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Article

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Combining sediment management and bioremediation in muddy ports and harbours: a review

Polrot, A.[1]*, Kirby, J.R.,*[1] *Birkett, J.W..*[2] *and Sharples, G.P.*[2]

[1] School of Biological and Environmental Sciences, Liverpool John Moores University, Byrom Street, Liverpool, L3 3AF. Corresponding author: A.K.Polrot@2017.ljmu.ac.uk

[2] School of Pharmacy and Biomolecular Sciences, Liverpool John Moores University, Byrom Street, Liverpool, L3 3AF.

Abstract

This paper reviews two important sources of innovation linked to the maritime environment and more importantly to ports: the potential coupling of sediment management and (bio)remediation. The detrimental effects of dredging are briefly considered, but the focus here is on a sustainable alternative method of managing the problem of siltation. This technique consists of fluidising the sediment *in situ*, lowering the shear strength to maintain a navigable under-keel draught. Preliminary investigations show that through this mixing, aeration occurs, which results in a positive remediation effect as well. An overview of port contamination, remediation, and the recent research on aerobic (bio)degradation of port contaminants is made in order to show the potential for such innovative sediment management to reduce dredging need and remediate contaminated mud in ports. This review also highlights the lack of full-scale field applications for such potential remediation techniques, that remain largely confined to the laboratory scale.

Keywords:

sediment resuspension, anti-siltation, fluid mud, aerobic biodegradation, sediment contaminants, active nautical depth.

Introduction

Sediment contamination and siltation are among the major issues impacting port operations and management. Dredging has been the answer to these issues for years. During dredging sediment is excavated to maintain navigable depth and disposed of outside the port or harbour. This process needs to be regularly repeated due to continued sediment movement and redeposition within the coastal system. The practice of dredging comes with significant financial and environmental cost. For example, there are strong perturbations of the ecosystem during excavation, transportation and disposal (Erftemeijer and Lewis, 2006; Manap and Voulvoulis, 2015; Todd et al., 2015). Some alternative methods have therefore been developed to manage sediment at lower cost and/or with less disturbance to the environment (Bianchini et al., 2019; Kirby, 2011). When the sediment is contaminated, however, the solution again is dredging, this time to put the sediment in confined facilities (Group 5, 2002) where it remains without further processing, or it is subjected to *ex-situ* remediation, usually at high cost (Du et al., 2014).

This paper reviews the existing methods to replace or reduce the need for dredging as well as the ways to remediate contaminated sediment, in particular through *in situ* biodegradation. By doing so, it highlights the potential for a procedure that uses mud mixing and aeration to render the sediment navigable, with the potential additional benefit of also acting as a remediation method (Kirby, 2013, 2011; Polrot et al., 2018). Such a procedure would encourage the growth of aerobic microorganisms, which are capable of contaminant degradation. This could represent an innovative way to manage sediment in ports at potentially lower costs and with a beneficial impact on the environment.

1. Sediment management

1.1. Dredging

Most ports and harbours in the world experience siltation problems that have hindered ship navigation since ancient times. In ancient Egypt, workers used to drag the mud manually until the method improved when the first dredging machine was developed in 1796 (Knight and Lacey, 1843). Dredging consists of the excavation of the sediment from the site, followed by its transport and disposal in a designated area, normally offshore. Both the excavation and the disposal are strictly regulated and subject to legislation aimed at minimising environmental impact, especially because of the potential presence of harmful chemical contaminants. In England, the Marine Management Organisation (MMO) is the licencing authority for dredge disposal sites and operate under OSPAR[a] commission's guidelines (OSPAR, 2004).

1.1.1. Environmental impact of dredging

The negative impacts of dredging comprise the effects of the excavation method itself (locally) and the effects of contaminated sediment manipulation (more widely). These effects can be physical, chemical, and biological. These are discussed below in relation to the dredging of uncontaminated and contaminated sediment.

