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Evaluating Management Options for the Disposal of Dredged Sediments

ABSTRACT: This paper presents a screening framework that evaluates management options for the disposal of dredged sediments on the basis of chemical and toxicological characterization tests. The paper provides a combined brief overview of disposal options for dredged sediments and relevant testing, together with simplified decision rules for the feasibility of each option. The framework provides for two stages of assessment. In Stage I, disposal in open waters is evaluated based on the combination of results from chemical and toxicological analyses, supplemented by physical characteristics of sediments. For sediments that do not meet requirements for open-water disposal, Stage II evaluates the suitability of disposal in landfills or in confined facilities, upland and underwater. Decisions in Stage II are facilitated by well-established effluent criteria applied to ambient water and wastes admitted to landfills, as well as by calculations estimating the impact of sediment contaminants to the vicinity of the subaqueous confining structures. The application of the decision-making methodology is demonstrated using test results from sediment samples from Piraeus Harbor, Greece, and Indiana Harbor, IL, USA. Results showed that the framework is able to discriminate well among sediments from different subareas to be dredged. Disposal to open water is not feasible for most contaminated sediments from areas with increased shipping activities. Disposal in confined facilities (subaqueous or upland) proved to be a viable option for most of the sediments, whereas all sediments were characterized as nonhazardous and can therefore be accepted in an ordinary landfill.

KEYWORDS: sediment disposal, dredged material, environmental decision-making

Introduction

Dredging of sediments for port maintenance and navigation requirements has long been recognized as a disposal problem with characteristics of its own [1,2]. It is an issue akin to another well recognized need, that of remediation of sediments at contaminated sites [3], with one major difference. Management of dredged sediments deals with significant quantities of sediments that are repeatedly generated, often from areas of low- to medium-level diffused contamination. In contrast, restoration of contaminated sites calls for a one-time remedial measure for more highly polluted, localized sediments. As a response to these needs, detailed guidance documents for sediment characterization [4] and remediation [3] have been prepared. Meanwhile, decision-making frameworks have been proposed for the evaluation and disposal of dredged sediments [1]. These substantive sources provide valuable guidance but, understandably, do not facilitate a concurrent overview of disposal alternatives and characterization needs.

This paper extends previous work on dredged sediment management [5], aiming to combine in a single framework (i) available disposal options for dredged sediments, (ii) relevant testing, and (iii) simplified decision rules for the feasibility of each option. The framework was initially developed for marine sediments, but it can equally be applied to lake or harbor sediments. Throughout the paper, "open water" refers to both saltwater and freshwater bodies. The framework is meant to serve as a screening tool that is based on sediment-specific data when all disposal options are still "on the table." This is an important difference compared to the decision-making framework proposed by USEPA/USACE [1], according to which initial screening of disposal alternatives is conducted on the basis of general site characteristics, without results from sediment testing. The rationale behind the framework presented herein is that, given the routine nature of maintenance dredging, it is deemed preferable to include some of the more easily obtainable sediment characterization data already in the screening process. Naturally, for the final deployment of the

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selected option, the complete spectrum of environmental effects should be assessed [1,6] and results from more elaborate characterization tests should be obtained [4,7,8] as necessary.

Management Framework Components

Management Options

The most desirable management option for dredged sediments is their beneficial use. The single most important sediment-related criterion for the feasibility of this option is grain size distribution: a sizeable coarse-grained fraction offers useful earthwork material. USEPA/USACE [1] present a list of beneficial uses for the coarse fraction, which should be evaluated in the light of the contaminant load associated with the coarse fraction. However, the realization of such an option often hinges more on the fortuitous existence of a need for the coarse-grained fraction that makes economic sense, rather than on the contaminant load of the coarse fraction. For this reason, beneficial use is not incorporated into the proposed sediment disposal framework, although it should be considered first on a case-specific basis.

The next most desirable option for dredged sediment is open-water disposal. The dredged sediment will be transported at a suitable location and released to the sea or lake floor with various placement techniques [1]. When open-water disposal is not feasible, a variety of confined disposal options can then be evaluated. Confined aquatic disposal (CAD) options are gaining acceptability lately [9]: in addition to in-place capping of contaminated sediments, applications of ex situ subaqueous capping of dredged sediments are also being reported [10]. Dredged sediment is placed in an existing or excavated depression at the sea or lake bottom and covered with clean earth material, often coarse-grained, which offers physical and chemical isolation of the sediments. For the less contaminated sediments, placement in geocontainers may also be feasible if the physical isolation offered by the geosynthetic walls of the geocontainer turns out to be adequate. Geocontainers are filled with sediment in a sausage-like fashion and are most typically used for dewatering or as marine structures. When sediment characteristics preclude confined aquatic disposal, or when a suitable placement location is not available, confined land disposal (CLD) provides an alternative confined disposal alternative. Dredged material is discharged into open confined basins, built onshore, close to the coastline. Solids settle out and consolidate at the bottom, while excess supernatant water is allowed to periodically overflow to the sea or lake, following a suitable initial dilution. Finally, the least desirable option is disposal of sediments in landfills. In this case, waste acceptance criteria set for different landfill categories become relevant.

