (in planning stage) [b]
Spain	–	X	–

[a] These SQGs can be derived from comparisons of polluted sediments with contaminant levels in reference areas
[b] The United Kingdom would likely use a combination of both field and laboratory data, but research is in the planning stages

Table 8: Status of implementation of BEBA frameworks in Europe

Country	BEBA *in situ* risks	BEBA *ex situ* risks (of dredged sediment)
Belgium	In monitoring programs and for prioritisation	–
France	In development (in monitoring programs and for prioritisation)	In development: in monitoring programs, for prioritisation and for disposal
Germany	–	In development, used for prioritisation
Italy	Only in case of evident environmental degradation	–
Netherlands	In DSS	In DSS (for disposal at sea); for disposal of freshwater sediment on land still in development
Norway	In development	In development
United Kingdom	In development, used for prioritisation	In DSS (for disposal at sea and in estuaries)
Spain	In development	In development

large (field) datasets, and thirdly based on a combination of laboratory toxicity data in combination with a distribution model (equilibrium partitioning) among sediment, water and living organisms (Di Toro et al. 1991). All three types of SQGs are being used in sediment management in Europe.

The 'predictive ability' of most of these SQGs was not tested against national or regional datasets for validation: after they were derived, they have been applied without checking whether some degree of toxicity in the sediments is observed when these limits are exceeded. When comparing SQGs numerical values, some differences can be noted, which come from the differences in derivation methods (see Babut et al. 2003). For use of SQGs in river basins, clearly there is a need for harmonisation of SQGs.

In **Table 8**, a comparison is made of the degree of implementation of BEBA approaches in legal frameworks in Europe. As for SQGs, there is a large variation between EU countries with regard to the role BEBA plays in decision making frameworks. Only in a limited number of countries BEBA is part of a legal framework. In the other countries BEBA approaches are absent or in development. The same situation is found in the EU countries not listed here.

The application of bioassays provide ERA approaches with more information about the exposure of organisms in contaminated sediment. At the same time, this step forward also creates concern with regard to quality assurance of the techniques. Several issues are of great importance when using bioassays for the evaluation of sediment quality. Firstly, bioassays are subject to a number of confounding factors that may have nothing to do with contaminant load (such as grain size, ammonia, and countless other issues). Secondly, it is very important to define references and controls that are meaningful for the site under consideration. A third point of concern is the question whether all relevant modes of action can be covered by a set of bioassays. For instance, if only bioassays are used that measure acute toxicity, sublethal modes of toxicity (effects on fecundity, growth, immuno-competence etc) could be overlooked with important consequences for ecosystem health.

Chemical measurements have also developed over the past decade. At present, much more sophisticated methods are available that can characterise the bioavailability of contaminants (see e.g. Cornelissen et al. 2001, Burgess et al. 2003). These techniques may prove to be powerful in constructing lines of evidence between contamination and effects on organisms living in the sediment.

In order to be able to harmonise *in situ* and *ex situ* BEBA approaches between countries, it is necessary to discuss the goals for sediment management. From the perspective of the EU Water Framework Directive (EU WFD), it seems logical to harmonise the approaches on a river basin level. Because the EU WFD focuses primarily on water quality, it can be expected that *in situ* BEBA approaches will be used mostly as a diagnostic tool, i.e. to determine whether a poor ecological status of waters is caused by sediment contamination. These approaches may already be part of regulations, or will be. Therefore, harmonisation will be difficult, or take a long time, and probably dependent on a top-down approach (European directive). But for *in situ* sediment quality assessment this may not be necessary, except for the fact that clearly a need is felt to intensify the exchange of information on the criteria used to infer effects and to classify sediment quality. The situation might be different for *ex situ* BEBA approaches. When the disposal of dredged material in surface water is concerned, there will be a clear need to harmonise the sediment quality assessment in river basins. The *ex situ* BEBA can help to more effectively prioritise dredged sediment with high ecological risks, that should be transported to confined disposal sites. Such prioritisation can better be made based on effect observations than on chemical measurements, because biological responses integrate the effects of all biologically available contaminants.