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[a] From the unification and extension in 1992 of the OSlo and PARis conventions which occurred respectively in 1972 and 1974

When dredging uncontaminated sediment, different problems can be encountered. Erftemeijer and Lewis (2006) reviewed the impact of dredging on seagrass and reported, in addition to the impact of physical removal at the excavation site and burial at the disposal site, a potential effect of the turbidity and subsequent sediment deposition. The resulting decrease in photosynthetic activity as well as smothering causes a loss of seagrass vegetation. The impact of turbidity would be higher on fast growing species as slow growing species can resist the decrease of light for a longer time and are therefore more resilient to turbidity events (Erftemeijer and Lewis, 2006). Another review was later published to report the lethal effect of dredging induced turbidity and sedimentation on coral reefs, with an impact ranging from no detectable effect to 80% of coral loss (Erftemeijer et al., 2012). High coral mortality following dredging operations have still been observed in the past years, for example more than 560 000 corals were reportedly killed during dredging operations of Port of Miami between 2013 and 2015 (Cunning et al., 2019). The turbidity could also reduce the production of phytoplankton and affect the gills and membranes of membrane-feeding organisms (Balchand and Rasheed, 2000).

During excavation, an abundance of nutrients are released into the water column. This causes a strong perturbation to the ecosystem, which can have an impact on the macrobenthic fauna by causing the population of native organisms to decrease in number (Ponti et al., 2009). The habitat is also modified during the process, with a potential change in sediment properties at the disposal site. This can affect the ability of the benthic fauna to recover after the dredging perturbation (Cooper et al., 2011).

In addition, the removal of sediment from the coastal system has a strong impact on the surrounding physical environment, leading to long-term changes to the adjacent shoreline indirectly through modifications of wave patterns and directly via the filling of the excavation cavity by sediment transported from the elsewhere in the coastal system (Demir Hüseyin et al., 2004). A secondary impact of dredging is the emission of greenhouse gas that occurs mainly during the transportation phase but also during the excavation itself. It has been estimated that dredging activities could release between 6.5 and 11.7 kg CO_2 per ton of dredged sediment (Bianchini et al., 2019).

An acoustic impact of dredging has also been subjected to research. The noise produced by dredging can be as high as 170-190 dB re 1 $\mu Pa^2 m^2$ at 50 Hz (Todd et al., 2015). These levels are thought to be too low to provoke physical damage to animals but they can induce stress, which may hinder their reproduction, modify their foraging behaviour and could have other detrimental consequences on their survival, for example, through diseases induced by toxin production (Pirotta et al., 2013; Todd et al., 2015). The overall consequence of these phenomena is a decrease in benthic faunal diversity after dredging operations (Barrio Froján et al., 2011; Kenny and Rees, 1996).

For the dredging of contaminated sediment, the negative effects increase significantly, as the process increases the exposure of flora and fauna to the toxicity of the contaminants (Manap and Voulvoulis, 2015).The resuspension of sediment during the excavation can result in the release of contaminants around the excavation site (Munawar et al., 1989; Roberts, 2012) and the excavation exposes a new layer of potentially highly contaminated sediment. Some of these contaminants, such as heavy metals, are immobilized in the form of sulphide complexes in anoxic sediment which are dissolved through an oxidation process during resuspension (Roberts, 2012). This increases their bioavailability and therefore their ability to exert toxicity towards the surrounding organisms (Roberts, 2012). These processes are however constrained by numerous factors that can limit them and mitigate the increase in toxicity. In many cases for example, the oxidised iron rapidly acts as a scavenger for the other dissolved metal forms and prevents them from becoming further oxidised to more toxic forms (Roberts, 2012). The spreading of contaminants can also occur during sediment transportation to the disposal site as, in practice, dredging often continues after the hopper is full, even during the transport, and it leads to an excess of sediment that overflows from the hopper (Manap and Voulvoulis, 2015). The targets of contaminant exposure comprise three types: the organisms living in the sediment (benthic fauna), pelagic organisms (fish and plankton) and consumers (fish, birds, mammals and humans) (Bridges et al., 2010). Strong increases in the bioavailability (Eggleton and Thomas, 2004) and bioaccumulation of contaminants have been reported after dredging activities (Hedge et al., 2009; Martins et al., 2012; Winger et al., 2000), which leads to the distribution of these toxic compounds through the entire food chain.

