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Land disposal options of contaminated dredged material

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Contents

1. Introduction

The problems associated with contaminated dredged material disposal (basically lack of disposal sites and potential adverse environmental impacts) have become major issues in many industrialized and developing countries.

The sediment removed for environmental reasons is obviously contaminated, but in many cases the sediment removed during normal maintenance dredging of waterways and harbours also contains a wide range of potentially toxic substances. In the Netherlands for instance, out of the 50 million m^3 of dredged material produced annually during normal maintenance dredging work 20 million m^3 is contaminated to such a degree that its dispersion into the environment without measures to impede contaminant release is unacceptable (Vellinga, 1989).

This paper gives a review of the disposal and treatment options currently in use or considered to have the potential for practical use in the near future with an emphasis on land disposal. It briefly discusses the main contaminant release pathways, the governing processes and the state-of-the-art methodology, used to assess potential environmental impacts.

2. Characteristics of dredged material

Sediment is a combination of fine grained minerals (clay, silt, sand) and organic (detritus) materials. Due to its chemical and physical properties it adsorbs a wide variety of contaminants introduced into the water system.

There are different ways to classify contaminants occurring in dredged material. One system proposed by Malherbe (1989) is as follows:

- Based upon persistency:
  - indefinitely persistent (heavy metals, ...);
  - highly persistent (PCBs, chlorinated hydrocarbons);
  - slightly persistent (some petrochemical hydrocarbons).

- Based upon toxicity:
  - acute toxicity;
  - chronic toxicity;
  - selective toxicity (toxicity to certain organisms, organs)

- Based upon chemical composition:
  - heavy metals (Cd, Hg, etc...);
  - anorganic compounds (chlorides, cyanids,..);
  - organic compounds
    - organo-halogenic compounds (pesticides, PCBs...);
    - poliaromatic hydrocarbons (PAHs);

- aromatic hydrocarbons (benzene..);
- organophosphor-pesticides;
- organo-metallo compounds;

radio-isotopes.

Chemical analysis alone cannot provide a sound basis for the evaluation of the degree of contamination of sediments in order to assess the potential environmental impact and decide which disposal criteria are applicable. Amongst the other reasons, the difference between the accuracy of chemical analysis test procedures can be mentioned. Very important factor is that no chemical analysis test procedure can reproduce the physico-chemical conditions which determine contaminant mobility. Chemical analysis does not reflect the potential environmental impact in the food-chain and cannot give any indication about the bioavalability or eco-toxicology of the contaminants. In section 4.4 some approaches to overcome this problem are described.

3. Management alternatives

The possible methods of processing contaminated sediments can be grouped into three main categories:

- Disposal: Dredged material and associated contaminants are contained within the disposal site.

- Disposal and rehandling: Dredged material is held for a period at a temporary disposal site and later either removed to another site for ultimate disposal or after introducing new precautionary measures or treatment methods based on the technical and scientific knowledge gained in the meantime, kept at the initial site. This solution is used in the case of the Papegaaiebek in Rotterdam, where the permit for disposal is only temporary. Before 1997 a plan must be made to provide a permanent solution. (Vellinga, 1988).

- Treatment: Dredged material is modified physically, chemically, or biologically to reduce its volume and/or its toxicity, mobility or to process it into sediment products.

4. Disposal options

Basically there are two main disposal options:

- Disposal in aquatic environments, like rivers, lakes, estuaries or oceans. Several special discharge methods are possible: e.g., discharging to form "mounds" on the seabed, disposal in pits, depressions, confined disposal with above-water or under-water dams etc. *These methods are usually used in combination with capping by clean material.

- Disposal on land. Again several possibilities are available (some are shown in *Figure 1*): disposal above the ground surface, disposal in dry pits, in water filled pits, in pits partially filled with water etc. Land disposal do not always mean dry disposal.

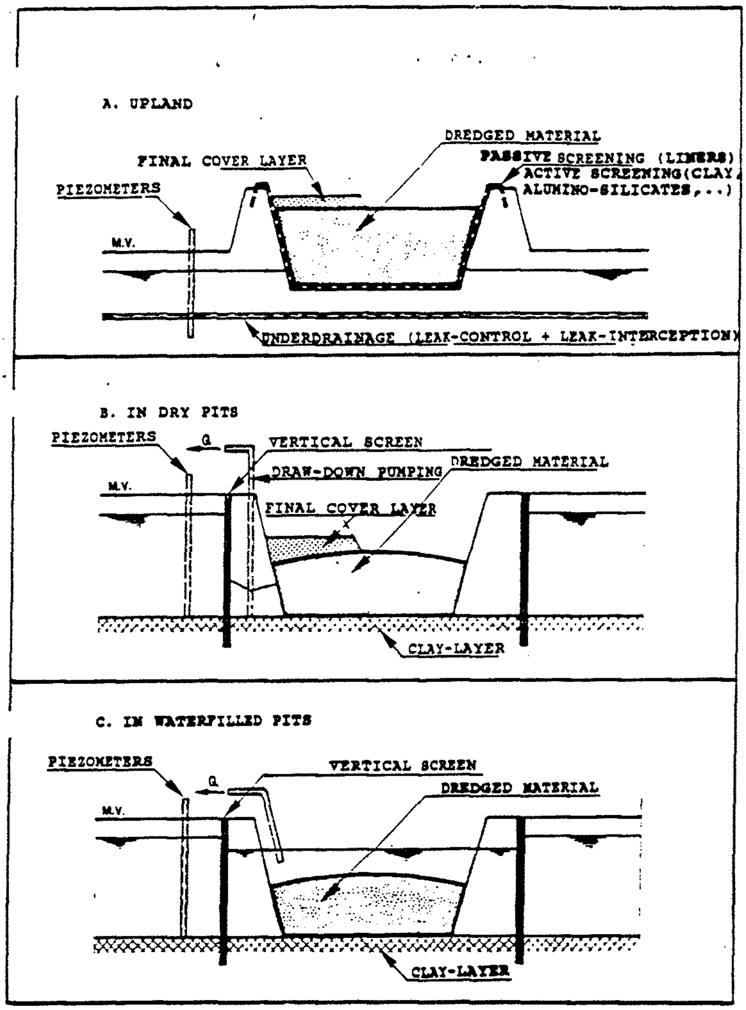

Figure 1
Options for Land Disposal (Malherbe, 1989)

Obviously in each case it is assumed that appropriate protective measures[1] impede the release of contaminants.

