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Ecological effects of dumping of dredged sediments; options for management

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Abstract. Dredging and dumping of dredged sediments in estuarine and coastal waters may lead to increased turbidity and enhanced sediment deposition at dump sites. This mainly affects primary production by phytoplankton, performance of visual predators (e.g. fish, birds), and growth and survival of benthic organisms. This paper combines a compilation of literature information and results of additional experimental studies on the effect of enhanced concentrations of suspended matter (SPM) on growth of bivalve molluscs, and on survival of macro- and meiozoobenthos after dumping of dredged sediments. Furthermore, it focuses on non-toxic dredged sediments only.

Release of nutrients from dredged sediments did, so far, hardly influence estuarine phytoplankton production. Increased turbidity may affect dab as well as prey location by sandwich terns. Enhanced SPM-concentrations are unfavourable for young herring and smelt. Growth of filter-feeding bivalves may be impaired, especially at SPM-concentrations >250 mg/l. Estuarine nematodes can survive burial by 10 cm of dumped dredged sediment provided that its physical characteristics are similar to those of the original sediment. Sessile benthos organisms such as mussels and oysters can cope with sediment deposition of only 1-2 cm. Other macrozoobenthos can survive sediment deposition of 20-30 cm. Recovery of benthos at a dump site will occur if the interval between successive dumpings is sufficiently long.

Options for management of dumping of dredged sediments are described, relating to different locations of dump sites in estuarine and coastal waters, to different seasons, and to the actual use (area and frequency) of dump sites.

Keywords: Birds; Dredge spoil; Eelgrass; Ems Estuary; Fish; Macrozoobenthos; Meiofauna; SPM; Wadden Sea.

Abbreviation: SPM = Suspended matter.

Introduction

Shipping in Dutch coastal waters requires extensive maintenance dredging of navigational routes and harbours and deepening of shipping channels. Ports situated in estuaries (Westerschelde, Rotterdam waterway, Dutch Wadden Sea, Ems estuary) contribute with an annual transhipment of ca. 400 million tons. Of this total, the port of Rotterdam takes ca. 290 million ton, or 77 % (F. Otto et al. Int. rep. 1993; Min. Verkeer en Waterstaat, Int. rep. 1996).

Dredged sediments range from mud to silt. The dredged material is partly contaminated to such an extent that dumping is allowed only in contained sites, of which the so-called ÕSlufterÕ near Hook of Holland with a capacity of 120 $\times 10^{6}$ m^3 is the biggest (Vellinga pers. comm.). According to actual Dutch policy, only non- and low-contaminated dredged sediments are allowed to be dumped in the coastal and estuarine systems. In 1995, ca. 30 $\times 10^{6}$ m^3 of dredged sediments were dumped in Dutch estuarine and coastal waters, of which ca. 56% originated from the Rotterdam harbours, ca. 16% from the Western Scheldt, and 17% from the Wadden Sea and Ems estuary. Dumping of contaminated dredge spoil has a variable impact on the ecosystems, depending on the nature and amount of the contaminants (e.g. Reish 1980, 1981).

Dumping of non-contaminated sediments, however, may also have adverse effects on the functioning of the aquatic ecosystem. Suspension of fine sediments may influence nutrient dynamics in estuaries, affect growth of filter-feeding organisms and impair the performance of visual predators. At dump sites, local benthos may be affected by burial and by changing sediment properties. Dump sites in Dutch coastal waters range from 0.2 (e.g. in Wadden Sea) to 30 km^2 (near Hook of Holland).

For many years, the question of where to dump dredged sediments was mainly determined by two items of economical nature. One being related to the cost of shipping the dumped sediments to the dump site, the other related to the chance of recirculation of dumped sediments to the original dredge site. Since 1990, environmental effects have become more and more important in the policy regarding dumping of dredged sediments (F. Hallie Int. rep. 1990). As a consequence, in the late 1980s a research project ÕBAGHWADÕ was initiated by the National Institute for Coastal and Marine Management in cooperation with other ÔRijkswaterstaatÕ Directorates. This project focused on non- and low-contaminated dredge spoil; its aims were (1) to make available relevant information, (2) to obtain additional information on the effects of increased sedimentation at dump sites and of enhanced suspended matter

(SPM)-concentrations due to dredging and dumping operations, and (3) to provide options for management. Some preliminary and selected results have already been published (Essink et al. 1992; Essink 1996).

The main effects of dumping of dredged sediments relate to (1) burial of benthos at dump sites; (2) reduction of primary production due to increased turbidity; (3) impaired functioning of filter feeding organisms due to enhanced SPM-concentrations.

Effects of nutrient inputs

Estuaries play an important role in the cycle of nitrogen, phosphorus and silicium. In the Ems estuary and the Dutch Wadden Sea, P rather than N is the limiting element determining annual phytoplankton production (DeGroodt & de Jonge 1990; M.M. Engelkes Int. rep. 1989; de Jonge 1990). Therefore, attention was focused on release of P from dredged sediment at dumping and on the behaviour of suspended particle bound phosphorus during estuarine transport.

The estuarine phosphorus cycle

The estuarine P-cycle is shown in Fig. 1. Organic matter is not only being produced inside the estuary (e.g. by phytoplankton) but may also be imported. Through mineralization in the estuary, dissolved organic and inorganic P is formed, which in turn are transported to the sea (Postma 1954; de Jonge & Postma 1974). In the estuary itself, dissolved P is not evenly distributed as a result of geochemical, physical and biological processes; here, the behaviour of suspended particulate matter is important.

Calcite, which is formed at sea and is a component of the SPM, is transported up the estuary. During this transport phosphate adsorbs to calcite. In the upper estuary, however, part of the calcite dissolves due to lower ambient pH, releasing the phosphate. Other components of SPM, e.g. clay minerals and iron oxides, also can adsorb phosphate, especially in the upper estuary. In this way, transport of SPM may play a significant role in the estuarine P-cycle (de Jonge & Villerius 1989; Postma 1981).