1.1.2.Regulation

In recognition of the significant environmental impacts of dredging, a range of rules and regulations have been implemented at local, national and international level with the aim to control and reduce the negative effects of the process. Firstly, restrictions have been put in place by the London Convention (IMO, 1972) that "*prohibits the dumping of certain hazardous materials in the sea and requires a prior special permit for the dumping of a number of other identified materials and a prior general permit for other wastes or matters*". Several international convention agreements have followed (Abriak et al., 2006) and consequently, laws and directives have been created across the world with obligatory procedures in place before dredging is authorised. These include for instance: an evaluation of sediment contamination; framing of contaminated sediment disposal and remediation; justification of dredging methods used; agreement for the follow-up monitoring of the dredged site.

Various EU Directives exist to protect habitats, water and the environment. Whilst none of these address the dredging process directly, some of them have an impact on dredging projects through international conventions and guidelines, which prevail on EU law and impact on marine dredging activity (Mink et al., 2006). The EU's Water Framework Directive (WFD) requires performing a WFD assessment for all activities that take place within the water body (European Council, 2000). This assessment aims at evaluating how the dredging work would impact water status and habitats locally. The EU's Waste Framework Directives deal with the management of dredged sediment while the Habitat and Birds Directives have indirect consequences on dredging projects, which are located near protected sites, forcing higher monitoring requirements and increasing their cost (Mink et al., 2006).

Still in the EU, for the management of dredged sediment specifically, several disposal or recycling options are given depending on the physicochemical condition of the sediment, especially its contamination state. For uncontaminated sediment, a beneficial use is usually targeted. Possible disposal solutions include sea deposit, using the sediment to support sediment-based habitats, shorelines and infrastructures, for habitat restoration such as wetlands, coastal features, beaches or even engineering use for example as capping material (OSPAR Commission, 2014). For contaminated sediment, however, the re-use is strictly regulated, and options can only be considered after a decontamination treatment if the sediment then meets the specific requirements. If sufficient remediation cannot be achieved, contaminated sediment can be disposed in a Contained Disposal Facility (CDF), a Contained Aquatic Disposal (CAD) or most often at a landfill site. Such disposal is very expensive and usually constitutes the main part of a dredging project's budget (Palermo and Hays, 2014).

In parallel to the implementation of laws aimed at legislating dredging operations, efforts have been made to develop tools and methods of management to match the new regulations (Cooper, 2013). Different organisations such as the Central European Dredging Association (CEDA) or the Permanent International Association of Navigation Congresses (PIANC) provide resources for the selection of dredged-sediment management solutions. For the North East Atlantic, "Guidelines for the Management of Dredged Material at Sea" are described by OSPAR, with the most updated version dated from 2014 (OSPAR Commission, 2014). For dredging projects in general a wide range of concepts and decision-making frameworks have been proposed (Bates et al., 2015; Manap and Voulvoulis, 2014; Palermo et al., 2008) in an attempt to limit and reduce the environmental consequences. The complex legislation and the negative public perception of dredging make managing the process a challenge (Cutroneo et al., 2014; Hamburger, 2002). Conflicts can appear between the different stakeholders and projects are consequently subjected to delays or cancellation.