With regard to the dominant physico-chemical characteristics controlling contaminant mobility, and thus the potential for contaminated related impacts, distinct disposal environments can be distinguished (Lee, 1991). In dry unsaturated environments oxidising conditions and possibly low pH are characteristic, enhancing the mobility of heavy metals. In totally saturated environments reducing conditions prevail, with near neutral pH, which do not favour the release of heavy metals. In practice very often, depending on variations in the level of the water table, the combination of these two environments occur.

From the above it clearly follows that for several aspects no clear distinction can be made between aquatic and land disposal.

At this point attention has to be drawn to some confusion with regard to the terminology used in the literature. In many of the U.S. Army Corps Engineers (USACE) documents the term upland disposal refers to placement under dry, unsaturated conditions. Very often in the literature the term upland disposal is used for any form of land disposal. In this paper, the subsequent use of the term upland disposal is consistent with the USACE terminology.

A major consideration in evaluating the above options for the disposal of contaminated sediments is to assess the environmental acceptability of the proposed alternatives. This incorporates two main steps:

- To assess the potential contaminant release with regard to all relevant pathways.
- To interpret the results with regard to the existing standards and guidelines.

4.1 Pathways of contaminant release

When assessing contaminant release in general, the following routes of transport have to be taken into account (Lee et al., 1991; Loxham et al., 1988):

- Wind erosion of dry material and transport of adsorbed or free phase contaminants.
- Release of contaminants in the effluent during disposal operations in either dissolved or suspended particulate form.
- Surface runoff of contaminants in either dissolved or suspended form following disposal.
- Leaching into the ground water and surface waters.

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[1] The different technical solutions are discussed in chapter 4.4

- Direct uptake by plants (bioavailability), followed by indirect animal uptake from feeding on vegetation (bioaccumulation).
- Direct animal uptake (bioavailability), followed by possible indirect uptake by predators (bioaccumulation).
- Gaseous or volatile emissions during and after placement of dredged material.

The amount of contaminants released via these pathways is determined by the following factors:

- The type and amount of contaminants present in the sediment.
- The prevailing physico-chemical characteristics affecting the mobility of contaminants.
- The governing processes.

4.2 Mobility of contaminants

Because of their different physico-chemical characteristics the heavy metals, the organic micropollutants and the mineral oil behave in a completely different manner.

Heavy metals
Under anoxic conditions the solubility of most of the heavy metals is rather low. Usually they occur as metallic sulphide precipitates, thus the proportion of the dissolved fraction in the pore water is relatively low.

In an oxidising environment more soluble forms of metal oxides occur. The sulfide compounds are oxidized forming more soluble sulfates. Depending on the buffer capacity of the system, the pH usually drops owing to the oxidation processes, and this enhances metal solubility.

Organic micropollutants
In the case of organic micropollutants (and also arsenic) the availability of oxygen is irrelevant. For these substances the adsorptive characteristics determine the partitioning between the dissolved and adsorbed phase.

Mineral oil
In the case of mineral oil the adsorptive characteristics are again significant. In dredged material the mineral oil consists specifically of aromatic and aliphatic hydrogen compounds with high molecular weight. These compounds strongly adsorb onto the organic matter in the dredged material, as a result of which they can be considered relatively immobile.

4.3 Relevant processes governing contaminant release[2]

Because of the essential differences between the physico-chemical conditions of saturated and unsaturated disposal environments, the processes are discussed separately for dredged material depots above and below the water tables.

The following three environmentally adverse processes occur:

- Mobilization of contaminants.
- Expulsion of water and advective transport.
- Diffusive and dispersive transport.

Mobilization of contaminants

Depot above the water table: As soon as the dredged material is relocated and exposed to the atmosphere, oxidation processes begin increasing the mobility of contaminants. The rate of these processes depends on the hydrological conditions (precipitation, evaporation, permeability), thickness of dredged material etc. The dissolved contaminants either move with the pore water (advective transport) or move through it driven by concentration gradients (diffusion). As long as the dredged material remains in reducing conditions there is little danger of contamination of the subsoil or ground water by most of the heavy metals (with the exception of arsenic and chromium).

Depot under the water table: Dredged material under saturated conditions remains in an anoxic state with a nearly neutral pH. The amount of contaminants escaping is similar to that from a depot above the ground water level in the consolidation phase.

Expulsion of water and advective transport

When dredged material is relocated undergoes sedimentation followed by consolidation. As a consequence pore water is forced up and out of the depot. This water, which may contain dissolved and particulate associated contaminants, is discharged from the site as effluent.

Depot above the water table: During the consolidation period no precipitation can penetrate into the dredged material. It disappears via surface runoff. When no more expulsion of water occurs, part of the precipitation can infiltrate into the depot and percolate through to the subsoil carrying contaminants with it. It can reach adjacent aquifers or may enter surface waters. Depending on the physico-chemical conditions in the upper layers the surface runoff quality may show big variations.