Experiments showed that calcite plays a minor role in the estuarine phosphorus cycle. Iron(oxy)hydroxides proved to be more important by binding ca. 85% of particulate inorganic P (M.M. Engelkes Int. rep. 1989; de Jonge & Engelkes 1993; Rehm 1985). Besides Fe, Al plays a role in estuarine phosphate dynamics (van Beusekom & de Jonge 1994).

Release of phosphate from dredged sediments

Release of phosphate from dredged sediments at dumping was studied experimentally (M.M. Engelkes Int. rep. 1991). Original harbour sediment from the ÔEemshavenÕ (Ems estuary) was used, containing 20-40 mmol PO_4 per gram dry sediment. Aliquots were added to distilled water, artificial sea-water of 25 psu, artificial sea-water of 30 psu and filtrated water from the Ems estuary. For each water type three treatments were prepared with addition of 0, 1 and 2 mM PO_4, respectively.

Release of phosphate from harbour sediment was lowest in highly saline water, (contra Carritt & Goodgal 1954) and does not lead to the recommendation for dumping in a specific estuarine salinity zone in order to achieve a limitation of phosphorus release. Presence of PO_4 in the water reduced the release of P from the sediment. As there will always be phosphate in estuarine waters, it makes sense to recommend dredging and dumping operations not to take place in spring when phosphate concentrations in the water are low due to the spring bloom of phytoplankton. A better period would be autumn or winter, because then phosphate concentrations are high. Simulation of the effect of a 30-day dumping operation of 7000ton dry weight harbour sediment in the Ems estuary resulted in an increase of phosphate concentrations of 0.3 mM at most (M.M. Engelkes int. rep. 1991). Such an increase in autumn would not have a significant effect on phytoplankton growth.

Discussion and Conclusions

The role of Fe-, Mg- en Al-hydroxides in phosphate dynamics during estuarine transport needs to be further unravelled before relevant options for management can be formulated regarding the location of dumping of dredged sediments in the estuarine gradient. Although a large proportion of nutrients is contained in the sediments of the Ems estuary (van Beusekom & de Jonge 1998), release of phosphate from dumped harbour sediments is not likely to cause an unwanted increase of phytoplankton blooms. In fact, the effect of increased turbidity due to dumping is likely to surpass, and thus counteract, the effect of increased phosphate concentrations (DeGroodt & de Jonge 1990). Increase of nutrient concentrations,

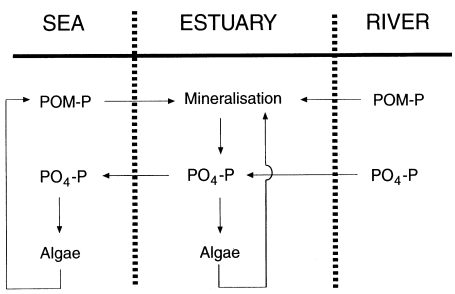

Fig. 1. Schematic presentation of the estuarine phosphorus cycle. POM = Particulate organic matter. Horizontal arrows indicate transport; vertical arrows indicate transformation processes (after de Jonge 1990).

Table 1. Management options for the location of dumping of dredged sediments in view of the effect on nutrient dynamics and nutrient-dependent biota.

Option for dumping	Effect
Near eelgrass bed	Possible increase of epiphyte growth and deterioration of eelgrass
Near mouth of estuary or in coastal water	Release of nutrients, little chance of increased phytoplankton bloom
In upper estuary or near tidal watershed	Release of nutrients, no chance of increased phytoplankton bloom Burial of nutrients followed by later release from sediments

however, may lead to increased growth of epiphytic algae on sea-grass leaves (Neverauskas 1987). A thick epiphyte cover restricts gas exchange and photosynthesis of seagrasses (Phillips et al. 1978; Shepherd et al. 1989).

In Table 1 some options for dumping of dredged sediments are listed with regard to the effects of associated nutrient release on phytoplankton and eelgrass.

Effects of increased turbidity

Dredging and dumping operations will cause local and temporal resuspension of sediments, in turn causing increased turbidity. Dredging at specific sites, where removal of sand bars leads to a change in the erosion-sedimentation equilibrium in estuaries, may even cause increase of SPM-concentrations in a large part of the estuary for a much longer period of time (de Jonge 1983). Increased turbidity may impair functioning of organisms such as phytoplankton, microphytobenthos, eelgrass and visual predators (e.g. fish and fish-eating birds).

Microphytobenthos and phytoplankton

Microphytobenthos of intertidal flats in areas such as estuaries and the Wadden Sea, is almost completely dependent on the period of tidal emergence for photosynthesis, due to the natural turbidity of the aquatic system. The total annual primary production at an intertidal flat is largely dependent on its elevation which determines its exposure to incident radiation (Colijn 1982; Colijn & de Jonge 1984). Consequently, increased water turbidity due to dredging or dumping operations will not have any significant effect on the productivity of micro-algae inhabiting intertidal flats.

Phytoplankton production is directly dependent on light penetration into the water column, which in its turn is strongly impeded by high SPM-concentrations. Old data (CadŽe & Hegeman 1974; Manuels & Postma 1974) show that annual primary production in the more turbid outer Ems estuary was ca. 50% of that in the western Dutch Wadden Sea, without significant differences in algal standing stock, measured as chlorophylla, between Wadden Sea and outer Ems (CadŽe & Hegeman 1974).

So, dredging and dumping operations are likely to affect phytoplankton productivity by increasing the water turbidity; however, this will be rather local and restricted in time, and therefore have little effect on total primary production of an estuary or of a tidal basin in which the dumping operation is carried out. Only in cases where local dredging at the seaward end of an estuary or tidal basin leads to increased SPM-concentrations in a larger part of the water body (cf. de Jonge 1983), a significant decrease in biomass and productivity of phytoplankton, zooplankton and filter-feeding benthos can be expected. In a simulation experiment for the Ems estuary, DeGroodt & de Jonge (1990) showed that a decrease in primary production would be larger in the inner than in the outer half of the estuary.

