Research Articles: Dredged Materials

Monitoring of Subaqueous Depots with Active Barrier Systems for Contaminated Dredged Material using Dialysis Samplers and DGT Probes

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Abstract

Background, Aims and Scope. Disposal of dredged material in subaqueous depots is increasingly considered an economic and ecologically sound option in managing contaminated dredged material. The concept of subaqueous disposals capped with active barrier systems has been developed to minimize this risk of contaminant release. As such a depot represents a permanent installation within a sensitive ecosystem, it requires a thorough monitoring concept. It is the goal of this work to develop such a concept regarding general considerations and results of laboratory and field investigations.

Methods. In addition to the state-of-the-art techniques developed for other under-water constructions, this monitoring concept is developed with particular respect to the chemical isolation of the dredged material from the overlying water body. It comprises the use of seepage meters, dialysis samplers, and DGT gel probes for determining the migration of selected target solutes. The capability of the dialysis samplers is demonstrated by comparing field results with model calculations. The appropriateness of DGT probes to assess the impact of humic substances on trace metal speciation and on copper toxicity is demonstrated with the aid of laboratory experiments.

Results and Discussion. The experimental results show that, by using dialysis samplers, the temporal changes in concentration-depth-profiles of heavy metals in the pore solution can be monitored. Additionally, the application of DGT probes facilitates the in situ detection of labile species of a metal in the presence of dissolved humic substances, which serves to reflect its toxicity.

Conclusions. Three subsequent monitoring phases are distinguished on the basis of both general considerations and the findings from field results: A hydraulic phase that is characterized by compaction and pore water expulsion, a geochemical phase in which the demobilization of pollutants can occur due to substantial changes in the physico-chemical conditions (pH, EH), and a steady-state-phase where pore water flow and geochemical conditions are approaching their minimum.

Recommendations and Outlook. The monitoring concept suggested here provides a versatile tool to assess the chemical isolation of subaqueous sediment depots and other contaminated sediment sites. This is of great importance as subaqueous disposal is increasingly considered a future management strategy as space for upland disposal is limited and treatment, in general, proves to be too costly.

Keywords: Active barrier system; DGT; dialysis sampler; dredged material; monitoring; seepage meter; subaqueous depot

Introduction

Where sediments are dredged either for the maintenance and the expansion of ports and waterways (maintenance dredging, capital dredging) or for the remediation of aquatic ecosystems (remedial dredging), the enormous amounts of contaminated dredged material (CDM) and its complex composition comprising diverse contaminants require a thorough management. As often (1) the material is not suited for open-water disposal, (2) treatment techniques are not applicable to the complex pool of contaminants, and (3) on-land disposal is unfavorable due to high costs and a lack of suitable landfills, Jacobs and Förstner (2001) pointed out the role of subaqueous depots with active barrier systems (SUBAD-ABS) as an economic and ecologically sound management option. The implementation of a SUBAD-ABS comprises a controlled deposition of CDM onto the river or lake bottom followed by the placement of an active barrier system (ABS), i.e. a reactive capping layer that may consist of a sequence of different reactive and stabilizing layers (Jacobs and Förstner 1999).

To prevent short-term contaminant losses during the implementation phase, the remobilization of sediment particles must be inhibited. It is hence most advisable to chose dredging and relocation techniques that meet these environmental needs, primarily by enhancing dredging and transportation without suspending the material with additional water (Möbius 2002).

The ABS minimizes long-term contaminant losses from the depot by stabilizing and isolating the sediment. For the long-term inhibition of contaminant losses the chemical isolation is of prominent importance, demobilizing the target contaminants by chemical reaction with the reactive ABS matrix.

The SUBAD-ABS concept entails an adequate monitoring program as an integral part of the concept itself. Monitoring programs for dredging and transportation, as well as for the subaqueous deposition, may be considered state-of-the-art and are discussed, for example, in detail in the PIANC guidelines for managing CDM (PIANC 1998). Monitoring the chemical isolation efficiency of ABS, however, has not been included in monitoring programs to date. It is therefore the aim of this manuscript to suggest such an additional monitoring component focussing on chemical isolation efficiency in SUBAD-ABS projects.

1 Monitoring Concept

Where a monitoring program for a SUBAD-ABS project is to be established, the following scheme shall be considered its basis:

- Identify all possible undesired effects of the project on the environment.
- Choose a means for monitoring the above effects.
- Define threshold values for key parameters.
- Set up an action plan for the above threshold values being exceeded.

While short-term adverse effects on the environment will mostly be related to the resuspension of fine sediment particles forming turbidity plumes, long-term effects will be controlled by the dissolution and subsequent diffusive or advective transport of contaminants with the pore water. The monitoring concept suggested here thus aims to measure and predict potential changes in pore water chemistry. As depicted schematically in **Fig. 1**, it is formed by three components:

- monitoring pore water flow
- monitoring concentration-depth profiles of key contaminants
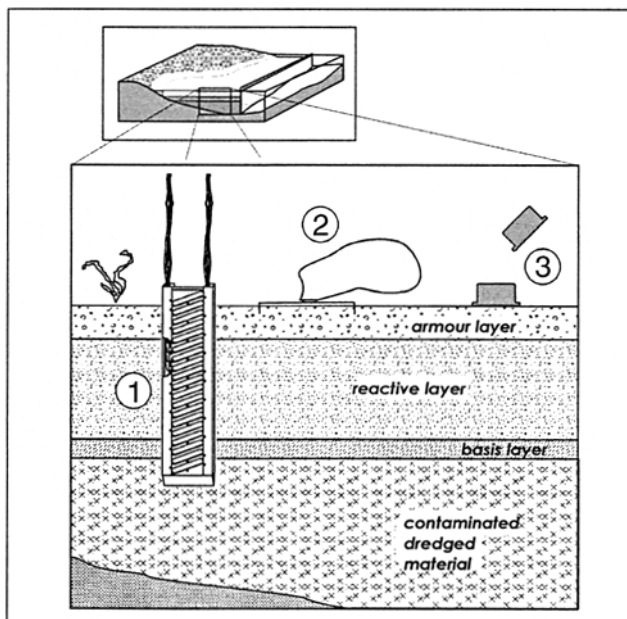- monitoring the release of key contaminants from the ABS surface

Fig. 1: Monitoring a SUBAD-ABS with (1) dialysis pore water sampler, (2) seepage meter, and (3) DGT.probes

1.1 Pore water flow

Changes in seepage rates, induced either by a compaction of the depot or by influent ground water, are of major importance when the efficiency of the chemical isolation by an ABS is to be assessed, as the volume of the infiltrating pore water along with its contaminant concentration provides the load the ABS has to compensate.

Seepage rates can be measured with simple seepage meters (e.g. Shaw and Prepas 1989, Blanchfield 1996, Cable et al. 1997a, Cable et al. 1997b) and, alternatively, automated devices have been proposed (Taniguchi and Fukuo 1993). A simple seepage meter consists of a metal drum that is inserted into the sediment with its open bottom end. On the top side there is an outlet where a collection bag is attached (Fig. 1). The amount of water collected in this bag at a given exposition time and a known area covered by the drum allows the direct calculation of the seepage rates.

1.2 Contaminant depth-profiles

Dialysis pore water samplers are employed for monitoring the depth profiles of key parameters. Pore water sampling devices based on diffusive equilibration were first developed by Hesslein (1976) and Mayer (1976). A pore water sampler as described by Hesslein, also known as 'peeper', consists of an acrylic body with a vertical array of dialysis chambers which are separated from the environment by a dialysis membrane. The chambers are filled with de-ionized water, which attains chemical equilibrium with the surrounding pore water by diffusive transport through the membrane after the peeper is deployed into the sediment. Subsequently, the peeper is retrieved and the samples obtained are preserved and/ or analyzed instantaneously.

Dialysis pore water samplers have become a well established tool for in situ sediment pore water collection in aquatic systems (e.g. Azcue et al. 1996, Bertolin et al. 1995, Brandl and Hanselmann 1991, Carignan et al. 1985, Shi et al. 1998) and numerous researchers have modified the basic peeper design according to their specific needs. Jacobs (2002) suggested a peeper design to meet the particular requirements of monitoring subaqueous sediment caps. To allow for temporal variations of pore water composition in addition to the spatial variation, a conventional peeper had to be deployed and retrieved repeatedly. This, however, (1) is costly where divers have do be employed, (2) may induce inconsistencies in the horizontal and vertical positioning of the sampler, and (3) may seriously disturb the natural condition in the sediment investigated. The peeper presented by Jacobs (2002) is designed to provide the capability of sampling and replenishing the dialysis chambers without retrieving the peeper from the sediment and, thus, to permit a continuous in situ monitoring at a given position within the SUBAD-ABS. This is achieved by sampling and recharging tubes being attached to each of the dialysis chambers (Fig 1). The sampling and recharging process can then be carried out from a boat or a pontoon by pumping the sample up and pumping de-ionized, oxygen-free water down at the same time. Two prototypes have been developed according to this principal. One, which is used for metal analysis in the field test described below, is made from Plexiglas and is designed to sample a profile of 1 m with a spatial resolution of 5.5 cm. PTFE and stainless steel were used for a second prototype, thus making it a more versatile tool, e.g. applicable to organic trace components as well. This prototype has a length of 50 cm and provides a spatial resolution of 2.5 cm.

When used for monitoring SUBAD-ABS the peeper primarily has to provide information on contaminant migration through a vertical profile of several decimeters. This profile comprises (1) the top of the contaminated sediment (mobilization zone), (2) the cap itself (retention zone) and (3) the

layer of surface water on top (protection zone). While temporal changes, indicating a possible breakthrough of contaminants, are of high priority, a relatively coarse spatial resolution of a few centimeters is satisfactory.

1.3 Labile metal species and effective pore water concentrations

In concert with the seepage measurement and the determination of concentration depth profiles, the application of DGT-probes provides an important monitoring option where toxic trace metals contribute to the target contamination. The DGT technique (Davison and Zhang 1994) uses the correlation between the diffusive flux of an aqueous species through a hydrogel and its actual concentration gradient. A DGT probe as employed here (DGT-Research Ltd., Lancaster UK) is a cylindrical piston assembly. On its top, two hydrogels are fixed by means of a tight fitting sleeve. The inner gel layer (resin gel) contains beads of a strong cation exchange resin (chelex) while the outer gel is simply a diffusive hydrogel on the basis of acrylamide and an agarose cross linker, providing the same diffusion coefficients for metals as in an aqueous solution (Fig. 2).
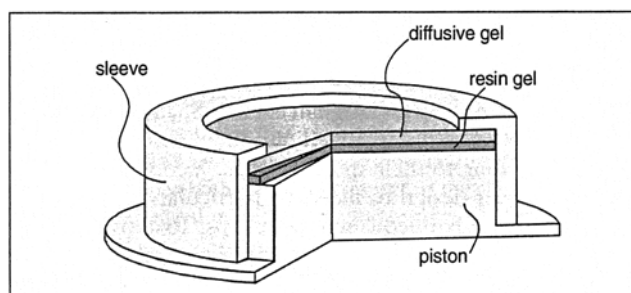