Characterization Tests

This section discusses the types of characterization tests that are suitable for the disposal options considered, at the screening level of the proposed framework. The most standard characterization test is *chemical analyses of sediment samples*. This test yields the total contaminant load, regardless of its potential to ever become dissociated from the sediment matrix and, hence, it provides the most conservative estimate of the sediment's impact at the disposal location. A more discriminating screening among alternative disposal options is possible when results from *chemical analyses of sediment pore water* are available. This becomes especially pertinent when the sediment matrix is physically isolated from ambient water, as is the case for underwater disposal in geocontainers or under a cap. Likewise, when sediment is placed in a confined upland facility, from where only the supernatant water is periodically disposed of, results from a *leaching test* are used to simulate the quality of the supernatant and assess its potential risk to marine and freshwater organisms [11]. The leaching test used for sediment samples evaluated in this study, from locations in Piraeus Harbor, Greece, and Indiana Harbor, IL, USA, was the test specified by European Standard EN 12457/1 [12]. This European Standard is specified in Council Decision 2003/33/EC [13] and establishes criteria in the form of limit values for the acceptance of waste to landfills. For the sediment from the United States location, the toxicity characteristic leaching procedure (TCLP) test (USEPA Standard Method 1311) was also used to characterize if the sediment is toxic and requires disposal in a hazardous waste landfill [14].

In addition, as numerous studies have shown that only some fraction of the contaminant load is bioavailable, results from toxicological analyses are also desirable. Given that the function of the proposed framework is that of a screening tool, tests providing indication of potential toxicity are deemed adequate. Two such tests were adopted for testing Piraeus Harbor sediments. The first is the *acid volatile sulfides/simultaneously extracted metals (AVS/SEM) test*, which is a proxy bioavailability test suitable for contamination by metals. The AVS/SEM test was originally proposed by Allen et al. [15]. Since then, many investigators have shown that the concentration of sulfides is a factor that controls the bioavailability of metals in sediments [16,17]. The concentration of AVS measures sulfide species in the sediments, to which metals are bound and become unavailable to microorganisms. If the total concentration of metals that are simultaneously extracted by this method is smaller than the AVS concentration, i.e., if $\Sigma\text{SEM}-\text{AVS} < 0$, then this is an indication of the absence of potential toxicity. The second toxicity screening test is a *luminescent bacteria test* used to determine toxicity of industrial waste effluent [18]. For the Piraeus Harbor sediments, it was carried out in a Lumistox 300, Microtox® type apparatus on liquid samples from pore water and leachate. Toxicity is estimated on the basis of luminescence loss (inhibition) of the microorganisms *Vibrio fischeri* that come to direct contact with the toxic substance for 25 min in a controlled environment (2 % salinity and temperature of 15°C). Toxicity evaluation of Indiana Harbor sediment was not performed in this study. It should be noted at this point that the specificity of toxicological analyses is amply emphasized in the literature [4] and the development of surrogate devices for bioavailability assessment is a topic of ongoing research [19]. Results from these devices will provide valuable input to frameworks guiding management decisions, such as the one presented herein.

Finally, as already mentioned, the feasibility of landfill disposal is evaluated based on leachate tests stipulated by established waste acceptance criteria. In order to minimize the amount of input data needed for the application of the framework, results from the same leaching test can be considered during the screening of both landfill and confined upland disposal. In the United States, TCLP test is used to determine if the material is toxic and requires disposal in hazardous waste landfill. Other landfill disposal regulations may stipulate additional criteria, such as prohibition of, or limitations on, free-draining liquid substances. In order to determine if the material contains free liquids, tests such as the paint filter test (USEPA Standard Test Method 9095) are required [14]. If free liquids are present, or present above limits, additional dewatering of the sediment will be necessary, which would render landfill disposal an even less acceptable option.

Screening Rules

The proposed framework employs screening rules of three types. Certain decisions involve comparisons of chemical analysis results to allowable limits. Landfill disposal belongs in this category, where contaminant concentrations in the leachate are compared to limits set for disposal of different waste categories. Another type of decisions is based on joint consideration of chemical data thresholds, toxicological data, and physical characteristics of sediments. For example, in the absence of indication for toxicity potential (e.g., $\Sigma\text{SEM}-\text{AVS} < 0$), sediments with contaminant concentrations between target and intervention levels are candidates for open-water disposal, especially if they have a low fines fraction. Similarly, for confined land disposal, the results from chemical and toxicological analyses of the leachate are jointly evaluated. Finally, evaluation of the underwater confined disposal options is based on estimates of the sediment's impact on the vicinity of the disposal locations, which can be obtained from a variety of simplified models, some of which are presented below.

Screening for Disposal in a Geocontainer—In order to estimate the impact of the contained sediment, approximate solutions for contaminant transport through the geocontainer wall and into the surrounding waters are needed. The simplest geometric approximation of the geocontainer is that of a line source. For diffusion-driven transport, if it is assumed that the entire contaminant mass M is applied instantaneously at time $t=0$, the contaminant concentration in ambient water, C_{aw} , at a distance r from the line source, at time t , can be calculated from the following expression [20]:

$$C_{aw}(r,t) = \frac{M}{4\pi Dt} \exp(-r^2/4Dt) \quad (1)$$

where D is the diffusion coefficient of the contaminant in water. In order to estimate the contaminant mass M , the measured total contaminant concentration in the sediment, C_t , expressed in contaminant mass per dry mass of solids, needs to be converted to contaminant mass per total sediment volume, C_v , as follows:

$$C_v = C_t \times \rho_d \quad (2)$$

where ρ_d is the dry density of the sediment. An upper bound estimate of the contaminant mass M diffusing in water (per unit length) can be obtained by multiplying the volume of the geocontainer (per unit length) with C_v . Realistic reductions of this upper bound estimate are possible if the proportion of the contaminant mass that can be dissociated from the sediment matrix is determined through sequential extraction tests (e.g., Ref. [21]). Then, C_v can be reduced accordingly.