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References

Ahlf W, Gratzer H (1999): Erarbeitung von Kriterien zur Ableitung von Qualitätszielen für Sedimente und Schwebstoffe – Entwicklung methodischer Ansätze. UBA Texte 41/99, 1–171

Ahlf W, Förstner U (2001): Managing Contaminated Sediments. I. Improving Chemical and Biological Criteria. JSS – J Soils & Sediments 1 (1) 1–7

Ahlf W, Braunbeck T, Heise S, Hollert H (2002a): Sediment and Soil Quality Criteria. In: Burden FR, McKelvie I, Förstner U, Guenther A: Environmental Monitoring Handbook. McGraw-Hill, New York, 17.1–17.18
Ahlf W, Hollert H, Neumann-Hensel H, Ricking M (2002b): A guidance for the assessment and evalutation of sediment quality: A German approach based on ecotoxicological and chemical measurements. JSS – J Soils & Sediments 2 (1) 37–42
Apitz SE, White S (2003): A Conceptual Framework for River-Basin-Scale Sediment Management. JSS – J Soils & Sediments 3 (3) 132–138
ATV-DVWK (Ed.), (2000): Die Elbe und ihre Nebenflüsse – Belastung, Trends, Bewertung, Perspektiven. Hennef, GfA Gesellschaft zur Förderung der Abwassertechnik e.V., 168 pp.
Babut MP, Perrodin Y (2001): Evaluation écotoxicologique de sédiments contaminés ou de matériaux de dragage. (I) Présentation et justification de la démarche. Rapport d'Etudes. Voies Navigables de France (VNF)-Centre d'Etudes Techniques Maritimes et Fluviales (CETMEF), 47 pp. (http://www.lyon.cemagref.fr/bea/tox/dragage.html)
Babut MP, Perrodin Y, Bray M, Clement B, Delolme C, Devaux A, Durrieu C, Garric J, Vollat B, Becart D, Charrier C (2002): Evaluation des risques écologiques causés par des matériaux de dragage: proposition d'une approche adaptée aux dépôts de gravière en eau. Revue des Sciences de l'Eau 15, 615–639
Babut MP, Ahlf W, Batley GE, Camusso M, De Deckere E, Den Besten PJ (2003): International Overview of Sediment Quality Guidelines and Their Uses. In: Wenning RJ, Ingersoll CG, Batley GE (Eds.), Use of Sediment Quality Guidelines and Related Tools for the Assessment of Contaminated Sediments – SETAC, in press
Bates AD, Hooper AG (1997): Inland dredging – Guidance on good practice. CIRIA Publication Code: R169, 184 pp. ISBN: 0 86017 477 8
Bundesanstalt für Gewässerkunde (BfG) (1999): Handlungsanweisung für den Umgang mit Baggergut im Küstenbereich (HABAK-WSV). BfG-Nr. 1100
Bundesanstalt für Gewässerkunde (2000): Handlungsanweisung für den Umgang mit Baggergut im Binnenland (HABAB-WSV). BfG-Nr. 1070
Burgess RM, Ahrens MJ, Hickey CW, Den Besten PJ, Ten Hulscher TEM, Van Hattum B, Meador JP, Douben PET (2003): An overview of the partitioning and bioavailability of PAHs in sediments and soils. In: Douben PET (Ed.), PAHs: An ecological perspective. Wiley, West Sussex, 2003, pp. 99–126
Burton GA Jr, Batley GE, Chapman PM, Forbes VE, Smith EP, Reynoldson T, Schlekat CE, Den Besten PJ, Bailer J, Green AS, Dwyer RL (2002): A Weight-of-Evidence Framework for Assessing Ecosystem Impairment: Improving Certainty in the Decision-Making Process. Human and Ecological risk assessment 8, 1675–1696
Callebaut K, Vanhaecke P (1999): Inventarisatie en evaluatie van de beschikbare gegevens omtrent emissie, immissie en gebruik van pesticiden voor de identificatie van de probleemstoffen in Vlaanderen en als basis voor de prioritering van de reductiemaatregelen. Eindrapport i.o.v. Ministerie van de Vlaamse Gemeenschap, Dpt. Leefmilieu en Infrastructuur, Administratie Milieu-, Natuur-, Land- en Waterbeheer, Afdeling Water
Calmano W (Ed.), (2001): Investigation and assessment of sediments: Ecotoxicological and chemical test methods (in German). Heidelberg, Springer-Verlag
Chapman PM (1996): Presentation and interpretation of Sediment Quality Triad data. Ecotoxicology 5, 327–339
Chapman PM (2000): The Sediment Quality Triad: then, now and tomorrow. Int J Environ Pollution 13, 351–356
Cornelissen G, Rigterink H, Ten Hulscher TEM, Vrind BA, Van Noort PCM (2001): A simple Tenax extraction method to determine the availability of sediment-sorbed organic compounds. Environmental Toxicology and Chemistry 20, 706–711