A further significant issue with dredging is its high financial cost, comprising the cost for the operation and the cost for the disposal. The cost varies widely depending on the technology and equipment used, as well as the volume of sediment targeted, frequency of operations, the distance to the disposal site and the presence of contaminants. As an example, in 2005, 30 million cubic meters of sediment were dredged from the Dutch ports, of which 2 million cubic meters had to be disposed in CDFs due to their contamination levels, the rest of it was dumped in the North Sea. The cost related to the disposal of the contaminated sediment was estimated around 20€ per m^3, whereas for non-contaminated sediment it was 5€ per m^3, giving an extra cost of 30 million euros per year only for the disposal of contaminated sediment (Walker et al., 2011). Moreover, since ports and harbours are adapting to enable the entry of larger vessels, the need for dredging increases and in consequence, so does the associated cost (Kirby, 2011; Manap and Voulvoulis, 2015). Exact costs of maintenance dredging for European ports are difficult to obtain, more data can however be obtained from the U.S. Army Corps of Engineers which demonstrate a high variability of cost between location and the increase of the cost over years, independently of the dredged volume. A report showed maintenance dredging costs between 2014 and 2018 varying from 2.84€ per m^3 in New Orleans to 26.34€ per m^3 in San Francisco (Frittelli, 2019). The same report showed an increase in mean maintenance dredging costs in the US over years, going from 1.89€ per m^3 in 1970 to 6.26€ per m^3 in 2018 which was attributed to numerous factors including inflation, lack of competition for dredging contracts and changes in the disposal of dredged material (Frittelli, 2019).

1.2. Alternative sediment management methods

Considering the environmental impact, the cost, the constraining legislation and the conflicts related to dredging, research has been undertaken to find alternatives (Bianchini et al., 2019; Kirby, 2011). Most alternatives can be defined as "anti-siltation methods", as they are designed to prevent sediment from accumulating in the targeted area. The major advantage of this kind of method is that a significant part of the issue disappears, since there is no need for disposal and no need for a dredging licence, although all sediment management projects are subject to approval.

1.2.1.Overview of alternatives to dredging

The 43rd PIANC working group reviewed the different methods used as an alternative to dredging for sediment management in ports and harbours (Kirby, 2011). They categorised the techniques into three groups: Keep Sediment Moving (KSM), Keep Sediment Out (KSO) and Keep Sediment Navigable (KSN), also grouped as "sand by-passing plants", "anti-sedimentation structures" and "remobilising sediment systems" in a more recent review (Bianchini et al., 2019). A wide range of techniques have been created to adapt to specific situations but can nevertheless serve as useful examples. However, some of them can be considered as generic and they could be applied to different harbour configurations. A summary of the methods is displayed in Table 1. A detailed assessment of the environmental impact and cost of most of these technologies can be found in Bianchini's review (2019), where it is concluded that these alternative technologies cost on average 30% less than traditional dredging.

Keeping sediment out usually involves the design of structures that will physically prevent siltation by altering the effect of waves, currents and sand movement. These structures have been stated to be less efficient for fine-grained sediment, particularly cohesive clay (Bianchini et al., 2019). Anti-sedimentation structures have been well described and comprise, sand traps, seawalls, current deflection walls (CDW), or even pile groynes (Bianchini et al., 2019; Kirby, 2011). It should be noted that these structures can potentially have negative impacts on the surrounding environment if they are not designed carefully, as modification of wave patterns can impact near-shore processes with a detrimental consequence for wildlife and ecosystems . An example of CDW is shown in Figure 1.

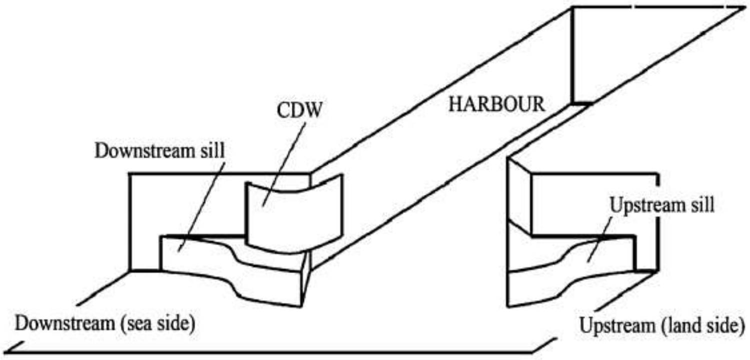

Figure 1: Current Deflection Wall as built in Delft Tidal Flume *(Hofland et al., 2001; Kirby, 2011)***.**