Alternatively, the geocontainer geometry can be approximated with a half cylinder of infinite length and radius α . For the assumptions of an instantaneous source again and diffusion-driven transport in a homogeneous medium with a diffusion coefficient D , Crank [20] gives the concentration C_{aw} at a distance r from the axis of the cylinder, at time t , as:

$$C_{aw}(r,t) = \frac{C_o}{2Dt} \exp(-r^2/4Dt) \int_0^\alpha \exp(-r'^2/4Dt) I_0\left(\frac{rr'}{2Dt}\right) r' dr' \quad (3)$$

where C_o is the uniform concentration in the half cylinder initially and I_o is the modified Bessel function of the first kind of order zero. The integral in Eq 3 can be evaluated numerically by mathematical software. The initial concentration in the half cylinder, C_o , is taken equal to C_v , i.e., it corresponds to a fictitious concentration of the total contaminant mass distributed in a half cylinder containing only water.

The screening rule for the feasibility of disposal in a geocontainer consists of comparing the concentrations resulting from Eqs 1 or 3, for distances reasonably close to the geocontainer and for reasonably extended lengths of time, to allowable concentration limits in ambient water. It should be noted that, irrespective of the assumptions involved in the calculation of M in Eq 1, or, equivalently, C_o in Eq 3, both equations produce conservative solutions, since ignoring the presence of the sediment matrix and, to a smaller extent, of the geocontainer wall results in faster diffusion.

Screening for Disposal Under a Cap—Two different approaches can be followed in order to estimate the impact of the capped sediment. In the first approach, the steady-state contaminant flux, F_{ss} , is calculated from the following expression [22]:

$$F_{ss} = \frac{D_e(C_{pw} - \bar{C}_{aw})}{H_c} \quad (4)$$

where D_e is the effective diffusion coefficient in the cap, C_{pw} is contaminant concentration in the pore water, \bar{C}_{aw} is the average contaminant concentration in the water column (which can be assumed—conservatively, and only for Eq 4—to be equal to zero) and H_c is the cap thickness. In the absence of pore-water measurements, C_{pw} can be estimated from C_t , as:

$$C_{pw} = C_t/[K_p + (n_s/\rho_d)] \quad (5)$$

where n_s is sediment porosity and K_p is the partition coefficient between the sediment's solid matrix and pore water. The contaminant mass flux can then be converted to some average concentration above the cap as follows:

$$\bar{C}_{aw} = (F_{ss} \times T_r)/H_{wc} \quad (6)$$

where T_r is the renewal time (or turnover time) of the waters at the disposal site and H_{wc} is the height of the water column, or, if an even more conservative estimate is sought, some other smaller height for the affected area above the cap. The screening rule then consists of a comparison between the average concentration calculated by Eq 6 and an allowable limit in ambient water.

Alternatively, the impact of the capped sediment can be assessed by calculating the breakthrough time of the allowable concentration limit through the cap. Soon after sediment and cap placement, transport

through the cap is driven by diffusion, advection caused by sediment pore water flowing out of the settling sediment, and mechanical dispersion. For the simplified calculations needed at this screening stage, the contribution of the outflowing pore water is only taken into account in mechanical dispersion, while advection is ignored. With these assumptions, and by considering transport through the cap material, the contaminant concentration in the water at the upper surface of the cap can be approximated as:

$$C_{aw}(H_c, t) = \frac{C_{pw}}{2} \operatorname{erfc} \frac{H_c}{2\sqrt{D_{hd}t}} \quad (7)$$

where C_{pw} is the contaminant concentration at the sediment-cap interface, while D_{hd} is the coefficient of hydrodynamic dispersion, calculated as:

$$D_{hd} = D_e + \alpha_L v_c \quad (8)$$

where α_L is dispersivity in the cap material and v_c is the average advection velocity due to compression of the settling sediment. Various expressions exist that correlate α_L to the scale of transport domain [23], including the expression below suggested for transport calculations through sediment caps [24]:

$$\alpha_L = 0.0175 H_c^{1.46} \quad (9)$$

Finally, the average advection velocity can be estimated as:

$$v_c = (S_t/T_s)/n_c \quad (10)$$

where S_t is the total settlement of the sediment, T_s is the time required for most of the settlement to take place, and n_c is the porosity of the cap. According to the screening rule involving Eq 7, capping would be a feasible disposal option for the length of time before the allowable concentration reaches the upper surface of the cap.

The screening criteria based on Eqs 6 and 7 are conservative, since both solutions assume a continuous contaminant source of constant strength. (Accordingly, they cannot be relied upon for long-term estimates, when the contaminant mass is reduced significantly.) This observation has one exception, however: soon after cap placement (within a few months in examples provided by Alshawabkeh et al. [25]) the ignored advective transport component may provide a significant transient concentration peak, higher than the approximations given by Eqs 6 and 7.