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2 The description of the processes is based on Laboyrie (1989), Loxham (1989) and Lee et al. (1991).

Depot below the water table: After the consolidation phase, ground water can flow through the depot. The amount of water which flows through the depot in this way depends on the ground water gradient and the permeability of the dredged material.

Diffusive and dispersive transport

Diffusion from a depot into the subsoil and into the adjacent aquifers occurs under the influence of concentration difference.

Depot above the water table: When there is sufficient capillary action diffusion may possibly play an important role in ground water contamination. It is surmised, but not demonstrated, that hydrophobic organic contaminants associate with naturally occurring dissolved organic carbon and thus can diffuse into ground water beneath a site (Lee et al, 1991). Further work is needed to substantiate this theory. The major factors affecting the diffusion flux are (Vellinga, 1989): concentration differences and contact surface between the depot and its environment; position of the water table in relation to the depot with regard to capillary rising; flow velocity and the thickness of the aquifer.

Depot below the water table: Under the influence of concentration differences between the pore water of the dredged material and the ground water, substances from the pore water will move by diffusion into the aquifer. The contaminants will be carried through the aquifer by slow flowing ground water, while as a result of diffusion and advective transport, the contaminants can be dispersed over long distances from the disposal site.

4.4 Methods in current use to predict contaminant release and to assess possible environmental effects

Loxham et al. (1988) describe a method known as the Source - Path - Target Methodology, a classical risk analysis technique, successfully applied to determine contaminant release via the pathways considered as relevant for each specific sites and to assist decision making in respect of necessary protection measures.

In this method, the material is considered as the Source of toxicological risk and is characterised by an emission and an emission strength. Targets in the surrounding environment are identified and characterised by a Maximum Allowable Concentration (MAC) level, and Pathways connecting the source and Targets are specified and characterised by dynamic parameters such as time dependent concentration development patterns (*Figure 2*).

In these terms the engineering design problem is to combine emission strengths and barriers along the pathway in such a way that in the considered time-scale, the MAC values of the target will not be exceeded. Four cases[3] have been analyzed by this methodology and the proposed procedures have been successfully implemented in three

[3] The four analyzed disposal sites are: 1) Onshore disposal (at Zelzate, Belgium), 2) Offshore disposal (Slufter depot, Rotterdam), 2) Sub-river bed disposal (1st Petroleum Harbour, Rotterdam), 4) Sub-sea bed disposal (Hamburg, BRD).

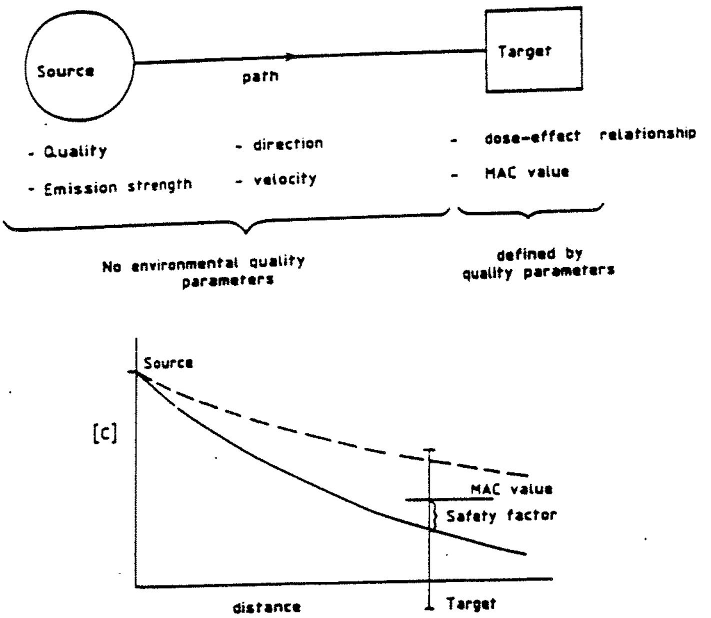

Figure 2
Schematic Representation of the Source - Path - Target Methodology (Loxham et al., 1988).

of these cases. The fourth is still under consideration.

In 1985, the USACE developed a management strategy for dredged material disposal alternatives, in the framework of which techniques predicting contaminant release have been elaborated (Francigues et al., 1985). These techniques have been in use since and their accuracy has been verified in a six year EPA/USACE Field Verification Program (FVP) (Dillon, 1988). The evaluative techniques which incorporate the predictions for effluents, surface runoffs, leachate, plant uptake and animal uptake[4] form an essential part of the expanded and refined strategy currently under the joint development of the Corps and EPA (Lee et al., 1991). In the FVP, the relative comparisons of the predictive techniques were expressed by qualitative ratings of good, fair and poor. A rating of good was given to the technique which can be reliably applied in routine evaluations for most parameters owing to the sufficient correspondence between tests and field results. If the correspondence was not consistently good for most parameters, the technique was rated fair. Poor ratings indicate little or no correspondence between tests and field data and as a consequence at present the technique cannot be relied upon for routine applications.

4 In the case of aquatic disposal obviously other factors are investigated. For details see Lee et al. (1991).

The USACE emphasizes that any test protocol to predict contaminant mobility should account for the physico-chemical changes occurring in the dredged material when relocated in a specific disposal environment.

Predicting effluent quality:
The quality of the effluent during the disposal and consolidation process is predicted from the combined results of modified elutriate tests and column settling tests. Detailed procedures for both tests are given by Palermo (1984). In the FVP, this technique was qualified as good (Dillon, 1988).