Fig. 2: Schematic section of a DGT probe (DGT-Research Ltd., UK) comprising piston and sleeve between which the resin gel and the diffusive gel are fixed

When the DGT probe is deployed into a solution, the diffusion of solutes through the window of the sleeve and into the diffusive gel starts immediately. Given that metal cations are sorbed quantitatively onto the resin gel and given that their concentration in the solution remains constant, a steady-state concentration profile builds up quickly after deployment. The diffusive flux is then defined according to Fick's law as:

$$J = D \ (C_{sol} - C_{gel}) \ / \ \Delta x \qquad (1)$$

where J is the diffusive flux, D is the diffusion coefficient, C_{sol} is the analyte concentration in the test solution, C_{gel} is its concentration in the resin gel and D is the thickness of the diffusion gel. For the steady state conditions described above, it may be reduced to:

$$J = D \ C_{sol} \ / \ \Delta x \qquad (2)$$

Furthermore, when the deployment time t, the area A and the mass of the accumulated analyte M is known, J can be obtained from:

$$J = M \ A \ / \ t \qquad (3)$$

Equating equations 2 and 3 consequently yields:

$$C_{sol} = M \ \Delta x \ / \ D \ t \ A \qquad (4)$$

allowing a direct calculation of the analyte concentration in the solution from the measured mass of accumulated analyte in the resin gel. A detailed description for the evaluation is given by Davison et al. (2000).

The analyte concentration in the solution, as determined by DGT, corresponds to the concentration of those aqueous species that are readily accumulated by the exchange resin. These species, accordingly, are all cationic species and those which readily release exchangeable cations. Therefore, this species group may be summarized as the group of labile species. Other species such as uncharged inorganic complexes or ions associated with colloidal matter are not determined by this technique. This operational speciation by sampling is of principal importance as it can be assumed that the labile species of a toxic trace metal contributes strongly to its toxicity in aqueous systems. Sediment pore waters in limnic systems often show very high contents of colloidal organic phases and, in particular, of humic substances with a high binding capacity for heavy metals. Therefore, the trace metal behavior and toxicity may be largely controlled by the DOC content. The DGT technique yields a means of distinguishing between the labile fraction and the DOC bound fraction of a trace metal within the solution, which will be demonstrated in the following section.

Hence, DGT probes represent an ideal tool for screening toxic fractions of trace metals in the water column overlying the SUBAD-ABS. Furthermore, the application of DGT benefits from the analyte being concentrated on the resin proportionally to the exposition time permit their detection at very low solution concentrations.

A second option in applying DGT-probes is to use them directly at the ABS-to-water interface. The probe is then pressed onto the ABS surface with its opening downwards and thus allowing a diffusion of the solutes from the pore water alone. The transport mechanisms are not as trivial as in the aforementioned case because non-steady-state conditions will apply as the pore water concentration cannot be assumed constant anymore. The pore water concentration here is controlled by the diffusive transport from the surrounding pore water and by sorbed species being released from the sediment matrix. An evaluation of the results accounting for both phenomena therefore has to employ numerical models such as DIFS, a computer program provided by the University of Lancaster. The concentration found can then be defined as an 'effective concentration' comprising the actual pore water concentration and an additional release term. This effective concentration was shown by Zhang et al. (2001) to correlate well with soil or sediment toxicity.

2 Results and Discussion

2.1 Field testing of dialysis sampler

Two protoypes of the new rechargeable dialysis sampler were used in a pre-pilot scale capping field experiment. This field experiment was conducted at the Vollert-Süd test site, a small lake near Leipzig, Germany. A cylindrical ABS test module (i.d. = 0.8 m, h = 1.0 m), divided into two separate vertical chambers, was filled to a height of 27 cm with a sediment from the *Weiße Elster* river. The sediment, being highly con-

taminated with heavy metals, was obtained from an upland disposal where it had been stored for years. It can hence be assumed that it was in a completely oxidized state.

A capping layer, consisting of a mixture of 10% natural zeolite (phillipsite) in clean quartz sand, was placed as a barrier in one of the two chambers on top of the sediment layer. A rechargeable dialysis sampler was placed in each of the chambers. The module was then inserted into an enclosure (i.d. 1 m, depth 8 m) in the lake and was finally fixed at a depth of 6 m. A continuous monitoring of the depth profiles of dissolved pore water constituents, comprising sediment, barrier, and water column, was conducted throughout a period of one year by analyzing the samples from the pore water samplers.

Model calculations using the computer program PHREEQC 2 were developed on the basis of the PHREEQC thermodynamic data base and additional data previously obtained from laboratory experiments on the ion exchange properties of the zeolite material. These calculations (**Figs. 3,4**) predicted that diffusive transport of conservative substances (e.g. bromide, Fig. 4) was to approach a steady-state condition within the observation period. For the boundary conditions, constant concentrations were assumed within the water column and, for the sake of simplicity, constant concentrations were used for the sediment pore water as well. The calculation further predicted an efficient metal retention by cation exchange within the zeolite barrier. This applies for trace metals (Fig. 4) and for the prevailing cations (Fig. 3), e.g. calcium in the pore water and iron in the surface water.