Effects on eelgrass

The occurrence and growth of eelgrass *Zostera marina* is highly dependent on the transparency of the water, as was shown for the Dutch Wadden Sea by Giesen et al. (1990). De Jonge & de Jong (1992) showed a clear relationship between water column transparency and the maximal depth at which sublittoral stands of *Z. marina* are found. Decrease in light penetration into the water due to dredging and dumping may therefore impair conditions for growth of this and other macrophyte species.

Effects on visual predators

Fish

Light plays an important role in finding, recognizing and capturing prey. An increase in water turbidity due to increased SPM-concentrations may negatively influence the performance of a visual predator, not only by the decrease in light intensity but also by changes in the spectral composition and polarisation pattern of the light (J.M. Baveco int. rep. 1988).

The degree to which visual predators are hindered by increased turbidity is determined by many factors. Some of these are related to the predator itself, e.g. a different light threshold, different resolutions of the fish eye in juvenile and adult specimens. Other factors relate to the prey, e.g. prey size and enhanced escape chances in more turbid water. The net effect of increased turbidity will thus be the resultant of various influences, both on the predator and on its prey.

In Dutch coastal waters, the visual predators herring (*Clupea harengus*) and sprat (*Sprattus sprattus*) avoid turbid waters. After 1960, adult dab (*Limanda limanda*) a then dominant flat fish in the Dutch Wadden Sea declined; young plaice (*Pleuronectes platessa*) is now dominant. The decrease is related to an increase in turbidity in the western part of this sea (de Jonge et al. 1993).

Birds

Fish-eating birds, e.g. cormorants (*Phalacrocorax carbo*), use their eyes in chasing and capturing their prey under water. For this species, which also forages in the Wadden Sea, it is not known whether turbidity of the water affects its foraging success, but the sandwich tern (*Sterna sandvicensis*) needs clear water in order to locate its prey Ð mainly consisting of young herring, sprat and sand-eel (*Ammodytes* sp.). The increase in turbidity in the Dutch coastal zone since the 1960s is considered as a possible cause of reduced breeding success of sandwich tern breeding in the Wadden Sea area. This seems to be caused by the greater distance from the breeding colony adults have to cover in obtaining prey for their young (A. Brenninkmeijer & E. Stienen Int. rep. 1992; B. ten Brink & F. Colijn Int. rep. 1990).

Diving ducks such as the eider (*Somateria mollissima*) take their food (*Mytilus edulis*, *Cerastoderma edule*, *Carcinus maenas*) from the bottom of the Wadden Sea. It is not known whether visual location of prey is involved (C. Swennen pers. comm.) In common scoter (*Melanitta nigra*) which dives down to more than 20m to catch food (mainly *Spisula* spp.) (M.F. Leopold Int. rep. 1996), visual clues cannot play a significant role.

Discussion and Conclusions

Reduction of phytoplankton primary production through increased turbidity may be minimized by dredging in autumn or winter. Then, natural productivity, zooplankton activity and food requirements of filter feeders are low. The negative effect of increased turbidity as a result of dredging and dumping in spring or summer, will be dependent on (1) the extent of the area affected, (2) the extent to which light penetration in the water is reduced, and (3) the duration of increased turbidity conditions. These aspects will be determined by local conditions.

Increased turbidity would make the Wadden Sea and Ems estuary less attractive to fish like herring, sprat and smelt (*Osmerus eperlanus*). Herring occurs in the Dollard exclusively in winter (Anon. 1985), but during the greater part of the year in the western Dutch Wadden Sea (Fonds 1978). Sprat is present in both areas almost the year round. During summer, smelt is less abundant than in other seasons due to reduced tolerance of higher temperatures (Fonds 1978). In order to minimize effects on herring and smelt, dredging would have to take place in summer. This means that no common option can be advised for both phytoplankton and fish. Dredging and dumping in autumn may mean a compromise.

A large-scale increase of turbidity, such as may arise in specific dredging operations (see de Jonge 1983), is likely to affect demersal- as well as pelagic fish in estuaries and coastal waters. If dredging and dumping activities in the coastal zone and outer deltas of the Wadden Sea during spring would increase turbidity, this could negatively influence the development of the population of the sandwich tern in The Netherlands.

The above-mentioned results and considerations suggest several options for dredging and dumping, mainly relating to the location and time of the year (Table 2).

Table 2. Management options for the location and time of dumping of dredged sediments in view of the effect on water turbidity and light dependent biota.

Option for dumping	Effect on water turbidity and biota
Near eelgrass beds	Deterioration of eelgrass
Outer estuary,Wadden Sea	Some damage to herring, sprat and smelt
Near mouth of estuary	Reduced foraging for sandwich tern Decrease in phytoplankton production
In upper estuary or near tidal watershed	Some decrease in phytoplankton production
In spring/summer	Decrease of phytoplankton production Negative effect on visual predators
In winter	No decrease of phytoplankton production Negative effect on herring in upper estuary
In autumn	No decrease of phytoplankton production No negative effect on visual predators

Effects of enhanced SPM-concentrations

During dredging and dumping large amounts of sediment are brought into suspension; increased concentrations may interfere with food intake of filter-feeding benthos (bivalves) and copepods. Also, functioning of gills of fish may be impaired due to clogging.

Filter-feeding bivalves

Filter-feeding bivalves utilize the digestible part of the SPM, such as phytoplankton, bacteria and decomposable detritus. An increase of the admixture with anorganic particles (sediment), due to dredging or dumping, may lead to a decrease in the net food intake per unit of time. Widdows et al. (1979) presented a practical summary of the process of food intake and digestion and its relationship with SPM-concentration (Fig. 2). The eventually assimilated energy will be dependent on the content of utilizable components in SPM. This was corroborated by Essink & Bos (1985) who translocated bivalves (*Mytilus edulis*, *Cerastoderma edule*, *Mya arenaria*) and found that growth was strongly correlated with the chlorophyll a- and organic matter content of SPM. A similar indication of growth regulation by food quality of SPM was found in *Macoma balthica* (Laane et al. 1987), a bivalve which is a deposit- as well as a filter-feeder (Kamermans 1994).