Predicting surface runoff quality:
As reported by Lee et al. (1991), at present there is no single simple test to predict runoff water quality. In a research project initiated in 1984, a rainfall simulator has been developed. Detailed description of the procedure can be found in Skogerboe et al., (1987). According to Dillon (1988), the mean concentrations of most contaminants predicted by the laboratory test were in fair agreement with the values observed in surface runoff tests conducted on the upland site after it was filled and had dried to typical upland conditions. The procedure proved to be fairly useable for predisposal evaluations of proposed upland disposal and for post-disposal monitoring applications (Dillon, 1988). It has been used on a number of dredged materials from Oakland Harbour, California and New Bedford Harbour, Massachusetts, to Indiana Harbour, Indiana.

Predicting leachate quality:
As reported by Lee et al. (1991), at present there is no routinely applied testing procedure to predict leachate quality from dredged material disposal sites. Based on the principles of the mass transport theory, Hill et al. (1988) developed a mathematical model describing the processes governing leaching. Based on a state-of-the-art review of leaching procedures, they recommended a sequential batch leaching procedure for obtaining the coefficients needed in the mass transport equation. These experimental procedures have been used to evaluate the potential impacts of confined disposal of dredged material from several harbours in the USA. In a workshop held by Louisiana State University in 1988, the experts were of the opinion that the research conducted to date was good and generally validated the basic technical approaches of Hill et al. (1988). However, the consensus was, that much research remains to be done before leachate tests will be available for routine use (Lee et al. 1991).

Predicting plant uptake:
The plant bioassay test developed under the USACE Long-term Effects of Dredging Operations Program (for detailed description see Lee et al., 1991) was not in agreement with the bioaccumulation observed in the field for upland disposal during the FVP. In the relative comparisons of the techniques it was qualified as poor. As Dillon reports (1989), the variability in these predictions limits the utility of the upland plant bioaccumulation technique for predisposal evaluations.

Predicting animal uptake:
The utility for predisposal evaluations of the recommended test protocol, which is a modified version of a procedure developed for the European Economic Commission for determining the hazardous nature of manufactured chemicals prior to approval for sale

in the European Common Market, could not be determined during the FVP. The reason for this was the poor survival of earthworms under upland conditions (Dillon, 1988). In their report Lee et al. (1991), were of the opinion that the test should be able to indicate potential environmental effects of dredged material disposal in upland environment.

Summarizing, it can be stated that while the effluent and surface water quality predictive methods proved to be useful for predisposal evaluation of dredged material intended for upland disposal, the leachate quality, the plant and animal uptake prediction techniques require further development.

It is worth mentioning, that a qualitative comparison of the effects of placement of the same dredged material in upland, wetland, and aquatic environments revealed that upland disposal produced the greatest and most persistent impacts (Dillon, 1988).

With regard to the evaluation of different techniques in the FVP, it has to be emphasized that the results cannot be extrapolated to all dredged material evaluation because of the specific conditions of FVP (Dillon, 1988).

4.5 Interpretation of contaminant release data

To date, the problem of establishing reference values against which the predicted values of contaminant release should be tested in order to avoid unacceptable degree of contamination of the soil or ground water, has not been solved satisfactorily. A variety of temporary solutions is used in the different countries.

In the Netherlands provisional reference values have been fixed for soils, ground water and underwater sediments. These values are based on the current ecotoxicological knowledge (V&W, 1989).

Another example is the previously mentioned decision making framework of the USACE. This framework utilizes the management strategy illustrated in *Figure 3* and incorporates the results of the suite of test procedures described above. In this paper no detailed description of the steps to be followed in using the flow charts can be given. However, an example is shown by the evaluation scheme of effluent water quality in *Figure 4.*

In Belgium a classification system is currently being worked out as an Environmental Impact Procedure, which can be considered as a decision making process. The environmental impacts are weighted against economic, social and political considerations (Kreps-Heyndrikx et al., 1988).