Consequently, a strong increase in the sodium concentration occurs in the barrier as sodium is released by the zeolite during the exchange process.

Analyses of the pore water samples obtained from the dialysis samplers depict a pattern resembling the one predicted by the above numerical model (**Figs. 5,6**). However, compared to the simplifying model calculations, the results from the field test show two important deviations. Firstly, we find a comparatively fast upward transport of the dissolved metals in the first few weeks after implementing the ABS module. This can be explained by the compaction of the sediment suspension due to the additional of the cap expelling the excess pore water into the cap. Secondly, we find that the pore water concentrations of heavy metals, of iron and manganese and as well of calcium are subject to considerable variations over time. Cadmium and copper are depleted to zero in the sediment and in the barrier after approximately eight months. This is apparently due to the demobilization of these metals by processes other than ion exchange on the zeolite matrix, presumably by precipitation of hydroxide, carbonate, or sulfide minerals. After one year, a mobilization of copper and cadmium from the sediment is observed again in a layer of approximately 10 cm below the sediment-to-cap interface. This mobilization event after one year is also reflected by the zinc, nickel, iron, manganese, and calcium profiles. It is beyond the scope of this manuscript to discuss the mechanisms inducing these mobilization and demobilization processes in detail. However, it shall be stated that geochemical reactions within the capped sediment,
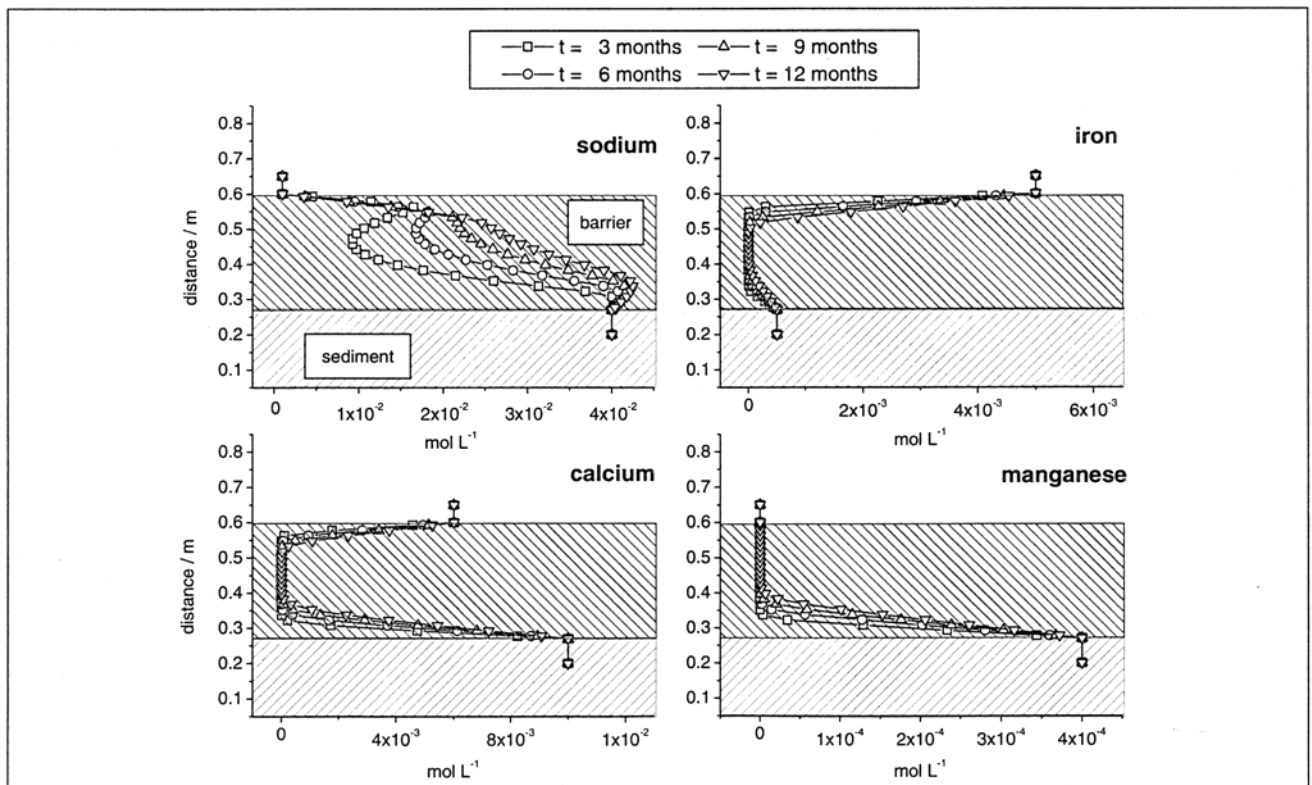