The filtration by bivalves of the ambient particle suspension over the gills can be regulated in relation to particle concentration (see Fig. 2). The quality of food particles Ð determined by the degree of mixture of silt and phytoplankton (Prins & Smaal 1989) Ð is important.

A second possibility to regulate food intake is adaptation of the size of gills and labial palps. Particles retained on the gills get sorted by the labial palps. Non-utilizable particles are diverted from the mouth and expelled as pseudofaeces. Theissen (1982) found that blue mussels (*Mytilus edulis*) have smaller gills and larger palps in the Danish Wadden Sea than in Baltic waters. These differences relate to the higher SPM-concentrations in the Wadden Sea. This was confirmed by data obtained on mussels from the Dutch Wadden Sea and North Sea (Fig. 3) and on mussels from a SPM-gradient in the Bristol Channel-Severn estuary from 5 - 500 mg/l (Mettam 1993). In the North Sea (0 - 20 mg/ l SPM), relative gill size is high in order to be able to catch enough particles, and relative palp size is low because little sorting has to be done. In the Wadden Sea (SPM-concentrations 40 - 400 mg/l), gills are relatively small (ample particle supply) but palps are relatively large because much sorting has to be done due to admixture of silt and detritus. Between 0 and 50 mg/l of SPM the relative size of gills and palps is linearly related to mean SPM-concentration (S. Brett & A. Wassenaar Int. rep. 1990). Indeed, reciprocal translocation experiments with *Mytilus edulis* carried out between Wadden Sea and North Sea showed a clear adaptation of the size of gills and palps of transplanted mussels to the SPM-regime of their new locations (Essink et al. 1989). Translocation within the Wadden Sea showed that mussels only adapted their palp size; gill size remained basically unchanged because of the highly variable SPM-concentrations (Essink et al. 1989). The energetic costs of these adaptations, which may result in reduced growth, are not known.

Indications of energetic costs of dealing with enhanced SPM-concentrations were obtained by exposing *Mytilus edulis* of 3cm shell length to three levels in the water column in a tidal channel of the Ems estuary (K. Essink et al. Int. rep. 1990). Near the bottom, mean SPM-concentration was 17% higher than near the surface. SPM-concentrations >250 mg/l occurred more frequently (2-3 ×) near the bottom than at mid-depth or near the surface. Growth of the soft parts under near-bottom conditions was reduced, though not significantly. This is in agreement with experimental results from Widdows et al. (1979). For 3-cm large mussels maximal filtration rate was found at SPM-concentrations of ca. 125 mg/l. At 225 mg/l of SPM, filtration rate had decreased to ca. 30% and at 250 mg/l to zero. So, reduced growth of near-bottom mussels was most likely caused by more frequently occurring high SPM-concentrations, not withstanding the adaptation of the musselsÕ filtration and sorting apparatus to SPM-concentrations in the Wadden Sea above ca. 50 mg/l (see above). The observations of faster growth of *Mytilus edulis* in more turbid environments (Hawkins et al. 1996) refer to SPM-concentrations of ca. 90 mg/l as compared to 10 mg/l. In this range of SPM, the bivalve is able to make a profitable use of the higher organic matter concentrations at higher SPM-concentrations, being in agreement with the first part of the optimum curve relationship presented by Widdows et al. (1979).

Zooplankton

In estuarine environments, salinity and temperature largely determine distribution and seasonal succession of zooplankton. It is not clear to what extent the distribution of zooplankton in the Ems estuary is dependent also on SPM-concentrations that might interfere with food selection by copepods (Baretta & Ruardy 1988). Findings by Chervin (1978), that net growth efficiency of *Acartia tonsa* is inversely related to the proportion of detritus in their food, indicate that growth of copepods can be affected by dumping of dredged sediments with a high organic matter content. One may wonder whether these findings can be transferred to the Ems estuary and Wadden Sea Ð ChervinÕs observations are from the Hudson river estuary and New York Bight, at SPM-concentrations of only a few mg/l. Results of laboratory experiments sug-

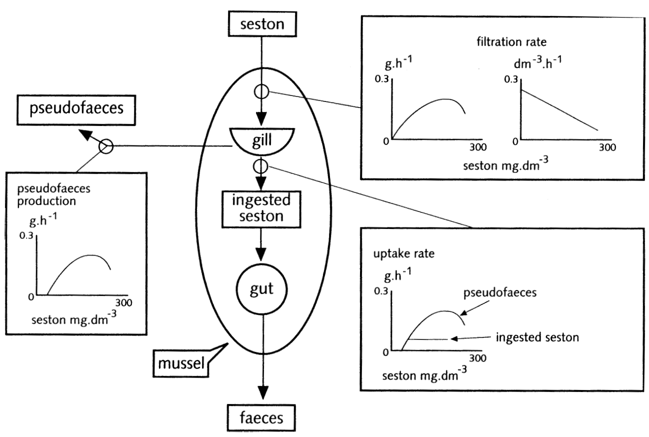

Fig. 2. Schematic presentation of filtration and ingestion of suspended matter (seston) and of pseudofaeces production by *Mytilus edulis* as a function of suspended matter concentration (after Widdows et al. 1979).