The second group of techniques, KSM, regroups the two categories called "sand by-passing plants" and "remobilising sediment systems" by Bianchini *et al*. (2019). Sand by-passing plants function by transferring the sediment out of the channels, therefore preventing siltation occurring in the first place. This is contrary to dredging, which happens after siltation occurred (Bianchini et al., 2019; Kirby, 2011). The physical transfer of sediment is performed through different pumping systems, which are adapted to port configurations (Bianchini et al., 2019). In Leer, for example, slopes were created in the docks, so that gravity naturally leads the sediment to flow into a collection sump where an underwater pump collects it and discharges it into the estuary (Figure 2) (Kirby, 2013, 2011). Remobilising systems, however, involve the resuspension of the sediment in order to put it back into the current for its evacuation from the blocked areas. The most well-known method being water injection dredging (WID), which uses a water-jet towards the seabed to create a density current which picks-up the sediment and takes it to a lower point (Bianchini et al., 2019).

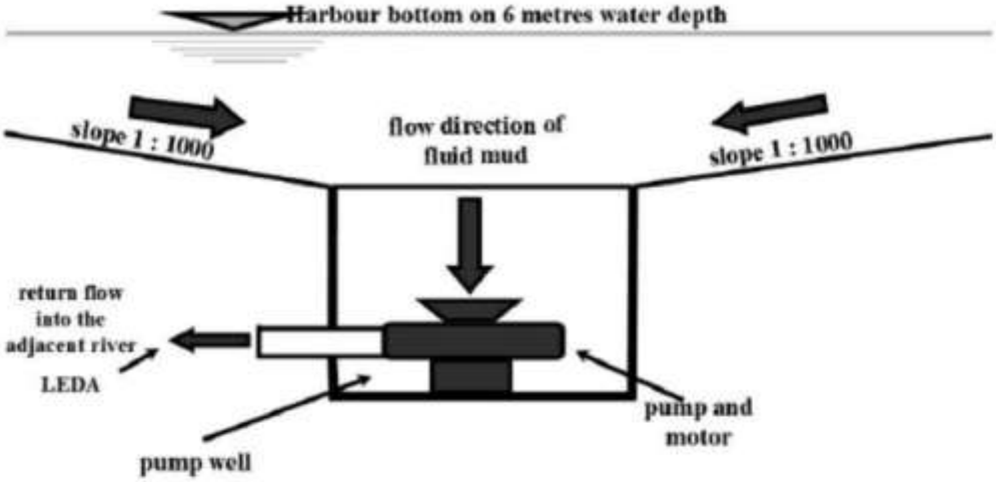

Figure 2: Auto-flushing system as applied in Leer *(Kirby, 2011)***.**

The last category described in Kirby's review (2011), KSN, is comparable to the remobilising systems but differs in the point that it does not aim at evacuating the sediment from the port or harbour, but instead relies on the fact that some sediment types are navigable when brought into suspension (Kirby et al., 2008; Welp and Tubman, 2017). Keep sediment navigable works on the concept of nautical depth and mostly involves the method called Active Nautical Depth (AND). It is a method emerging from "Passive Nautical Depth" (PND), which is a different way to define the depth in ports and harbours, using density parameters. Active Nautical Depth derives from this concept by the fact that fluid mud is created *in situ* by mixing and aerating the mud at the bottom of the water column which makes it navigable and therefore increases the nautical depth.

KSN techniques form the focus of this review and the only representative of this group, Active Nautical Depth, which could be used to couple sediment management and bioremediation in muddy ports is detailed in the next section.

Table 1: Sediment management alternatives.
Compilation of the alternatives to dredging as reviewed by Bianchini et al. (2019) and Kirby et al. (2011) and comment on their suitability to deal with sediment contamination and on their sustainability with regards to sediment management.