Fig. 3: Predicted changes of concentration depth profiles for Na, Ca, Fe, and Mn as calculated with a simplified 1-D transport model accounting for cation exchange as the only retention mechanism. Constant concentration in the sediment and in the water column were set as boundary conditions
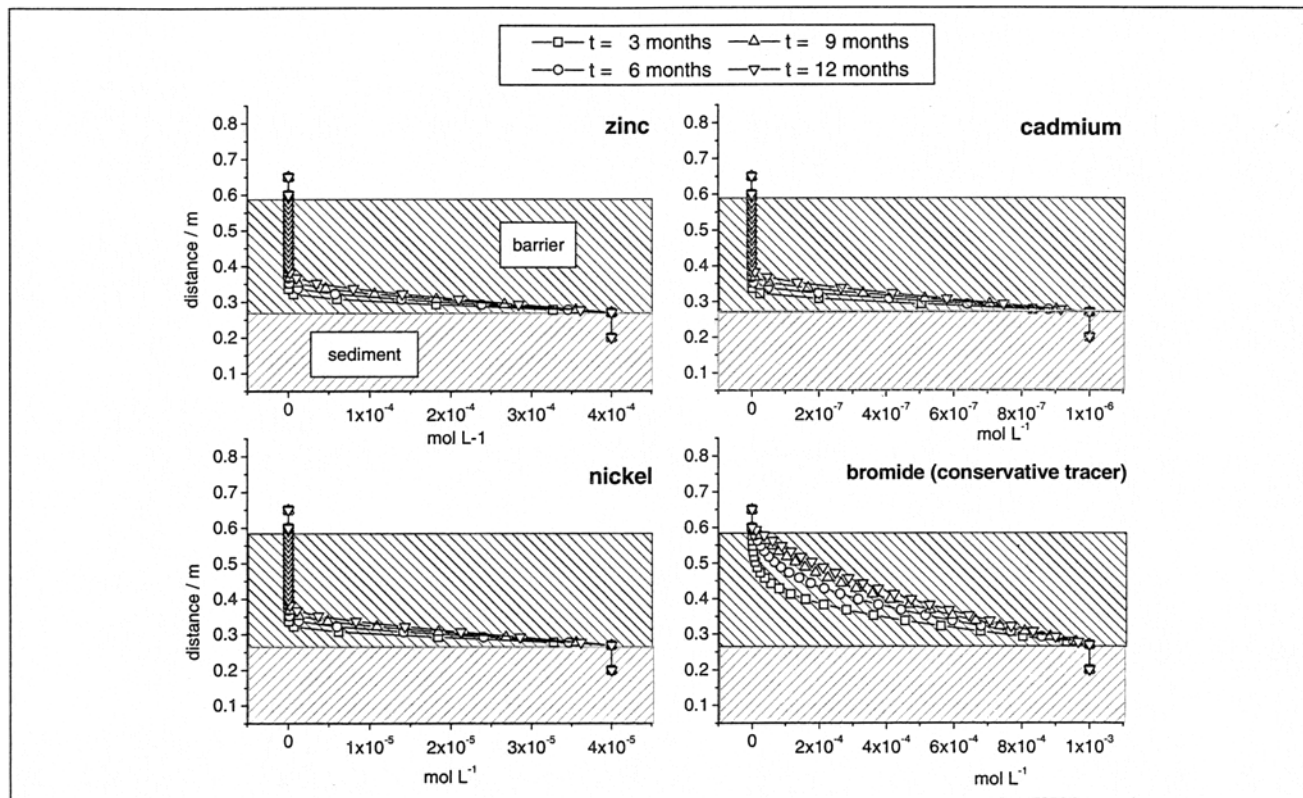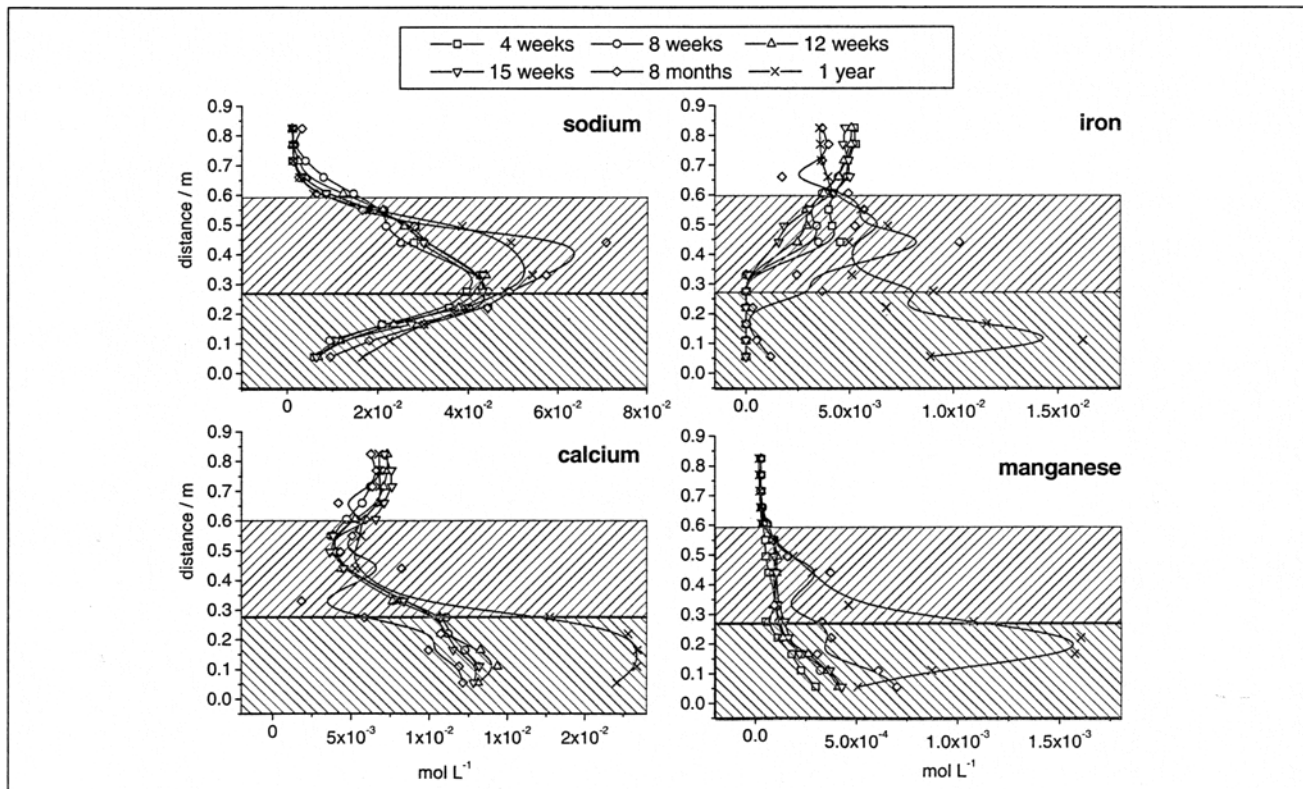
Fig. 4: Predicted changes of concentration depth profiles for Zn, Ni, Cd, and bromide as conservative solute calculated with a simplified 1-D transport model accounting for cation exchange as the only retention mechanism. Constant concentration in the sediment and in the water column were set as boundary conditions