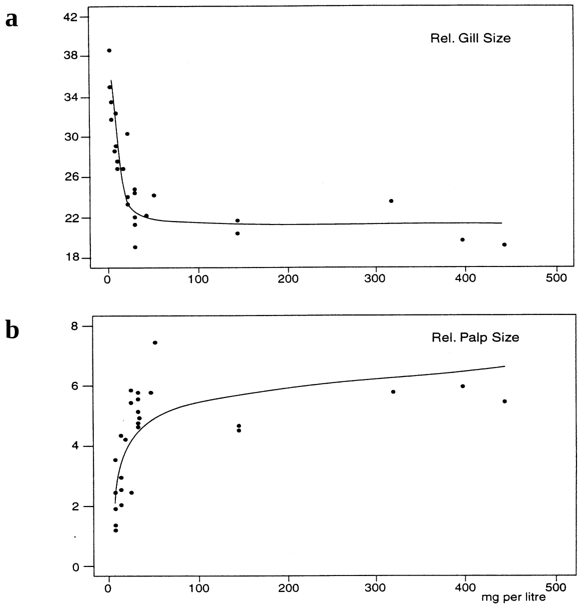

Fig. 3. Relationship between mean suspended matter concentration (SPM; mg/l) and **a.** Relative gill size and **b.** relative palp size of *Mytilus edulis* from Dutch coastal waters, North Sea, Baltic Sea, Limfjord, Kattegat. SPM data for Dutch waters and North Sea refer to April-August; data for Danish waters from Theisen (1982) (from S.E. Brett & A. Wassenaar Int. rep. 1990).

gest that food intake by copepods such as *Acartia tonsa* and *Eurytemora affinis* is negatively influenced at SPM-concentrations > 250 mg/l (Sherk et al. 1975).

Fish

According to a literature survey by J.M. Baveco (Int. rep. 1988), enhanced SPM-concentrations can cause sub-optimal functioning of gills by clogging, causing decreased gas exchange. The sensitivity of fish to this varies with species and life stage, nature of suspended particles, temperature and dissolved oxygen content of the water. It is assumed that pelagic fish are more sensitive to high SPM-concentrations than demersal fish. Plaice survived laboratory exposure to SPM better than some pelagic species (Newton 1973 - cited by Moore 1977). Generally, juveniles are more sensitive than adults. J.M. Baveco (Int. rep. 1988) does not report significant fish mortality near dredging operations. Neither was increased mortality demonstrated in caged fish near dredging locations.

Discussion and Conclusions

No boundary value for SPM is known above which gill functioning in fish is seriously impaired. If the results of Sherkin et al. (1975) hold for field conditions, copepods will meet unfavourable conditions in a large part of the Ems estuary, i.e. where the natural SPM-concentration is < 250 mg l.

Table 3. Management options for the location and time of dumping of dredged sediments in view of the effect on suspended matter concentrations and suspension feeding biota.

Option for dumping	Effect
Near mouth of estuary	Possible negative effect on zooplankton Negative effect on growth of filter feeding benthos if increase of SPM > 20%
In upper estuary or near tidal watershed	No negative effect on zooplankton (?) Less negative effect on filter feeding benthos
In autumn/winter	No negative effect on filter-feeding benthos

Enhancement of mean SPM-concentrations in estuaries with 10 - 20% will not cause problems for growth of the blue mussel, but if dredging or dumping leads to an increase in the size-specific filtration rate limiting SPM-concentrations, negative effects on mussel growth is expected. Results for *Mytilus* may be extrapolated to other filter feeding bivalves (Essink & Bos 1985; M¿hlenberg & RiisgŒrd 1978, 1979; Prins & Smaal 1989).

With respect to the effects of increased SPM-concentrations some options for dumping are possible, most of which relating to the location of dumping (Table 3).

Effects of enhanced sedimentation

At dump sites, local benthos has to cope with deposition of sediments which are in many cases strongly anaerobic. Sensitivity of benthos to being covered by dredged sediments is strongly dependent on the thickness of sediments and their ability to restore contact with the overlying water (R. Bijkerk Int. rep. 1988).

Microphytobenthos, macro-algae and eelgrass

Microphytobenthos, mainly living on intertidal sand- and mud flats, will only be confronted with deposition of dredged sediments after resuspension and secondary deposition in the vicinity of a dump site. Epipelic diatoms among the microphytobenthos are well adapted to sediment reworking by waves and currents. With help of their raphe they can move through the sediment at rates of 0.2-25 mm/s, depending on the species. Maximal crawling velocity is about twice as high (Harper 1977).

In an area of the Dutch Wadden Sea with 85 *Arenicola marina* and 500 *Heteromastus filiformis* individuals per m^2 CadŽe (1976,1979) found that sediment reworking by these polychaetes amounts to ca. 30 and 4 cm/yr, respectively. There is no indication for a reduced standing stock of microphytobenthos in assemblages with *Arenicola* and *Heteromastus*. Diatoms can survive sediment burial for 20 - 80 days, even when the sediment is anoxic (Admiraal & Peletier 1979).

No data were found regarding the amount of sediment deposition fatal to intertidally occurring macroalgae, such as *Ulva* and *Enteromorpha*.

Stability of intertidal sediments plays an important role in the establishment and maintenance of beds of *Zostera marina* and *Z. noltii* (Philippart et al. 1992). No data were found in the literature on the sensitivity of eelgrass to deposition of sediment.

Macrozoobenthos

In the extensive literature search by R. Bijkerk (int. rep. 1988) on effects of enhanced sedimentation, attention was focused on factors such as (1) thickness of the sediment deposited, (2) characteristics of the sediment, (3) sedimentation rate, and (4) season. Thickness will be an important factor at sites where large amounts of dredged sediments are dumped in a short period of time (Ôincidental depositionÕ). Sedimentation rate is relevant in situations of intermittent dumping spread over a longer period of time.