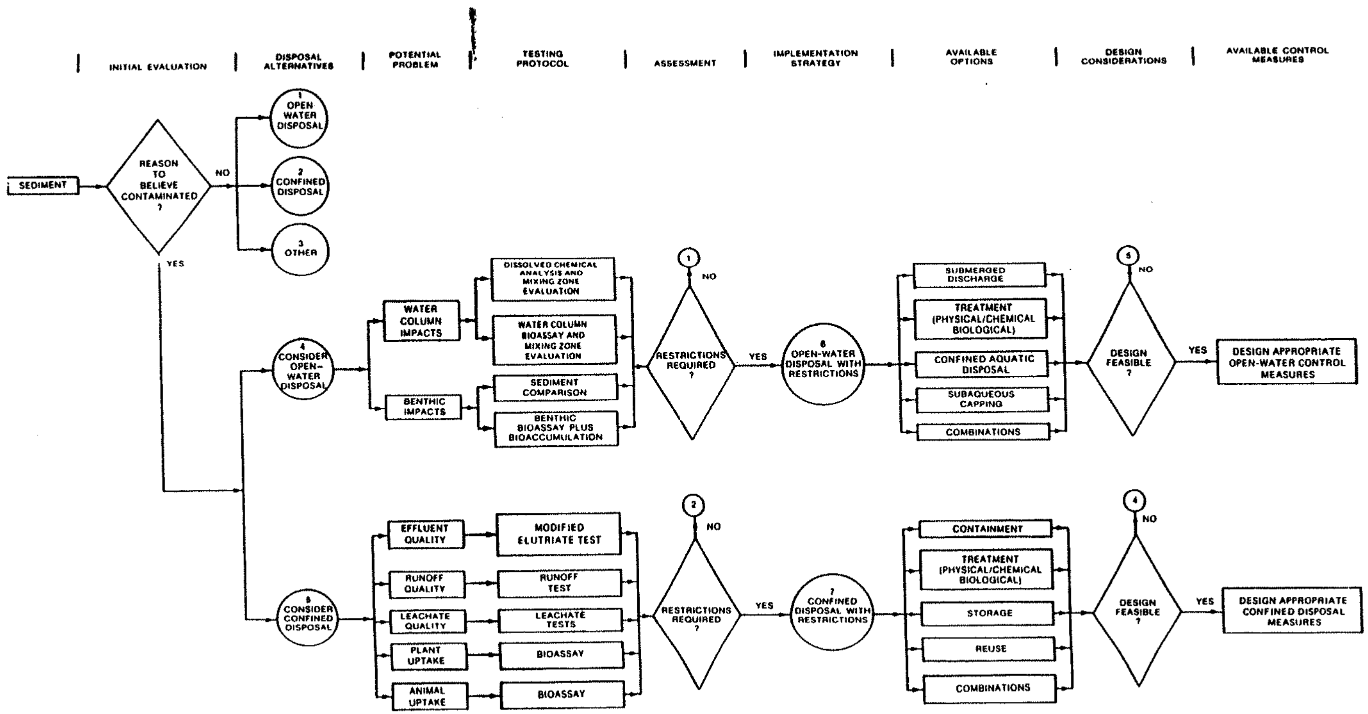

Figure 3
USACE Management Strategy Flowchart (Lee et al., 1991)

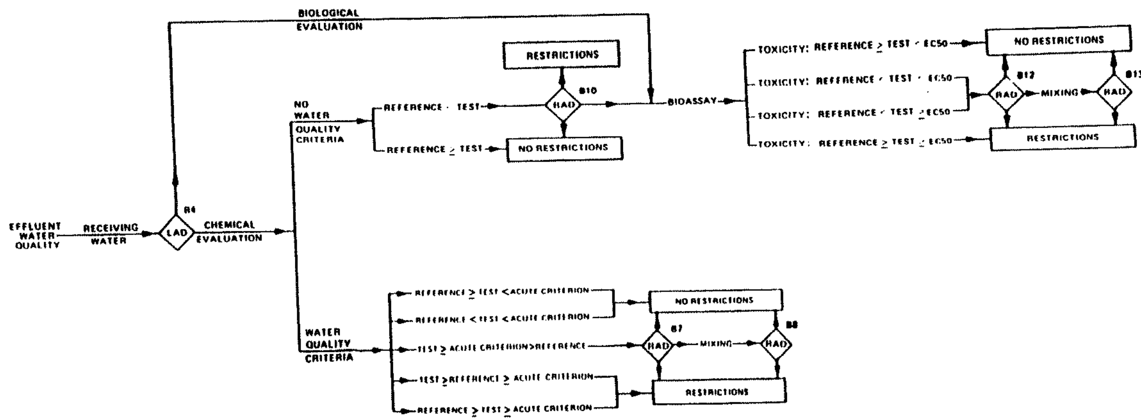

Figure 4
USACE Flowchart for Decision making for Effluent Water Quality (Lee et al., 1991)
RAD: Regional Administrative Decision

4.6 Technical measures to limit impacts

Based on the foregoing sections, the following control measures may be required to prevent unacceptable dispersion of contaminants from a disposal site (Lee et al., 1991):

- Effluent quality control during dredging operations.
- Runoff-water quality control after dredging operations.
- Leachate controls during and after dredging operations.
- Control of contaminant uptake by plants and animals during and after dredging operations.
- Control of atmospheric contaminants after dredging operations.

Before reviewing the control measures currently in use it has to be emphasized that they are very much site-specific. Special consideration of the physico-chemical characteristics of the dredged material and the disposal site is required to effectively design a disposal facility. In a number of cases only a combination of measures can provide satisfactory result.

The technical measures aimed at preventing contaminant dispersion fall into three main categories as follows:

- Site control measures: Insulation (lining, capping), placement of an adsorption layer, placement of a diffusion layer, hydrological control measures, underdraining etc.
- Effluent treatment.
- Leachate treatment.

4.6.1 Site control measures

With regard to insulation the importance of proper site selection cannot be emphasized enough. It can mitigate many contaminant mobilization problems. Proper site selection may reduce surface runoff and therefore contaminated runoff and contaminant release by flooding. Ground water contamination problems can be minimized through selection of a site with natural clay foundation rather than of a sandy area and by avoidance of aquifer recharge areas. If a favourable choice of location is not possible insulation measures (lining and/or capping) may be required.

Lining

Lining of the bottom and slopes of the site is performed to prevent leakage and seepage. Two main types of lining material can be used:

- Natural (or soil) liners: clay, peat.

- Artificial liners: impermeable sheeting, sand/bentonite layers.

For both types of liners the chemical compatibility (resistance) to the dredged material has to be tested.

Natural clean materials, such as clay and peat in particular are not only good liners that are relatively inert to chemical attack but will also act as a filter adsorbing many contaminants from the leachate. With regard to the fixing capacity for both heavy metals and organic micropollutants especially peat seems to be very effective due to its high organic carbon (humus) content (Veltman et al., 1988). The natural liners appear to be financially and technically attractive and they have another significant advantage over the synthetic foils and membranes widely used nowadays: namely, they are characterised by an unlimited life span. Laboratory scale and field research has shown that, depending on the type of contaminants, the thickness and composition of the insulation, it is in the order of hundreds to thousands of years (Veltman, et. al, 1988). With regard to clay, one disadvantageous property has to be mentioned: organic and inorganic acids and bases may dissolve a portion of the clay structure (Male et al., 1988).