Management Framework Overview

The framework used in the screening process is shown in Fig. 1 and, as already mentioned, evaluates disposal options in two stages. Stage I (Fig. 1(a)) assesses the feasibility of open-water disposal. It first evaluates the chemical characterization of collected sediment samples, in terms of their total contaminant concentration in the sediment. Results are compared herein to the range of values proposed by the Dutch Ministry of Housing, Spatial Planning and the Environment [26], which sets a lower desirable target level and a higher intervention level, above which action must be undertaken. Other agencies also have adopted the logic of a concentration range delimited by a lower-concentration desired limit and a higher-concentration cleanup level (e.g., Ref. [27]). If concentration values are lower than target levels, then open-water disposal is feasible. If concentration values are higher than intervention levels, there is a potential toxicity risk and, hence, confined disposal methods have to be considered. In no clear-cut cases, e.g., when concentration values are between target and intervention levels, the results from toxicity or bioavailability tests are considered as supporting evidence for the management decision, together with grain-size distribution data.

Although the framework is developed primarily for the disposal of dredged sediments, it can also provide an indication for whether further characterization of sediments is needed for possible environmental remediation. This is feasible because the logic of the proposed methodology is consistent with existing decision-making frameworks for sediment remediation, which typically evaluate the two lines of evidence considered herein, sediment chemistry and sediment toxicology, as well as a third line of evidence, the structure of the benthic community [28,29]. Following this logic, it is proposed that further sediment characterization is not advised simply if intervention values are exceeded, but only if, in addition, the potential for toxicity is identified.

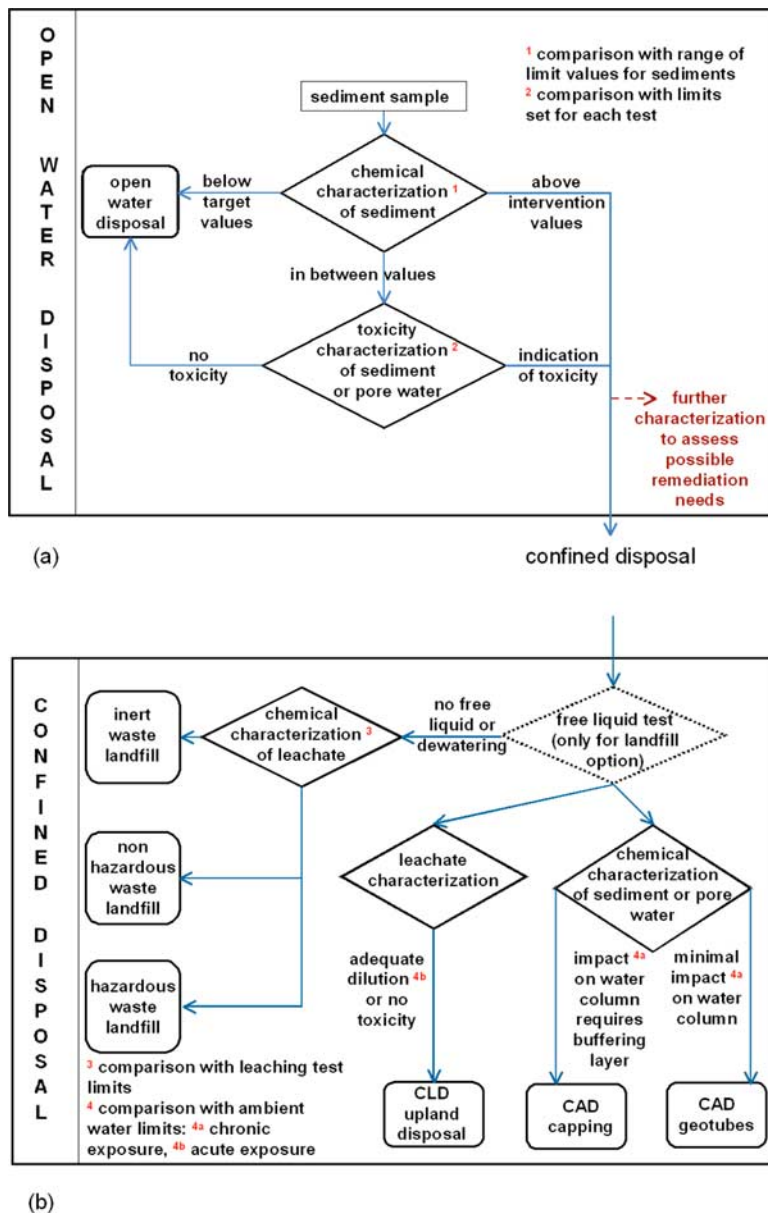


FIG. 1—Screening for (a) open-water and (b) confined disposal of dredged sediments.

Stage II (Fig. 1(b)) evaluates the three types of confined disposal considered: (1) confined aquatic disposal (CAD) in underwater enclosures provided by geocontainers or caps, (2) confined land disposal (CLD) in facilities close to the shore, and (3) landfills. The evaluation of subaqueous disposal requires chemical characterization of the sediment (Eqs 1 and 3) or the pore water (Eqs 6 and 7), depending on the screening rule used.

For disposal in a geocontainer, the concentrations given by Eqs 1 and 3 are compared to the Criterion Continuous Concentration (CCC) provided by USEPA as an estimate of the highest contaminant concentration in surface water to which an aquatic community can be exposed indefinitely (chronic exposure) without resulting in an unacceptable effect [30]. For capping, these same values are compared to the concentration given by Eq 6 or, alternatively, the breakthrough time of these values determines the useful life of the cap (Eq 7).