In instances of incidental deposition, sessile species such as oyster (*Ostrea* sp.), mussel (*Mytilus edulis*) and mud anemone (*Sagartia* sp.) have a low tolerance of sediment cover. For motile species, tolerance varies strongly per species. For each species a Ôfatal depthÕ was derived indicating the maximal sediment thickness a species can overcome by crawling to the sediment surface (Fig. 4). Fatal depth is in many cases different for deposition of mud vs. sand. In general, species inhabiting sandy bottoms can withstand deposition by sand better than by mud. Furthermore, in many species juveniles do survive sediment deposition less than adults, with the exception of *Mya*, where the adults are not able to move upward after deposition of layers of mud or sand. Benthic animals that are covered by a layer of sediment experience anoxic conditions and increased sulphide concentrations. Species inhabiting muddy sediments such as *Mya* have a natural tolerance of such conditions. However, the brown shrimp (*Crangon crangon*) is very sensitive (Theede 1973).

Frequent dumping of dredged sediments with short intervals has an effect similar to increased sedimentation during the whole period of dumping. Even before fatal burial depth is reached, benthic animals can be affected, especially when sedimentation rate is higher than the rate at which the animals can maintain their position relative to the sediment-water interface in the newly deposited sediment. Fig. 5 shows an approximation of the maximal tolerance of various benthic species to continuing sedimentation of dumped fine sand and mud. In general, tolerance is greater for sedimentation of fine sand than for mud. Deposition of sediment on mussel beds may also hinder settling and survival of mussel larvae (Bender & Jensen 1992) in

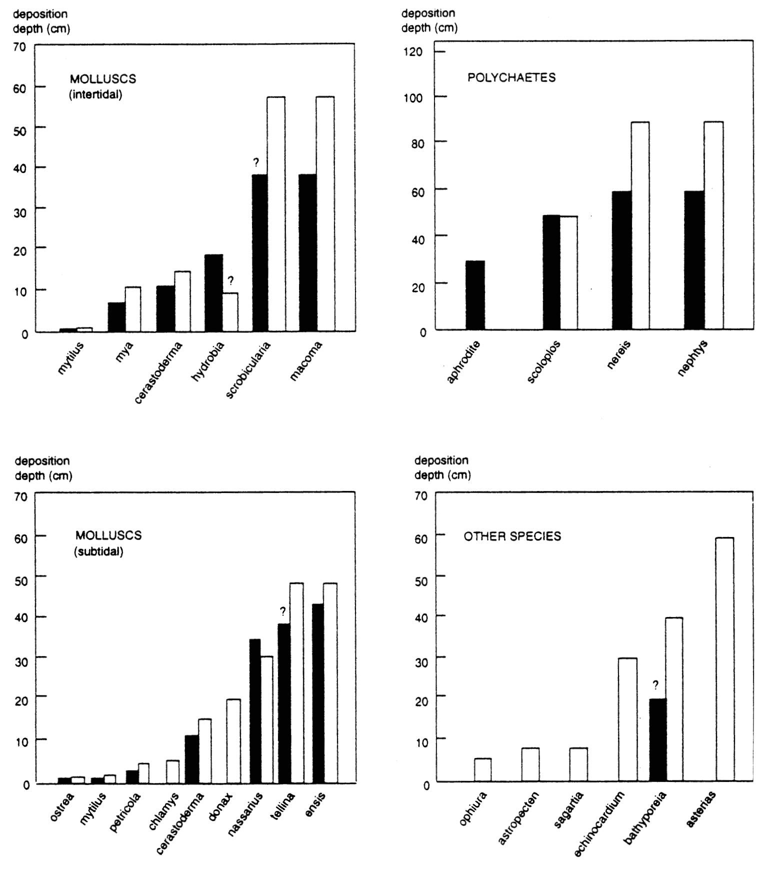

Fig. 4. Fatal depth (cm) for macrozoobenthos at incidental deposition by mud (dark columns) or sand (white column). After R. Bijkerk (Int. Rep. 1988).

this way preventing natural succession of mussel bed communities.

R. Bijkerk (Int. rep. 1988) compared results obtained at higher and lower temperatures (cf. summer and winter). At lower temperatures mortality among macrozoobenthos was lower and there was a higher tolerance of low oxygen conditions. The percentage of animals escaping from burial by crawling upward through the deposited sediment, however, was always lower at lower temperatures. These results are related to seasonal differences in metabolic activity of these poikilothermic animals.

Field observations were made at a dump site in the Ems estuary, where ca. 544000 m ^{3} of sediment from the Eemshaven harbour (70% <20 µm) were dumped in December 1989, and ca. 850000 m ^{3} in October 1990 (H.L. Kleef et al. Int. rep. 1992). After the first dumping campaign of ca. 3 weeks, a decrease in species richness and abundance of major species was observed at the dump site. This decrease was most clear in those parts of the dump site where the sediment layer deposited was thicker than ca. 0.3 m. For instance, the polychaete *Nephtys hombergii* was very sensitive to sediment deposition of 3 dm and more (Kleef et al. 1992); this was more sensitive than found in the literature (see Fig. 4). A similar conclusion was drawn with respect to *Macoma balthica* (≥ 1 yr) and *Scoloplos armiger*. *Macoma* decreased much more than expected from the literature with a fatal depth of 0.5 m (Fig. 4), while *Scoloplos* had completely disappeared. The almost complete disappearance of *Bathyporeia* sp. is in agreement with information found in the literature (see Fig. 4). Juvenile *Macoma balthica* (0-group), on the other hand, had doubled its density in the dump site. This apparent positive reaction of *Macoma* spat to the dumping of dredged sediments is in fact negative because in the reference area directly around the dump site *Macoma* spat increased more than sixfold from before to after dumping. The latter increase is the result of normal migration of *Macoma* spat from high intertidal flats to deeper locations in winter (Beukema & de Vlas 1989; Essink & Beukema 1991). The dumping of dredged sediments in October 1990 did not have the same effect on *Macoma* spat because then it was still too early for the normal winter migration.

During spring and summer of 1990, the benthic fauna at the dump site recovered both regarding species richness and abundance of *Nephtys hombergii* (Fig. 6).