CUWVO (1990): Recommendations for the monitoring of compounds of the M-list of the national policy document on water management 'Water in the Netherlands: A time for action'. Commision for the Implementation of the Act on Pollution of Surface Waters (CUWVO), The Hague, Netherlands (in Dutch)
Den Besten PJ, Schmidt CA, Ohm M, Ruys MM, Van Berghem JW, Van de Guchte C (1995): Sediment quality assessment in the delta of the rivers Rhine and Meuse based on field observations, bioassays and food chain implications. Journal of Aquatic Ecosystem Health 4, 257–270
De Deckere EMGT, De Cooman W, Florus M, Devroede-Vanderlinden M-P (2000): Handboek voor de karakterisatie van de bodems van de Vlaamse waterlopen, volgens de Triade, 2de versie (Guidance document for characterisation of Flemish waters according to the Triad, 2nd version – in Dutch). Ministerie van de Vlaamse Gemeenschap, AMINAL/ afdeling Water, Brussel
DelValls TA, Forja JM, Gómez-Parra A (1998): An integrative assessment of sediment quality in littoral ecosystems from the Gulf of Cádiz. Environmental Toxicology and Chemistry 17 (6) 1073–1084
DelValls TA, Chapman PM (1998): Site-specific sediment quality values for the Gulf of Cádiz (Spain) and San Francisco Bay (U.S.A), using the sediment quality triad and multivariate analysis. Ciencias Marinas 24, 313–336
DelValls TA, Forja, JM, Gómez-Parra A (2002): Seasonality of contamination, toxicity, and quality values in sediments from littoral ecosystems in the Gulf of Cádiz (SW Spain). Chemosphere 7, 1033–1043
De March BGE (1987): Mixture toxicity indices in acute lethal toxicity tests, Archives of Environmental Contamination and Toxicology 16, 33–37
De Pauw N, Heylen S (2001): Biotic index for sediment quality assessment of watercourses in Flanders, Belgium. Aquatic Ecology 35, 121–133
DiToro DM, Zarba CS, Hansen DJ, Berry WJ, Swartz RC, Cowan CE, Pavlou SP, Allen HE, Thomas NA, Paquin PR (1991): Technical basis for establishing sediment quality criteria for nonionic organic chemicals using equilibrium partitioning. Environmental Toxicology and Chemistry 10, 1541–1583
D.Lgs. 11 May 1999, n. 152 (1999): 'Disposizioni sulla tutela delle acque dall'inquinamento e recepimento della direttiva 91/271/CEE concernente il trattamento delle acque reflue urbane e della direttiva 91/676/CEE relativa alla protezione delle acque dall'inquinamento provocato dai nitrati provenienti da fonti agricole', published in G.U. 29 May 1999, n. 124
Environment Agency (of England and Wales) (1997): EC Dangerous Substances Directive (76/464/EEC), Monitoring Requirements 1997. 11 pp. 6th December 1996
Environment Agency (of England and Wales) (2000): Proposed environmental quality guidelines for dioxins and furans in water and sediments. Prepared for the Environment Agency by WRc plc, Grimwood MJ, Mascarenhas R, Sutton A. R&D Technical Report P48
Environment Agency (2002a): Review and Recommendations of Methodolgoies for the Derivation of Sediment Quality Guidelines. R&D Technical Report P2-092/TR. Prepared by CEFAS for the Environment Agency of England and Wales. August 2002
Environment Agency (2002b): Scoping Study – Sediments in England and Wales: Nature and Extent of the Issues. Prepared by Beth Power for the Environment Agency of England and Wales. February 2002
Federal Environment Agency (Germany, Ed.), (2001): Environmental Policy – Water Resources Management in Germany. Part II -Quality of Inland Surface Waters, Federal Ministry for the Environment, Nature, Conservation and Nuclear Safety, Division WAI 1B
Garric J, Bonnet C, Bray M, Migeon B, Mons R, Vollat B (1998): Bioessais sur sédiments: Méthodologie et application à la mesure de la toxicité de sédiments naturels. 97.9004. Agence de l'Eau Rhône Mediterranée Corse, 53 pp.
Garric J, Flammarion P, Bonnard R, Roger MC (2002): Etude de l'impact de la contamination toxique du sédiment sur les biocénoses benthiques: Mise en relation micropolluants – Liste faunistique (IBGN). Agence de l'Eau Rhône-Méditérannée-Corse, 24 pp.
Grothe DR, Dickson KL, Reed-Judkins DK (Eds.), (1996): Whole effluent toxicity testing: an evaluation of methods and prediction of receiving systems impacts. SETAC Pellston workshop on whole efflu-