In the case of upland disposal, as already mentioned, the impact of the disposed supernatant is assessed with the aid of leaching tests. Contaminant concentrations in the leachate, expressed as contaminant mass per leachate volume, are compared with ambient water standards, by means of the dilution required to meet these standards before supernatant is discharged to the environment. If the calculated dilution is unattainable, then such a discharge will not be feasible. The limit value for each contaminant considered herein is the Criterion Maximum Concentration (CMC) provided by USEPA as an estimate of the highest

TABLE 1—Physical characteristics of sediment samples.

Station	Total Dry Solids (DS) (M_s/M_t , g/g)	Volatile Solids (% DS)	Fine Material (%) <74 μm
<i>Piraeus Harbor</i>			
PH-1	0.5101	12.03 %	81
PH-2	0.5747	7.34 %	46
PH-4	0.5812	8.07 %	38
PH-5	0.5830	7.02 %	... ^a
PH-7	0.5596	16.65 %	73
<i>Indiana Harbor</i>			
IH	0.5200	17.04 %	29

^aNot measured.

concentration in ambient water to which an aquatic community can be exposed briefly (acute exposure) without resulting in an unacceptable effect [30]. Toxicity testing of the leachate provides additional evidence of its potential risk to marine and fresh-water organisms [11].

For the landfill disposal option, contaminant concentrations in the leachate, expressed as contaminant mass per mass of dry solids, are compared to limit values set in Council Decision 2003/33/EC [13]. Dredged material is then characterized as inert, nonhazardous or hazardous and can be disposed of to the relevant type of landfill. Depending on the relevant national regulations, specific leaching tests (e.g., TCLP) or tests that determine pertinent physical characteristics (e.g., presence or percentage of free liquids), or both, may be required to meet any additional disposal criteria.

Management Framework Application

The example application of the framework uses characterization data for five sediment samples obtained from the Port of Piraeus, Greece, and a bulk sample from Indiana Harbor, IL, USA. The data for Piraeus sediments have been reported previously [5], with the exception of Cd measurements in sediment samples, which were obtained for the present study. Sampling stations in Piraeus Harbor characterize different subareas, which are planned to be dredged in sequence. Station PH-1 is located in the most enclosed part of the harbor, from where most of passenger ships depart, which covers an area of 53 500 m². The sediment to be dredged amounts to 83 450 m³ and comprises almost 40 % of the total dredging volume. Dredging in this area is of first priority. Station PH-2 is located in a more open area of the harbor, where large cruise ships take their moorings. The volume to be dredged is 7.5 % of the total volume. Stations PH-4 and PH-5 represent two remaining subareas from the central part of the harbor. Dredging in these areas is of second priority, but will produce a significant amount of the dredged material, equal to 49.5 % of the total. Finally, Station PH-7 is located in the shipyard zone of the harbor. Dredging from this area amounts to 3 % of the total volume.

Physical characteristics of sediments are summarized in Table 1. Regarding beneficial use, sediment at locations PH-2 and PH-4 of Piraeus Harbor as well as Indiana Harbor sediment are possible candidates, all with fines content below 50 %.

Screening of open-water disposal is carried out with the characterization results included in Table 2, for all sediment samples and Table 3, for the pore water of selected samples. The mass of contaminants in pore water in Table 3 is expressed both as $\mu\text{g/L}$ (mass of contaminant per pore water volume, C_{pw}) and in $\mu\text{g/kg}$ (mass of contaminant per dry mass of the sediment sample containing the pore water); the latter enables comparison of concentration of contaminants in pore water to the concentration in sediment samples, C_t (given in Table 2). The comparison is expressed as a percentage of total mass in Table 3, indicated as “% of total.” It is evident from the Piraeus sediment results that an extremely small percentage of the total metal concentration, 0.63 % for the most soluble metal Ni, is bioavailable to microorganisms, resulting in the measured zero toxicity. The similarity of results, in terms of the percentage of total mass, obtained in both samples (Stations PH-2 and PH-4) permits the generalization of this conclusion to the other more contaminated samples from Piraeus Port. The percentages of the total metals present in the Indiana Harbor sediment pore water are even lower, indicating that the metals are strongly sorbed to the solid phase.

TABLE 2—Concentration of heavy metals, C_p , and limits relevant to open-water disposal.

Station	Cu	Pb	Zn	Ni	Cr	Cd	Hg	AVS	Σ SEM
	mg/kg							μ mol/g	
<i>Piraeus Harbor</i>									
PH-1	359.8 ^a	264.3	504.3	84.0	92.4	1.6	0.5	8.3 ^b	14.8 ^b
PH-2	199.6 ^a	147.2	387.7	52.0	67.4	0.74	0.48	6.6 ^b	7.8 ^b
PH-4	196.4 ^a	206.5	273.8	46.1	57.4	0.45	0.5	7.0	5.7
PH-5	161.1	362.2	730.2 ^a	64.5	71.4	0.56	0.75	7.6	4.7
PH-7	257.5 ^a	1628.8 ^a	768.8 ^a	46.1	59.0	1.74	10.2 ^a	5.8 ^b	8.4 ^b
<i>Indiana Harbor</i>									
IH	330.0 ^a	1000.0 ^a	8300.0 ^a	110.0	880.0 ^a	17.0 ^a	1.2	... ^c	... ^c
Target values ^d	36	85	140	35	100	0.8	0.3		
Intervention values ^d	190	530	720	210	380	12	10		

^aExceedance of intervention values.^b Σ SEM > AVS.^cNot measured.^dNetherlands Ministry of Housing Spatial Planning and the Environment [26].