Meiofauna

The survival of nematodes in sediment cores from the Ems estuary intertidal flats was measured in experimental dumping treatments (1 - 10 cm deposition) with various sediment types (K. Romeyn & J. Leiseboer Int. rep. 1989). A layer of up to 10 cm thick did not cause

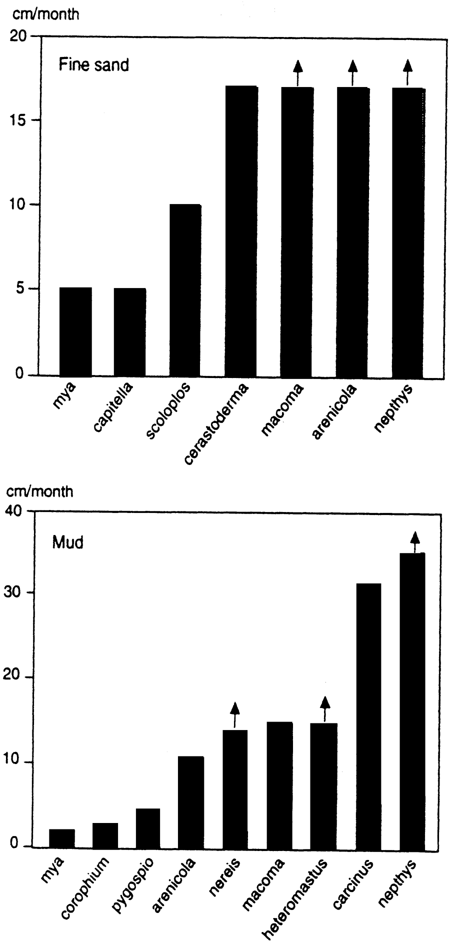

Fig. 5. Maximal tolerance of macrozoobenthos to continuing sedimentation (rate in cm per month) of dumped fine sand and mud. For different species measured with different methods (after R. Bijkerk Int. rep. 1988).

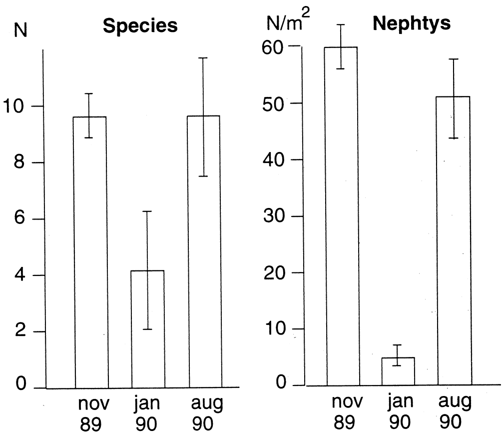

Fig. 6. Mean number of species per sampling station and mean abundance of *Nephtys hombergii* before dredge spoil dumping (Nov. 1989), shortly after dumping (Jan. 1990), and after eight months of recovery (Aug. 1990) (from H.L. Kleef et al. Int. rep. 1992).

any negative effect. Within 1 - 2 weeks time, nematodes migrated upward through the newly deposited sediment and survived, at least when both the original and the deposited sediment was fine sand. However, in case of deposition of mud on sand, upward migration of nematodes was limited to a few cm only.

Discussion and Conclusions

Nematodes are likely to survive deposition of sediment up to ca. 10 cm provided the deposited sediment is not too different from the locally present one. In tidal channels, nematode assemblages are usually impoverished; assemblages in intertidal flats are much richer (Bouwman 1983). As most dumping of dredged sediments takes place in tidal channels or other subtidal locations in the coastal zone, negative effects on intertidal nematode populations are not very likely.

Deposition of 10 cm of fine sediment, however, does have clear negative effects on bivalves, such as *Mytilus edulis*, *Mya arenaria* and *Ostrea* spp., on the anthozoan *Sagartia* spp., and on some starfish species (cf. Fig. 4). According to R. Bijkerk (Int. rep. 1988) *Cerastoderma edule* can not survive burial by more than 8 - 12 cm of sediment. Field observations and experiments by J. van Dalfsen (Int. rep. 1994), however, indicate a greater susceptibility of this species. Mortality was found to occur after deposition of 5-8 cm of fine sediment. The susceptibility of macrozoobenthic species to burial by dumped dredged sediments may vary to some extent dependent on local conditions. In the Baltic Sea, Olenin (1992) found *Nereis diversicolor* and *Macoma balthica* to be among the most resistant species, whereas *Corophium volutator* and *Pygospio elegans* were the most sensitive species.

Repeated surveys at the Ems estuary dump site by H.L. Kleef et al. (Int. rep. 1992) have shown that several species were more susceptible to deposition by dredged sediments than was known from the literature (R. Bijkerk Int. rep. 1988). Nevertheless, it seems warranted to conclude that most macrozoobenthic species other than *Mytilus edulis*, *Mya arenaria*, *Ostrea* spp., *Sagartia* spp. and some starfish species will not be seriously affected as long as sediment deposition is restricted to 0.2 - 0.3m.

In view of the great susceptibility of *Mytilus edulis* and *Ostrea* spp. to deposition of sediment (fatal depth: 1-2 cm) it would be advantageous to avoid sedimentation of dredged sediments after dumping on intertidal and shallow subtidal mussel and oyster beds. Also the cockle would benefit from such an approach. These bivalve species represent considerable economic values in many estuarine and coastal areas.

Macrozoobenthic species were found to have a greater tolerance to deposition of sediment at lower than at higher temperatures. At lower temperatures, however, their crawling activity, and therefore their chance of escaping from being buried, is smaller. Dumping of dredged sediments in autumn, when water temperatures are not yet really low, and the condition of most animals is good (being at the end of the growing season) would therefore be a promising option for management.

In-between dumping occasions recovery of macrozoobenthos at the dump site may occur provided that the time for recovery is long enough and includes the period of reproduction and settlement of recruits. One might consider a benthic community to be recovered when at least 80% of the species diversity and biomass has been restored (cf. H.L. Kleef et al. Int. rep. 1992; Newell et al. 1998). Regular use of a dump site more than 1 - 2 times per year is likely to create a long lasting impoverishment of benthic fauna at the location(s) where the dumped sediment will deposit. Recovery will vary from species to species.