ent toxicity; September 16th–25th 1995; Pellston, MI. Pensacola FL: SETAC Press, 340 p.
Hamelink JL, Landrum P F, Bergman H L, Benson W H (1994): Bioavailability: Physical, Chemical and Biological Interactions. CRC Press, Inc., Lewis Publishers, Boca Raton, FL., p. 239
Harris, B, Tinsley D, Power B (2002): Navigating a course through dirty mud in England and Wales. In: Porta A, Hinchee RE, Pellei M (Eds.), Management of Contaminated Sediments, Proceedings of International Conference on Remediation of Contaminated Sediments. S1-2, p. 27–34
Hagner C, Peters C (2001): Current and Future Policies and Regulatory Framework – The European and International Policy Framework. In: Gandrass J, Salomons W (Eds.), Dredged Material in the Port of Rotterdam – Interface between Rhine Catchment Area and North Sea, Geesthacht, Germany
Heise S, Ahlf W (2002): The need for new concepts in risk management of sediments. JSS – J Soils & Sediments 2 (1) 4–8
Henschel Th, Maaß V, Ahlf W, Krebs F, Calmano W (2001): Gefährdungsabschätzung von Gewässer-Sedimenten – Handlungsempfehlungen und Bewertungsvorschläge. Teil III. Document für den Fachausschuss Gewässersedimente. Untersuchung und Bewertung von Sedimenten – Ökotoxikologische und chemische Testmethoden. Springer Verlag, Berlin, S. 493–496
Hermens J, Canton H, Steyger N, Wegman R (1984): Joint effects of a mixture of 14 chemicals on mortality and inhibition of reproduction of *Daphnia magna*. Aquatic Toxicology 5, 315–322
Hollert H, Heise S, Pudenz S, Brüggemann R, Ahlf W, Braunbeck T (2002): Application of a sediment quality triad and different statistical approaches (Hasse Diagramm and Fuzzy Logic) for the comparative evaluation of small streams. Ecotoxicology 11 (5) 311–322
Hylland K (1996): Bioaccumulation of environmental pollutants from marine sediment – Establishment of a test-system (in Norwegian). NIVA report 3537
Imbert T, Py C, Duchene M (1998): Enlèvement des sédiments – Guide méthodologique – Faut-il curer ? Pour une aide à la prise de décision. Pôle de compétence sur les sites & sols pollués Nord/Pas de Calais – Agence de l'Eau Artois-Picardie, Douai 161 pp.
Ingersoll CG, Dillon T, Biddinger GR (Eds.), (1997): Ecological Risk Assessment of contaminated sediments – Proceedings of the Pellston workshop on Sediment Ecological Risk Assessment, 1995. SETAC Special publications series, 389 pp.
Källqvist T (1993): Toxicity tests for characterisation and classification of polluted sediments (in Norwegian). SFT report 93:15. Norwegian Pollution Control Authority, Oslo. 26 pp.
Kraaij RH (2001): Sequestration and bio-availability of hydrophobic chemicals in sediment. Thesis Institute for Risk Assessment Sciences Utrecht University, Utrecht, The Netherlands
Long E, Chapman PM (1985): A sediment quality triad: Measures of sediment contamination, toxicity, and infaunal community composition in Puget Sound. Marine Pollution Bulletin 16 (10) 405–415
MacDonald DD, Ingersoll CG, Berger TA (2000): Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. Archives of Environmental Contamination and Toxicology 39, 20–31
Marine Pollution Monitoring Management Group (1998): National Monitoring Programme: Survey of the Quality of UK Coastal Waters
MD (1997): Regulation. The control of dredging and dumping in the sea and watercourses (in Norwegian). Fastsatt av Miljøverndepartementet 4. desember 1997
Müller G (1979): Schwermetalle in den Sedimenten des Rheins: Veränderungen seit 1971. Umschau 79, 778–783
Neumann-Hensel H, Ricking M, Hollert H, Ahlf W (2000): Empfehlung zur Bewertung von Sedimentbelastungen. Bodenschutz 4, 111–117