Comparison of measured values and limits in Table 2 shows that for the Piraeus Port and Indiana Harbor sediments, target values are exceeded in all locations, for all metals measured, with the exception of chromium for Piraeus sediments. Intervention values are exceeded in all locations for one or more metals. Although toxicity data are not available for Indiana Harbor, based on the relatively higher total metal concentrations as compared to the intervention values, disposal of Indiana Harbor sediment in open lakes or harbors is an unacceptable choice. For Piraeus sediment, only in Stations PH-1, PH-2, and PH-7, Σ SEM was higher than AVS, indicating a potential risk of toxicity. Hence, sediments from Stations PH-4 and PH-5 can be disposed in open sea. Open-sea disposal also remains as an option at this stage for Station PH-2, because there is no indication of toxicity for its pore water (inhibition 0 % in Table 3) and its coarse fraction is sizeable (54 % from Table 1). Hence, the remaining disposal options of Stage II will be evaluated herein for Piraeus Stations PH-1 and PH-7 only.

Assessment of the data considered in Stage I also evaluates whether some sediments require further characterization, in order to determine the need for possible environmental remediation. Recall that intervention values were exceeded in all Piraeus Port stations for one or more metals (Table 2), while the AVS/SEM test indicated potential toxicity for the samples from Stations PH-1, PH-2, and PH-7. With the exception of PH-2, which tested negative for toxicity in pore water (Table 3), the remaining two locations are candidates for further characterization. On the contrary, the positive difference AVS– Σ SEM in Sta-

TABLE 3—Concentration and toxicity of heavy metals in pore water.

Station		Cu	Pb	Zn	Ni	Cr	Cd	Hg	Inhibition (%)
PH-2	C_{pw} , μ g/L ^a	46	349	50	373	<50	<10	<0.1	0
	μ g/kg ^a	40	330	43	325	<44	<8.5	<0.08	
	% of total ^b	0.02	0.17	0.01	0.63	<0.06	11.5 ^c	<0.02	
PH-4	C_{pw} , μ g/L ^a	39	583	110	287	<50	<10	<0.1	0
	μ g/kg ^a	34	510	97	250	<44	<8.5	<0.08	
	% of total ^b	0.017	0.25	0.01	0.60	<0.06	15.2 ^c	<0.02	
IH	C_{pw} , μ g/L ^a	<50	19	130	84	<20	<10	<7.5	... ^d
	μ g/kg ^a	<46	17.5	120	77.5	<18	<9	<7	
	% of total ^b	<0.01	0.002	0.001	0.07	<0.002	<0.05	<0.6	

^aPore water concentration is expressed in reference to pore water volume (μ g/L) and to dry mass of sediment sample containing the pore water (μ g/kg).^bProportion of the contaminant mass in pore water compared to the total contaminant mass in the sediment sample.^cCalculated value is not meaningful due to the high detection limit for Cd in water.^dNot measured.

TABLE 4—*Confined aquatic disposal in a 2-m wide geocontainer: concentrations of metals at a distance of 10 m from geocontainer's center, 50 years after sediment disposal and respective limit values.*

	Concentration at $r=10$ m from Geocontainer's Center, at $t=50y$				Concentration at $r=10$ m from Geocontainer's Center, at $t=50y$			
	Station PH-1		Station PH-7		A ^a	Station IH		B ^a
	Eq 1	Eq 3	Eq 1	Eq 3		Eq 1	Eq 3	
	$\mu\text{g/L}$							
Cu	0.003	0.016	0.002	0.012	3.1	0.003	0.014	9.0
Pb	0.002	0.012	0.014	0.077	8.1	0.008	0.042	2.5
Zn	0.004	0.023	0.006	0.035	81	0.064	0.352	120
Hg			8E-05	4E-04	0.94			
Cr						0.007	0.037	11 ^b
Cd						1E-04	7E-04	0.25

^aA=Chronic toxicity saltwater limits and B=Chronic toxicity freshwater limits [30].

^bValue for Cr(IV).

tions PH-4 and PH-5 indicate that there is no need for further characterization, despite the exceedance of the intervention values for copper (PH-4) and zinc (PH-5).

Screening of disposal in a geocontainer is carried out using Eqs 1 and 3. For both calculations, the key quantity is total contaminant mass in the dredged sediment. All required input parameters are listed in Table 7 in the Appendix. Calculations were performed for a geocontainer approximated as a half cylinder with a 2-m or a 4-m diameter. Table 4 includes results of calculations at $t=50y$ for a distance of $r=10$ m from the center of the 2-m wide geocontainer. The allowable limits are not exceeded for any contaminant of concern for at least 50y at a 10-m distance from the disposal location. However, closer to the geocontainer, e.g., at $r=2$ m, calculations show that the limits are exceeded as early as one year or less. These conclusions hold for the 4-m wide geocontainer as well. As already mentioned, the contaminant loads in Eqs 1 and 3 are upper bound estimates. Sequential extraction tests for Piraeus sediments gave a potentially bioavailable percentage of 85 % for Pb, the most problematic metal, 70 % for Cu and 52 % for Zn [31]. These percentages could be used to reduce the contaminant load at a subsequent evaluation stage. Comparisons of the results from Eqs 1 and 3 shows that their difference is about one order of magnitude for $r=10$ m, which is adequate at this screening stage (the difference goes up to one-and-a-half orders of magnitude for the 4-m wide geocontainer). However, the difference can become significant at smaller distances or for a significantly bigger geocontainer.