Fig. 5: Measured concentration depth profiles for Na, Ca, Fe, and Mn in the capped compartment of the ABS test module monitored in the 'Vollert Süd' field test
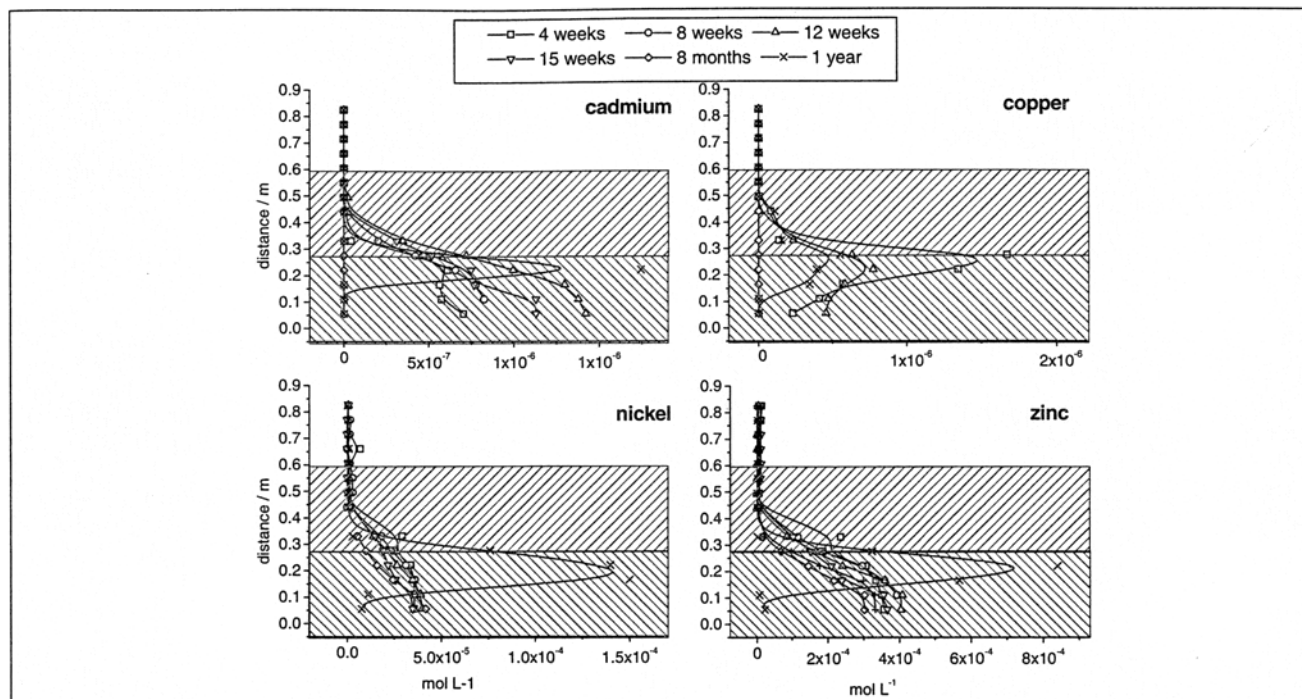
Fig. 6: Measured concentration depth profiles for Cd, Cu, Ni, and Zn in the capped compartment of the ABS test module monitored in the 'Vollert Süd' field test

which are mostly related to microbially mediated redox processes, must be expected to trigger extensive changes in pore water concentrations of various solutes and, hence, of the boundary conditions for evaluating and predicting the ABS-efficiency. A permanent monitoring of redox sensitive parameters and sediment-bound pollutants is therefore crucial, especially in the sensitive phase directly after the SUBAD-ABS implementation. Among these parameters are iron, manganese, nitrate, sulfate and E_H and pH.

In this field investigation, pH had only been determined by using test stripes and was found to constantly be at approximately 6.5. To give a more precise interpretation of redox associated processes, a more precise measurement of both key parameters, E_H and pH, is required. Unfortunately, the direct measurement is hampered by the sampling conditions, i.e. the sample slowly dripping out of the sampling tube at a flow rate of 1 mL/min under atmospheric conditions, and thus making measurements within the sample vials an unsuitable means. The easiest and most elegant, though costly, method to avoid this problem is to provide the sample tubes with flow-through cells and microelectrodes for on-line measurement.