NRA (1995): Risk assessment of contaminated sediment. Prepared by WRc plc. R&D Note. Prepared by Fleming R et al.
Ministry of the Environment, Venice Water Authority, President of Veneto Region, Major of Venice, Major of Chioggia, President of Venice Province (1993): Protocol 8/4/93 for the Classification of dredged material, Venice, Italy
Riba I, DelValls TA, Forja J M, Gómez-Parra A (2002): Influence of the Aznalcóllar mining spill on the vertical distribution of heavy metals in sediments from the Guadalquivir estuary (SW Spain). Marine Pollution Bulletin 44, 39–47
Riba I (2003): Evaluación de la calidad ambiental de sedimentos de estuarios afectados por actividades mineras mediante métodos integrados. Tesis Doctoral. Universidad de Cadiz, 384 pp.
Saaty TL (1980): The analytic hierarchy process. Mc Graw-Hill, New York
Skei J, Olsgard F, Ruus A, Oug E, Rygg B (2002): Risk assessments related to contaminated sediments: With focus on Kristiansandsfjorden (in Norwegian). SFT report TA-1864/2002. Norwegian Pollution Control Authority, Oslo. 106 pp.
SNIFFER – Scotland and Northern Ireland Forum for Environmental Research (1995): Freshwater sediment assessment – Scoping study. Prepared by WRc plc, Report No. SR 3931/1 for SNIFFER. Prepared by Fleming R, Johnson I, Delaney P, Reynolds P
SFT (1997a): Classification of environmental quality in fjords and coastal waters (in Norwegian). SFT report 97:03. Norwegian Pollution Control Authority, Oslo
SFT (1997b): Classification of environmental quality in freshwater (in Norwegian). SFT report 97:04. Norwegian Pollution Control Authority, Oslo
Stronkhorst J (2003): An Effect-based assessment framework to regulate the disposal of contaminated harbour sediments in Dutch coastal waters. PhD Thesis, Free University, Amsterdam; also submitted to Marine Pollution Bulletin
Tonkes M, De Graaf PJF, Graansma J (1999): Assessment of complex industrial effluents in the Netherlands using a whole effluent toxicity (or WET) approach. Wat Sci Tech 39 (10–11) 55–61
Van de Guchte C, Beek M, Tuinstra J, Rossenberg M (2000): Quality guidelines for the management of water systems. Commission for integral water management, The Hague (in Dutch)
Van der Gaag MA, Stortelder PBM, Van der Kooij LA, Bruggeman WA (1991): Setting environmental quality criteria for water and sediment in The Netherlands: a pragmatic ecotoxicological approach. Europ Water Poll Control 1:13–20
Van der Kooij LA, Van de Meent D, Van Leeuwen CJ, Bruggeman WA (1991): Deriving quality criteria for water and sediment from the results of aquatic toxicity tests and product standards: application of the equilibrium partitioning method. Water Res 25, 697–705
Van der Zandt PTJ, Van Leeuwen DJ (1992): A proposal for priority setting of existing chemical substances. Studie i.o.v. EC, Dir-Gen voor Milieu, kernenergie en civiele bescherming
Van Elswijk M, Hin JA, Den Besten PJ, Van der Heijdt LM, Van der Hout M, Schmidt CA (2001): Guidance document for site-specific effect-based sediment quality assessment. AKWA report 01.005 / RIZA rapport 2001.052, Institute for Inland Water Management and Waste Water Treatment (RIZA), Lelystad, The Netherlands (in Dutch)
Van Wezel AP (1999): Overview of international programmes on the assessment of existing chemicals. Institute for studies on Human Health and the Environment (RIVM) report 601503 015
Von Danwitz B (1992): Zur Abschätzung der Schadwirkung von Stoffkombinationen autf aquatische organismen (On the estimation of mixture toxicity for aquatic organisms, in German), Thesis, University of Bremen, Bremen, Germany

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