A wide variety of synthetic liners is in use. They vary not only in physical and chemical properties but also in installation procedures, costs and chemical compatibility with waste fluids (e.g., PVC tends to be dissolved by chlorinated solvents). The liners range in thickness from 20 mil to 140 mil and are made from polymers of rubber, plastics such as PVC, polyolefins and thermoplastic elastomers (Male et al., 1988). Synthetic liners such as High Density Poly Ethylene (HDPE) are widely used as underseal but mostly only above water: sheeting underwater is considered to be scarcely feasible (Vellinga, 1989). The reason for this is, that most polymeric material will tend to swell when exposed to fluids, with disadvantageous consequences like increased permeability and potential for creep, loss of tensile and mechanical strength and elongation etc. Synthetic membranes are also subject to biological degradation (Male et al., 1988). The absolute impermeability of the sheetings used at present is temporary in nature (30-50 years) (Vellinga, 1989) therefore, using them can only be considered only as temporary solution. A comprehensive survey of synthetic liners is given by Male et al. (1988).

Sand/bentonite layers have the disadvantage that they are not absolutely impermeable. The process of ground water contamination is retarded but not prevented. A combination with other measures might provide advantages (Vellinga, 1989).

Capping

Surface capping is the placement of clean, low permeability material over the confined dredged material. It is performed for many of the reasons, for instance, to:

- Reduce wind erosion and dust emission.
- Reduce water erosion.
- Prevent acidification and to reduce dissolution.
- Reduce surface water infiltration and thereby leachate production.
- Planting selective vegetation to minimize contaminant uptake.
- Prevent contaminant uptake by plants and animals.

Various low permeable materials may be used like clays, admixtures (e.g., asphalt concrete, soil cement) and polymeric membranes. The application of artificial covers has the disadvantage that the gas which forms in the depot cannot escape and collects under the covering layer. There is a high risk of damage, unless a gas withdrawal system is made under the covering. Another disadvantage is the difficulty of detecting possible leaks (Vellinga, 1989). Typical final covers are composed of several layers.

The placement of an adsorption layer

A property of this layer is that contaminants moving out of the depot are fixed, immobilized in the layer that is they cannot disperse further into the subsoil or ground water. (See natural insulating layers as described above.) (Vellinga, 1988).

The placement of a diffusion impeding layer

A very permeable layer is placed beneath the depot to prevent capillary action in the ground water leading to diffusive transport. Naturally this measure is only worthwhile for disposal above water in combination with the use of a impermeable covering layer (Vellinga, 1988).

Hydrological control measures

A possible method is the influencing of the ground water flow levels in such a way that the dispersion of contaminants remains limited and/or that the ground water contaminated to an unacceptable degree is carried away by a drainage system, pump wells etc. An interesting solution which could be grouped into the category of hydrological control measures is provided by the solution used in the case of the Slufter depot in Rotterdam. Here a freshwater lens in the originally saline ground water prevents the dispersion of the contaminants into the surrounding environment. In the first instance, the freshwater lens would form as a result of the relatively low chloride content of the pore water that would percolate downward from the dredged material. This water would remain floating, as it were, above the ground water, which has a high chloride content (seawater). Under the ring dike, over a period of thirty years, the percolation of rainwater would form a similar freshwater lens. The contaminants from the dredged material would remain within hydrologically closed system of "freshwater" (Vellinga, 1988).

Underdrainage (Shafer, 1989)

Underdrainage and leachate collection is a dewatering method which may be used either individually or in conjunction with improved surface drainage. The collector pipes are placed in either a naturally occurring or artificially placed pervious layer prior to dredged material disposal. Free water in the dredged material migrates into the pervious underdrainage layer and is removed via the collector pipe systems. Usually three mechanisms are used for dewatering of dredged material:

- Gravity underdrainage: Downward flow of water from the dredged material into the underdrainage layer takes place by gravity.
- Vacuum-assisted underdrainage: This technique is similar to gravity underdrainage, but a partial vacuum is maintained in the underdrainage layer by vacuum pumping.
- Vacuum-assisted seepage consolidation: This technique combines the effects of seepage consolidation with those of an induced partial vacuum in the underdrainage layer.

4.6.2 Effluent and leachate treatment

Figure 5 summarizes the available treatment processes (Male et al., 1988) which are basically physico-chemical processes for waste water treatment. As Lee et al. report (1991) beyond chemical clarification, only limited data exist for treatment of dredged material disposal sites' effluents or leachates. In both cases the treatment can be carried out off-site or on-site.

5. Dredged material treatment options

Although, the treatment of contaminated dredged material to date is not considered as a feasible solution (for large quantities it is extremely expensive) the possibilities offered by this option cannot be left out of consideration. In fact, in several countries, where disposal of lightly or highly contaminated dredged material creates serious problems, at the beginning of the nineties research projects have been started with (or partly with) the objective to evaluate the utility of a number of technologies for dredged material treatment. As examples the following research projects can be mentioned:

- In the Netherlands, in 1983 research started, on the initiative of the Dutch Ministry of Housing Physical Planning and Environment and the Ministry of Transport and Public Works (van Dillen, 1989).

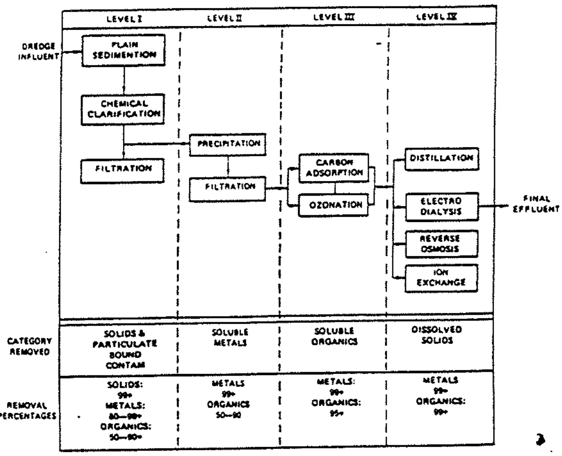

Figure 5
Schematic of Treatment Processes Showing Estimated Removal Efficiencies (Male et al., 1988)

- In the framework of The Great Lakes Clean-up Fund, Environment Canada the Contaminated Sediment Treatment Technology Program (CoSTTeP) was formed (Sherbin, 1992 ; Wardlaw, and Bucens 1992).