2.2 Labile metal species in the presence of humic acids

As pointed out above, sediment pore waters in fresh water systems often contain elevated levels of humic substances that can bind large amounts of trace metal ions. Accordingly, the metal inventories can be operationally distinguished into the DOC-bound species fraction and the labile species fraction. These groups generally have clearly distinct properties, e.g. regarding toxicity. The DGT technique can be used to determine labile metal species both in the laboratory and in situ. The capability of DGT to determine the effect of humic acids on trace metal speciation was investigated by laboratory model

experiments. The impact of the humic acid content on the labile fraction of different heavy metals in model solution, as determined with DGT, is depicted in **Fig. 7**. A set of model solutions, each containing 100 µg/L of lead, copper, zinc, and cadmium as nitrates along with 20 mg/L sodium nitrate, was prepared in 1000 mL HDPE bottles. The humic acid concentration in the different solutions was 0, 0.8, 4, 8, and 40 mg/L DOC, respectively. The pH was adjusted to 5.5 in all solutions to minimize the impact of inorganic complexation and to avoid any precipitation of metal hydroxides or carbonates. The temperature was held constant at 25°C.

After obtaining samples of 20 mL for later analysis, DGT probes were then deployed into the solutions over 24 hours. During the exposition time, the bottles were placed in an overhead shaker at 25°C to maintain well-mixed conditions in the test solutions, i.e. no concentration gradients outside the diffusive gel. Afterwards, the probes were retrieved from the solution, the resin gels were peeled off and transferred into 1.5 mL reaction vials. Subsequently, 1 mL of a 1 mol/L nitric acid (conc. HNO_3
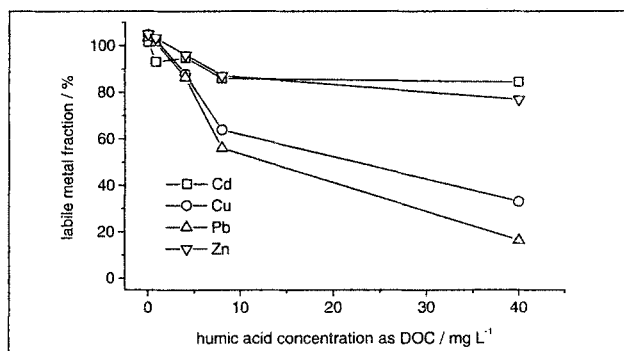

Fig. 7: Impact of the humic acid concentration on the labile fraction of heavy metals in model solutions containing 100 µg/L of Cd, Cu, Pb, and Zn

Suprapur in Millipore water) was added to each gel for elution over 24 h. After removing the probes again, solution samples of 20 mL were obtained from the bottles for analysis. Analysis for all heavy metals, and on all samples and eluates, was carried out with an ICP-OES (PerkinElmer Elan 2000).

Analyses of all samples from before and after the DGT experiment confirmed that no changes in the total metal concentration had been induced by the DGT-probe. The results of all resin gel analyses are depicted in Fig. 7 over the DOC concentration. These results clearly show that DOC concentrations, comparable to natural sediment pore water, have a notable effect on all tested metals. Furthermore, this impact is much more evident for copper and lead than for zinc and cadmium. For the first two metals, the humic acid associated fraction is by far the prevailing one even at moderate humic acid concentrations. This, of course, will play an important role with respect to the environmental behavior of these metals.

The impact of humic acid on copper speciation has been tested in another experiment, employing a set of humic acid solutions (10 mg/L DOC) with different total copper concentrations ranging from 25 µg/L to 300 µg/L at a pH of 6.8. The labile copper in these solutions was determined according to the procedure described above. The results yield a pattern of the copper distribution between the aqueous solution and the humic acid molecules (Fig. 8). Under the given experimental conditions, the free copper concentration rises slowly with the total copper concentration indicating a high distribution coefficient. At a total copper concentration of about 100 µg/L, the capacity of the humic acid is depleted and, henceforward, the labile copper concentration rises proportionally with the total copper concentration.

To investigate the effect of the distribution of copper between humic acid and aqueous solution, toxicity testing was carried out using the model solutions as in the previous experiment. An algae toxicity test, derived from the DIN 38412/ 33 test, using *Pseudokirchneriella subcapitata*, was applied over an incubation time of 72 hours at 20°C and permanent illumination. Inhibition of algae growth was then calculated from the measured chlorophyll fluorescence compared to reference solutions without copper.

$$inhibition\ [\%] = 100 - \left(\frac{flourescence\ \ [sample]}{flourescence\ \ [reference]} \cdot 100 \right) \qquad (5)$$

The inhibition was determined for dilution steps of 1:1.25, 1:2, 1:4, 1:8, and 1:16. The results of the first two dilutions
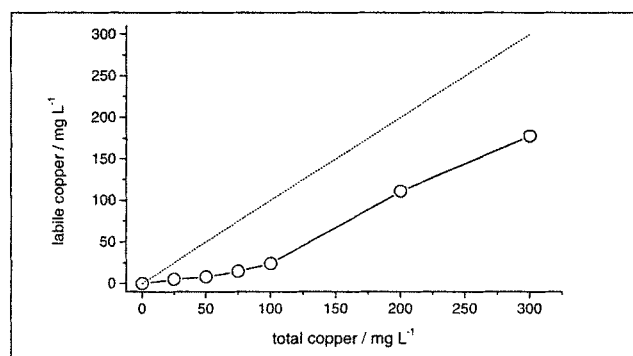
Fig. 8: Concentration of labile copper species plotted over the total copper concentration in the presence of humic acid (10 mg/L)