From the obtained results and considerations several options can be derived regarding dump site location, and time and frequency of dumping (Table 4).

Table 4. Management options for the location, time and frequency of dumping of dredged sediments in view of the effects of enhanced sediment deposition on benthic fauna.

Option for dumping	Effect
Near eelgrass beds	Possible deterioration of eelgrass
Near intertidal flats	Decreased production by microphytobenthos depending on sedimentation on intertidal flats Mortality among meiofauna depending on extent and type of sedimentation on intertidal flats Mortality among macrobenthos depending on extent and sedimentation type on intertidal flats
In tidal channels and other subtidal areas	Mortality among meiofauna depending on extent and type of sedimentation on intertidal flats Mortality among macrobenthos depending on extent and sedimentation type on intertidal flats
In spring/summer	Macrozoobenthos susceptibility larger than in winter
In winter	Little chance for macrofauna to escape from burial Bad for winter migration of juvenile *Macoma balthica*
In autumn	Chance for macrofauna to escape from burial Normal winter migration of juvenile *Macoma balthica*
Dumping at small site	Great impact on benthos of small area
Dumping distributed over large area	Little impact on benthos of dumping area
Use site each year	Incomplete recovery in-between dumping occasions
Each year different site	Better recovery of benthos at dump site

General Discussion

Most dredging and dumping activities take place in estuarine and coastal waters. These areas are generally relatively rich in nutrients (cf. McComb 1995). The nutrient release experiments and ecosystem model simulations performed in the Ems estuary indicate only little, if any, influence of dredging and dumping on nutrient dynamics and therefore on phytoplankton primary production. However, the ÔEemshavenÕ sediment tested was fine sand, whereas most harbour sediments will be finer and contain higher amounts of organic matter and nutrients. The effect of nutrient release from dredged sediments is much dependent of adsorption and desorption processes and the role of inorganic and organic suspended particles during transport in the estuarine gradient.

With both dredging and dumping, sediment goes into suspension, changing the SPM-concentration and turbidity of the water column. For proper management of dredging and dumping operations quantitative insight into the spatial and temporal distribution of suspended sediments would be advantageous. To this end, use and further development of mathematical models for distribution and transport of suspended sediments and simulation of ecological effects (e.g. Bach et al. 1992, 1997; de Kok 1992, 1995; Moritz & Randall 1995) is recommended. A model approach seems essential in the selection of dump sites in coastal systems, such as the international Wadden Sea, where commercial exploitation of natural resources (e.g. shellfish) and nature protection are clearly outlined policies. However, the predictability of functional system responses has certain limits (cf. de Vries et al. 1996).

Negative effects on benthos at dump sites can be minimized by selecting bottoms which are poor in benthic life. On the other hand, negative effects can also be reduced by dumping in such a way that the layer of deposited sediment does not exceed 0.2 - 0.3 m. Mathematical models describing the density changes of dredged sediments during dumping, the impact on the sea-bed and the subsequent horizontal dispersal of the sediments as influenced by water currents and bottom topography, such as developed by T. van Heuvel (Int. rep. 1988) and further evolved since then, may be a useful tool. Sediment transport models have the potential of becoming important management tools in the prediction of effects of dredging and dumping in areas with commercially important shellfish stock (mussel cultivation lots, cockles, *Spisula*, shrimps).

An alternative for dumping of dredged sediments in tidal channels would be storage in depots above the high water mark. This would, however, drastically alter the natural character of intertidal flats and shores of estuaries. From experience with the Ôdep™t de BilhoÕ in the Loire estuary we know that building up such a depot will have a great spatial effect on sediment composition, benthos, fish and birds (Gallenne 1990).

An other example of changing the Ems estuaryÕs morphology and accompanying SPM-regime by dredging, was already referred to (de Jonge 1983). In the Western Scheldt (SW Netherlands), increased dredging of shipping channels to the port of Antwerp has changed the morphology of the estuary. Since ca. 1970 dredging efforts increased, especially in the eastern part of the estuary (Huys 1995; J. Vroon et al. Int. rep. 1997). Dumping of the sediment mainly in secondary tidal channels has led to increased sedimentation on some intertidal flats. As a consequence, steepness in the estuary has increased, i.e. shallow subtidal areas have largely disappeared, and the total area of intertidal flats has decreased. Furthermore, the mean height of intertidal flats increased as well as the proportion of high dynamic flats within the intertidal area. Although complex by nature, these morphological and sedimentological changes did influence the composition of the macrobenthic communities, and thereby the numbers of birds feeding on the estuarine intertidal flats (J. Coosen pers. comm.). The latter examples merely indicate that dredging may very well have other, far-reaching, effects than those dealt with in this paper.

Most of the options for management as given at the end of each section in this paper, are qualitative rather than quantitative. No attempt is made towards an integrated presentation of the various impacts, with different weighing factors given to copepods, nematodes, cockles, eelgrass etc. The reason for this is that the applicability of these options will very much depend on the local hydrological (e.g. dispersal of dumped sediments) and ecological (e.g. presence of eelgrass beds, commercial shellfish stocks) conditions present in different water bodies. As an example, it may for several reasons be recommended to dump in the outer part of a particular estuary, but if in that outer part important stocks of cockles or eelgrass occur that option will not hold. Furthermore, the options also relate to differences in amount and type of dredged sediment, locality of (optional) dump site and of the method of dumping. Therefore, managers and authorities have to choose between the presented options in accordance with the local situation in their management area. Recently, Dutch harbour managers in the Wadden Sea and Ems estuary, in their application for new dumping permits, have to make clear to Dutch and German authorities that the dump site they want to use, and their practice of dumping, does not seriously affect specific natural resources.

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