- USEPA initiated the Assessment and Remediation of Contaminated Sediments (ARCS) Program (Horvatin et al., 1992).

Each of these programmes began by conducting a comprehensive survey to identify technologies that looked promising for the treatment of contaminated sediment. The ARCS survey for instance identified over 250 technologies that either were being, or could be applied to contaminated sediment. These technologies included traditional methodologies utilized in the field of hazardous waste, as well as non-traditional methods currently being used in the mining and metal processing industries, petroleum industry, at sludge treatment plants and at industrial waste treatment facilities (Horvatin et al., 1992). Bench-scale tests of the selected technologies have been carried out and technology demonstration projects (pilot-scale) are under way to show how well these technologies can be expected to perform in a full-scale treatment system . To date four of the planned five ARCS pilot demonstrations have been initiated and completed: the first in the fall of 1991 at the Buffalo River (Horvatin et al., 1992). In the framework of the CoSTTeP to date one successful demonstration have been completed (Toronto's inner harbour) (Sherbin, 1992). The demonstrations are considered to be very useful in assessing the benefits and limitations of technologies in treating contaminated dredged material. The following section is mainly based on the reported findings of these programmes.

In general, the treatment of dredged material can be accomplished by:

- Volume reduction.

- Decontamination.

- Processing the sediment into products.

Vellinga (1988) gives a further division of the various techniques already in use or, in principle, having the potential of practical use *(Figure 6)*. Currently the technical and financial feasibility of a number of these techniques is being investigated on a laboratory and on semi-practical scale. With regard to the listed methods in *Figure 6* it has to be emphasized that it gives an optimistic view as far as the number of possibilities is concerned: in many cases further research related to the technical feasibility is needed. In a number of cases the financial feasibility of the solution is seriously questioned. In the following sections only those solutions which seem to be promising and are currently extensively investigated are briefly considered.

Volume reduction

As in many countries it is increasingly difficult to provide large enough disposal sites for contaminated material (Maekawa, 1992; Kroezen, 1992; Kreps-Heyndrikx, 1988), volume reduction before disposal or treatment is an essential task. This means reducing the quantity of the material that has to be disposed of by concentrating the pollutants in a minor part of the material. Volume reduction is accomplished by two methods:

- Separation of the coarse and fine fractions of the dredged material by hydrocyclones.
- Dewatering.

Initially it was thought that since many of the contaminants were bonded to the fine fraction the coarse sandier fraction was relatively clean. In fact, carbon stars may adhere to the sand adsorbing harmful substances like heavy metals and/or PAHs (Vellinga, 1992a). In such cases it is necessary to resort to further processing to clean the coarse fraction. The possible processes for sand treatment are listed in *Figure 6*.

Two main types of the dewatering processes are used:

- Natural dewatering. This requires more space, but is cheaper.

- Mechanical dewatering. Because there is a definite tendency to integrate the dewatering process into the dredging operation (Annokkee, 1989; Maekawa, 1992), the significance of mechanical dewatering is going to increase in the future. In the current Dutch dredging operations sieve belt presses and decanter centrifuges are commonly used (Kroezen, 1992). According to Kreps-Heyndrikx et al. (1988) the chamber filter presses provide the highest volume reduction. In order to improve

TECHNIQUE	I	II	III
Fraction separation			
- hydrocyclones and separators	X		
- horizontal discharge in a site above water in which separation into sand and silt fractions occurs	X		
- gravitational separation methods		X	
- flotation		X	
Dewatering			
- natural dewatering at the depot	X		
- the application of flocculant		X	
- preliminary dewatering using lamella separators	X		
- sieve belt presses	X		
- centrifuge		X	
- chamber filter-press	X		
- vacuum drum sieve		X	
- subsequent drying of solid matter	X		
Cleansing of the sand fraction			
- biological cleansing		X	
- flotation (froth separation)		X	
- steam stripping		X	
- washing with (dilute) hydrogen peroxide			X
- the up flowing of sand. with the addition of acids. alkalies or other chemicals		X	
- heat treatment (800C)		X	
Cleansing of the silt fraction			
- biological cleansing in the depot		X	
- biological cleansing in bioreactors			X
- chemical cleansing in the depot		X	
- steam stripping		X	
- extraction: mobilization and elimination of contaminants		X	
- bacterial leaching (acid extraction)			X
- electro-reclamation of heavy metals			X
- pyrolysis at 1200 C (without oxygen)			X
- incineration at a high temperature (800 C)		X	
- fluid bed incineration		X	
Immobilization			
- making granules or pellets		X	
- ceramic processing		X	
- addition of binding agents. hardening		X	
- pre-treatment by means of oxidation. reduction and precipitation. the introduction of additives and hardening.			X

I : In use for the treatment of dredged material
II : Used to a limited extent. in many cases only operational for the treatment of other matter.
III : Not in use.

Figure 6
Treatment Techniques for Contaminated Dredged Material (Vellinga, 1989)

the dewatering capacity, the addition of conditioning chemicals may be necessary. Analysis carried out on the filtrate and the filter cake showed that heavy metals were concentrated in the filter cake: the degree of concentration seems to be directly related to the filter pressure. This especially holds for copper, lead and zinc. This is an other big advantage of mechanical dewatering (Kreps-Heyndrikx et al. (1988).