are plotted in **Fig. 9**, both over the copper concentrations of the labile fraction as determined using the DGT-probes and over the copper fraction bound to humic acid as calculated by difference. Comparing the correlation of labile copper and bound copper fraction with the toxicity on the algae, a correlation between the labile copper and the toxicity effect in the test solutions can be concluded.
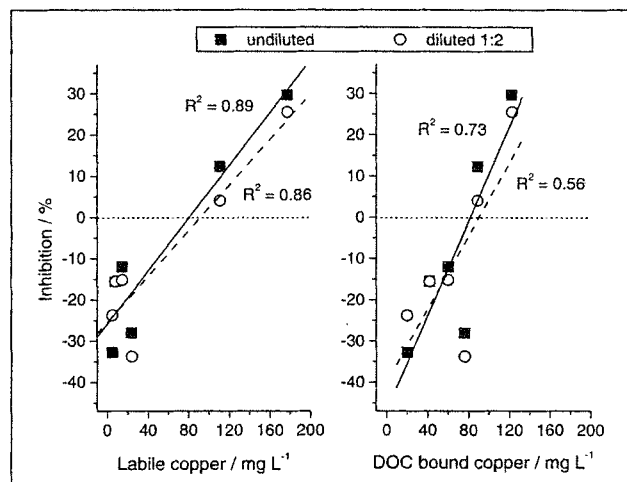
Fig. 9: Results from an algae toxicity test for copper in the presence of humic acid (10 mg/L) plotted over the labile copper concentration as analyzed by DGT (Labile copper, diagram left) and over the copper fraction associated with humic acid (DOC bound copper, diagram right). The correlation coefficients R^2 for the original test solution and a dilution 1:2 reflect the better correlation of toxicity with the labile copper concentration

3 Conclusions

Based on general considerations and on the field results the geochemical monitoring shall be subdivided into three phases.

*   Phase I (Hydraulic phase): During the first few weeks the natural seepage is superimposed by pore water being expelled from the sediment by compaction processes. The duration of this phase depends chiefly on the pore water drainage rates and thus on the water content and the hydraulic conductivity of both the sediment and the ABS. The contaminant load is mainly determined by the extraction of well soluble compounds or exchange processes. Monitoring has to be carried out frequently and must focus on the physical cap integrity (threatened by potential density inversion when drained sluggishly) and on the pore water seepage.

*   Phase 2 (Geochemical phase): The impact of sediment compaction is of minor importance for the flow conditions during this phase while steady state advective flow or diffusion become the prevailing transport mechanisms. Extensive release or demobilization of contaminants can take place induced by changes in the redox conditions or pH. For example, toxic metals can be released when ferric minerals are dissolved due to the sediment conditions approaching the corresponding redox conditions. These metals could then be eliminated again by precipitation as sulfides when the redox potential is further decreased. The duration of this second phase depends, first of all, on the redox and pH buffering properties of the sediment (v.d. Kammer et al. 2000), and it can be estimated to be between a few weeks and one year. Monitoring measures have to be

carried out frequently and focus must be made on potential changes in pore water chemistry.

- Phase 3 (Steady-state phase): The third phase is characterized by steady state geochemical and hydraulic conditions. Contaminant migration is quite well predictable and the monitoring intervals can be chosen less densely. This phase begins with the end of the second phase and continues until favorable conditions are achieved permanently. Such favorable conditions are attained, for example, by the sediment being entirely under sulphidic conditions or by freshly forming sediment on top of the SUBAD-ABS which acts as a natural barrier.

The three monitoring devices presented here have their specific advantages making them an important part of the suggested concept.

The use of seepage meters is the only direct way to determine the seepage rate and, thus, the advective transport component the ABS has to withstand.

The new rechargeable type of dialysis sampler plays an important role in the monitoring concept for capped subaqueous sediment depots as it

- is an easy-to-use tool for obtaining samples from the depot, the cap and the overlying water column.
- provides detailed information of changes in the depth profiles of pollutants or other target parameters over time.
- The DGT-probe provides features making it a most valuable tool for monitoring capped sediment depots:
- It is well suited for in situ use and it is easy to handle.
- It detects very low concentration target contaminants due to an enrichment within the resin gel during the deployment time.
- It measures effective concentrations at the cap to water interface, i.e. a concentration term that comprises the actual pore water concentration and a release term reflecting the tendency of the cap matrix to release demobilized contaminants.
- Results from DGT analyses provide an operational speciation and are softer than polarographic methods requiring sample preparation.
- DGT measured labile fractions have been shown to correlate with toxicity testing results.

4 Recommendations and Outlook

Subaqueous disposal of dredged material will play an increasingly important role as it provides agencies and port authorities with an economic and ecologically sound alternative in the management of dredged material. However, long-term monitoring of the disposal sites will be required to rule out negative impacts on the environment. The three parts of the monitoring concept suggested here are focused on the diffusive and advective transport of solutes and, therefore, on the chemical isolation efficiency of an ABS. It is, however, crucial to consider all mechanisms and processes which may either decrease the contaminant retention efficiency or potentially destabilize the depot and its cap. For the application within a future pilot-scale SUBAD-ABS project, a monitoring plan has recently been set up at the Technical University of Hamburg-Harburg putting particular emphasis also on groundwater seep-

age, on the one hand, and on gas formation and migration, on the other hand, as the latter may significantly threaten the physical stability of the depot construction.

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