Screening of disposal under a cap is carried out using Eqs 6 and 7. For both calculations, the key quantity is pore water concentration. All required input parameters are listed in Table 8 in the Appendix. Calculations were first performed for a reference cap of 1-m thickness and then repeated for thicknesses necessary to provide adequate protection. Results from Eq 6 are included in Table 5. It is seen that for Station PH-1 and for all metals of concern, the reference 1-m cap gives concentrations lower than the limits. In fact, at this station a 0.7-m thick cap would be adequate for Pb, the most problematic contaminant. On the contrary, the elevated Pb concentrations of Station PH-7 cannot be met with a few-meter thick cap. Indiana Harbor would require a 1.3-m thick cap to bring the concentration of cadmium below its limit. The input parameters for Eq 7 are also included in Table 8 and the results are summarized in Table 5. For sediments from Stations PH-1 and PH-7, a 1-m thick cap is adequate for precluding breakthrough of limit-level concentrations of copper and zinc for ten years, whereas this is not the case for lead. In order to ensure a ten-year useful life of a cap that will keep lead concentrations below ambient water limits, a thickness higher than 1 m is required. More specifically, the calculated required thickness is 1.5 m and 1.7 m for Stations PH-1 and PH-7, respectively. Similarly for the Indiana Harbor sediments, a cap thickness of 1.2 m is required to ensure breakthrough time less than ten years for all metals of concern. Comparison of the two screening approaches shows that the long-term consideration embodied in Eq 6, as well as the conservatism built in its application, imposes increased protection requirements for the capped sediments.

TABLE 5—Confined aquatic disposal under 1- to 1.7-m thick caps: average concentrations of metals in water column, respective limits and breakthrough times.

Metal, Cap thickness	Station	Average Concentration in Water Column Eq 6				Breakthrough time of Limit Value (A ^a or B ^a) Eq 7			
		μg/L				Years			
		Station PH-1	Station PH-7	A ^a	Station IH	B ^a	Metal, Cap thickness	Station PH-1	Station PH-7
Cu, 1 m	0.6	0.4	3.1	1.6	9.0	Cu, 1 m	10.2	12.5	37
Pb, 1 m	5.5	33.8 ^b	8.1	0.6	2.5	Pb, 1 m	6 ^c	3.7 ^c	25
						Pb, 1.5 m	10.2		
						Pb, 1.7 m		10.7	
Zn, 1 m	1.0	1.5	81	4	120	Zn, 1 m	>100	>100	>100
Cr, 1 m				0.6	11 ^d	Cr, 1 m			100
Cd, 1 m				0.32 ^b	0.25	Cd, 1 m			8.1 ^c
						Cd, 1.2 m			11.5

^aA=Chronic toxicity saltwater limits and B=Chronic toxicity freshwater limits [30].

^bExceedance of ambient water limits.

^cBreakthrough time (for 1-m thick reference cap) <10y.

^dValue for Cr(IV).

For disposal of Piraeus sediment in a confined upland facility, leachate does not conform to saltwater limit values for Cu, Pb, Zn, and Ni, which require a dilution higher than 1 (see Table 6). Moreover, the sample from Station PH-7 requires a very high dilution, 31 to 1, to meet the limit for copper and, therefore, is excluded from this management option. On the contrary, for Station PH-1, the limit value for copper can be met with a maximum dilution of 14 to 1, which can easily be achieved by discharging supernatant through a short outfall pipe. This dilution will also minimize the toxicity of the leachate (34.2 % in Table 6). For Indiana Harbor sediment, no dilution is required and hence disposal in a confined upland facility is a possible option.

Finally, disposal in landfills was proven feasible for Piraeus sediment from all stations and Indiana Harbor sediment evaluated. Piraeus samples conformed to limit values for nonhazardous wastes (see Table 6) and, as a result, dredged material can be disposed of in a nonhazardous waste landfill. Based on the leaching test results, Indiana sediment is classified as inert material. It is interesting to note that Indiana Harbor sediment was also categorized as nontoxic based on TCLP test.

TABLE 6—Confined disposal: leaching test results and respective limit values pertaining to confined upland facilities and landfills.

	Disposal to Confined Upland Facility					Disposal to Landfill				
	Concentration (μg/L)				Range of Required Dilution ^b	Concentration (mg / kg)				
	Station PH-1	Station PH-7	Station IH	Limit A/B ^a		Station PH-1	Station PH-7	Station IH	Limit C ^a	Limit D ^a
Cu	45	146 ^c	<10	4.8/13	31–1	0.09	0.13	<0.02	0.9	25
Pb	501	440	4.1	210/65	2.4–1	1.0	0.5	0.01	0.2	5
Zn	141	88	29	90/120	1.6–1	0.3	0.2	0.06	2	25
Ni	229	175	57	74/470	3–1	0.46	0.39	0.11	0.2	5
Cr	<50	<50	<4	1100/16	1–1	<0.1	<0.1	<0.01	0.2	4
Cd	<100	<100	<2	40/2.0	1–1	<0.2	<0.2	<0.01	0.03	0.6
Hg	0.1	0.1	<0.25	1.8/1.4	1–1	<0.2 ^e	<0.2 ^e	<0.5 ^e	3 ^e	50 ^e
Inhibition %	34.2	15.1	... ^d							

^aA/B=Acute toxicity saltwater/freshwater limits [30], C=Inert material limits, and D=nonhazardous material limits [13].

^bMaximum dilution required to meet limit values for the most contaminated sample (1=no dilution needed).