The scenario for the treatment of dredged material developed in the Netherlands is shown in *Figure 7*. It consists of two main routes:

- Hydrocyclones separation and dewatering, route A-B.
- Hydrocyclone separation and decontamination, rout A-C.

With regard to the suggested sequence of separation and dewatering it has to be mentioned that Kreps-Heyndrikx et al. (1988) in their paper do not agree with this sequence: according to their experiences using hydrocyclones prior to mechanical dewatering causes a deterioration in the dewatering characteristics of the material and as a consequence no significant volume-reduction can be achieved.

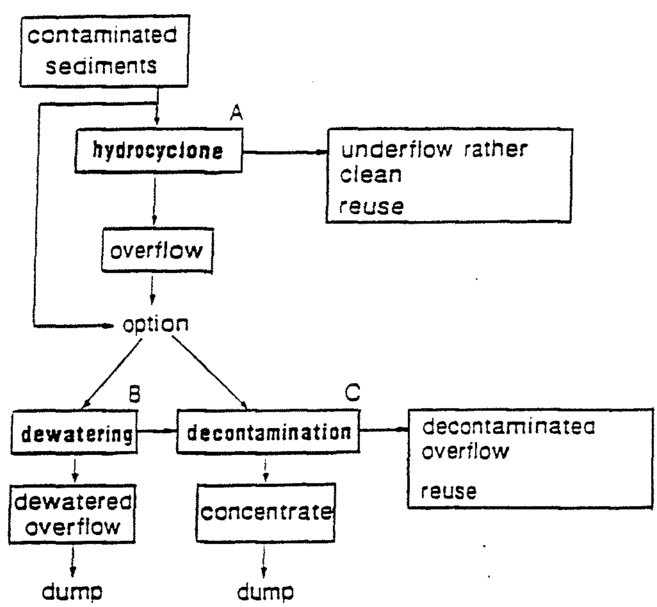

Figure 7
Scenario for the Treatment of Dredged Material (Annokkee, 1989)

Decontamination

The aim of decontamination is to separate the pollutants from the sediment particles or to destroy the pollutants to such an extent that the dredged material can be disposed of in an unconditional manner or under less expensive provisions. Possible application techniques can roughly be divided into two categories:

- Techniques for removing heavy metals.
- Techniques for removing organic contaminants.

As recently reported by Wardlaw and Bucens (1992) very few technologies claim to provide a universal solution ie. remediate sediment contaminated with metals and organic compounds. The CoSTTeP is not aware of any technologies that have been demonstrated to economically address all contaminants. Current thinking suggests that the most viable approach (both practically and economically) is the use of a sequence of processes, each of
which is dedicated to a contaminant type (Wardlaw and Bucens, 1992).

To date to following methods have been considered as viable for removing heavy metals from sediments:

- Acid leaching.
- Biological leaching by Thiobacillus.
- Steam stripping for mercury containing soils.

An example for acid leaching is the Metanetix process, which is the second element of the process train of the Toronto Harbour pilot-scale demonstration conducted under the auspices of the CoSTTeP. The Metanetix is capable of selective and non-selective extraction of heavy metals. It also provides the possibility of metal recovery (Wardlaw and Bucens, 1992).

Techniques for removal of organic contaminants can be divided into two groups:

- Concentration techniques: solvent extraction and washing.
- Destruction techniques: incineration and biodegradation.

To date of these techniques biodegradation and solvent extraction have been investigated on a bench scale in the Netherlands. Biodegradation is considered to be a technique with high practical potential for the following reasons (Annokkee, 1989):

- From an environmental point of view, the sediment is more acceptable product after biological treatment than after physicochemical or thermal treatment.

- It is expected to be less expensive than any other techniques.

However it has also some disadvantages (Wardlaw and Bucens, 1992; Annokkee, 1989):

- The long time usually required for treatment.
- The relatively high residual concentrations of contaminants, particularly in case of substances of high molecular weight and chlorinated hydrocarbons which are hardly biodegradable.

According to the Dutch experience with bench-scale investigations, solvent extraction can be used as an alternative technology (with high expenses involved) to overcome this latter difficulty. Another possible solution is represented by the SNC-Lavalin biochemical organic destruction process. (The process is an element of the process train of the previously mentioned the Toronto Harbour pilot-scale demonstration). In this process chemical oxidation is applied in order to oxidize organics of high molecular weight to species of lower molecular weight, which are more readily degraded biologically. Hence more rapid destruction occurs and lower residual contaminant levels are achieved (Wardlaw and Bucens, 1992).

Processing the sediment into products

Sediment can be processed (immobilized) in bricks, tiles or artificial gravel. Apart from the environmental impact, the success of this reuse depends on the social acceptance of the products manufactured from polluted sediment. In the Netherlands the level of this acceptance is rather low. On the other hand, because the standards are based on contaminant content and not on leachability, legally immobilization is not considered as quality improvement.

6. Conclusion

In spite of the recent developments in the field of dredged material treatment techniques the permanent storage of large quantities of polluted dredged material seems to be unavoidable. Further work is required to elaborate methodologies to safely predict potential contaminant release and consequent environmental impacts.

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Abbreviations

ARCS	Assessment and Remediation of Contaminated Sediments
CoSTTeP	Contaminated Sediment Treatment Technology Program
EPA	U.S. Environmental Protection Agency
FVP	Field Verification Program
HDPE	High Density Poly Ethylene
MAC	Maximum Allowable Concentration
PAHs	Polychlorinated Aromatic Hydrocarbons
PCBs	Polychlorinated Biphenils
RAD	Regional Administrative Decision
USACE	US Army Corps of Engineers

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