^cConcentration requiring unattainable dilution.

^dNot measured.

^eConcentration in μg/kg.

Conclusions

This paper presented a screening framework for the evaluation of alternative disposal methods for dredged sediments (open-water disposal, confined aquatic disposal in geocontainers and under caps, confined land disposal in near-shore facilities and landfills) and its application for sediments from Piraeus Harbor, Greece, and Indiana Harbor, IL, USA. The application of the methodology shows that, already at the screening level, the framework discriminates well among locations. More specifically, the following conclusions can be drawn. (1) Open-water disposal is possible for the relatively less contaminated sediments of Piraeus Harbor (subareas characterized by sampling Stations PH-2, PH-4, PH-5). In terms of volume, this means that almost 60 % of dredged material can be disposed in this way. (2) Geocontainer disposal would be possible for Indiana Harbor and the more contaminated sediments of Piraeus Port (subareas corresponding to Stations PH-1 and PH-7) only if it is possible to exceed limits within a short distance (less than 10 m) from a 2-m or a 4-m wide geocontainer. Otherwise, geocontainer disposal can only be used as an interim measure, in the short amount of time it takes for limit-level concentrations to break-through at the geocontainer wall. (3) For a ten-year useful life, capping is possible for Stations PH-1 and PH-7 as well as IH, for cap thickness of 1.5-m, 1.7-m, or 1.2-m thick, respectively. However, as a long-term solution, capping does not appear to be viable for Station PH-7. (4) Confined land disposal is possible for sediments from Indiana Harbor and from Piraeus Harbor Station PH-1 but not from Station PH-7. (5) Landfill disposal is possible for all.

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Appendix

Tabulated values used in calculations with Eqs 1, 3, 6, and 7.

TABLE 7—Confined aquatic disposal in geocontainers: input parameters used in calculations with Eqs 1 and 3 (contaminant-specific values are provided for Cu at Station PH-1).

Parameter Number	Input Parameter (Parameter Source)	Value
1	Dry solids content, M_s/M_t (Table 1)	0.55
2	Specific gravity of sediment solids (assumed)	2.55
3	Sediment porosity ^a , n_s (calculated from parameters No. 1,2)	0.67
4	Sediment dry density ^a , ρ_d (calculated from parameters No. 1,2)	0.82 kg/L
5	Contaminant concentration per solid sediment mass, $C_{t,Cu}$ (Table 2)	359.8 mg/kg
6	Contaminant concentration per total sediment volume, $C_{v,Cu}$ (calculated from Eq 2)	295 g/m ³
7	Geocontainer volume per unit length (2-m diameter)	1.57 m ³ /m
8 ^b	Contaminant mass per geocontainer unit length, M_{Cu} (calculated from parameters No. 6,7)	463 g/m
9 ^b	Diffusion coefficient (all metals), D (assumed)	10 ⁻⁹ m ² /s
10 ^c	Initial concentration in geocontainer, $C_{o,Cu}$ ($C_o=C_v$)	295 g/m ³

^aFor Indiana Harbor, $n_s=0.7$ and $\rho_d=0.76$ kg/L.

^bRequired only for Eq 1.

^cRequired only for Eq 3.

TABLE 8—*Confined aquatic disposal under caps: input parameters used in calculations with Eqs 6 and 7 (contaminant-specific values are provided for Cu).*

Parameter Number	Input Parameters (Parameter Source)	Value
1 ^{a,b}	Cap porosity, n_c (assumed)	0.5
2 ^{a,b}	Diffusion coefficient, D (Table 7)	10^{-9} m ² /s
3 ^{a,b}	Effective diffusion coefficient, D_e ($D_e = n_c \times D$)	0.5×10^{-9} m ² /s
4 ^a	Average contaminant concentration in water column, \bar{C}_{aw} (assumed)	0
5 ^{a,b}	Sediment porosity, n_s (Table 7)	0.67
6 ^{a,b}	Sediment dry density, ρ_d (Table 7)	0.82 kg/L
7 ^{a,b}	Partition coefficient (Piraeus Harbor), $K_{p,Cu}$ (assumed equal to the average from Stations PH-2 and PH-4)	4660 L/kg
8 ^{a,b}	Pore water concentration, $C_{pw,Cu}$ Piraeus Harbor (calculated from Parameters No. 5,6,7 using Eq 5) Indiana Harbor (Table 3)	77 μ g/L 50 μ g/L
9 ^a	Renewal time, T_r Piraeus Harbor Indiana Harbor	5y 20y
10 ^a	Height of water column, H_{wc} (conservative assumption)	10 m
11 ^b	Total sediment settlement, S_t (assumed)	0.3 m
12 ^b	Time required for most settlement to take place (assumed)	0.3y
13 ^b	Average advection velocity, v_c (calculated from Eq 10)	6.3×10^{-8} m/s
14 ^b	Dispersivity, α_L (calculated from Eq 9 for $H_c = 1$ m)	0.0175 m
15 ^b	Coefficient of hydrodynamic dispersion, D_{hd} (calculated from Eq 8) ^c	0.5×10^{-9} m ² /s

^aRequired for Eq 6.^bRequired for Eq 7.^cUsing values in this table, calculated breakthrough times are on the order of years, i.e., much bigger than the time period within which increased flow is expected due to settlement-related advection. As a result, it was deemed more relevant to take into account only the diffusive component of transport (i.e., $D_{hd} = D_e$).

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