Category as stated in the literature	Principle	Technologies	Sustainability	Ability to deal with sediment contamination	
				Advantage	inconvenient
Keep Sediment Out / Anti-sedimentation structures	Using structures to physically prevent sediment from entering and blocking ports, harbours, and channels	Sand traps (1)	High - Structures staying in place for years	NA	
		Seawalls (1)			
		Defection walls (1)			
		Piles groynes (1)			
Keep Sediment Moving / Remobilising sediment systems	Resuspending the sediment in a current that takes it out of the blocked areas	Water injection dredging (1)	Low – Techniques to repeat on a regular basis	Sediment resuspension could favour contaminant (bio)degradation (3,4)	Strong spreading of contaminants
		The Neptune (1)			
		Fluidization plants (1)			
		Submarine sand shifter (1)			
		Turbo units (1)			
Keep Sediment Moving / Sand by-passing plants	Using pumps to constantly transfer the sediment out of the channel through piping systems	Centrifugal pump (1)	Moderate/High – not always fixed and require maintenance		
		Jet pump(1)			
		Punaise pump (1)			
		Auto-flushing system (2)			

Keep Sediment Navigable	Resuspending sediment to make it navigable	Active Nautical Depth (2)	Moderate – Technique to repeat regularly, frequency reduced by EPS production	Resuspension + aeration can strongly favour contaminant biodegradation (3, 4, 5, 6, 7, 8, 9, 10)	Moderate spreading of contaminants

X (number of reference): 1 = (Bianchini et al., 2019), 2 = (Kirby, 2011), 3 = (LeBlanc et al., 2006), 4 = (Pourabadehei and Mulligan, 2016), 5 = (Beolchini et al., 2014), 6 = (Fahrenfeld et al., 2013), 7 = (Levi et al., 2014), 8 = (Schurig et al., 2014), 9 = (Wald et al., 2015), 10 = (Wang et al., 2016)

1.2.2.Passive and Active Nautical Depth

1.2.2.1. Principles

The application of the concept of Passive Nautical Depth has been one of the first steps implemented by ports and harbours around to world to reduce dredging need. It consists of changing the criteria for defining the nautical bottom. The nautical bottom is defined as the level at which the physical characteristics of the bottom can cause either damage or unacceptable effects on controllability and manoeuvrability by contact with a ship's keel (Kirby, 2011; McAnally et al., 2016). Before the application of this concept, the depth was measured with a fathometer, which records the time for a sound pulse to be reflected from the bottom and back to the device. Depending on the rheological parameters (*e.g.* density, viscosity) of the sea bottom (especially in muddy bays and estuaries), the fathometer generates ghost echoes that can either be associated with a solid bed or with fluid mud that would be navigable. None of the instruments used are able to differentiate ghost echoes from real solid bed (McAnally et al., 2007). By precaution, ghost echoes are always considered to be associated with a solid bed, which leads to a potentially unnecessary dredging of the fluid mud, resulting in needless expense and additional pollution that could be avoided.

When applying PND, the depth should be defined by the parameters that permit discrimination between a solid bed and fluid mud. The density criterion is generally used but density alone is not sufficient. Other parameters, such as shear stress, should be considered to establish whether the mud is fluid enough to be navigable (Wurpts, 2005). These parameters, however, are not easy to record routinely and different particle size arrangements (which are locally variable) also influence density, shear strength and therefore navigability. As a consequence, for each port the density at which the sediment is in a fluid mud state has to be determined. In muddy ports with low sand content, the most often used density threshold is 1,200 $kg.m^{-3}$ (Welp and Tubman, 2017). The concept of Passive Nautical Depth is now widely used in the world's ports and harbours with the advantage of reducing dredging use (McAnally et al., 2016). Whilst tackling the physical problem, however, it does not deal with the issue of chemical contaminants.

By derivation from the PND concept, an alternative method to manage sediment in muddy ports and harbours has been developed, called Active Nautical Depth (Kirby et al., 2008; McAnally et al., 2016).

The principle (see Figure 3) is to manipulate the fluid mud cloud to perpetuate its navigability by mixing and aerating it. Aeration is a critical step that determines the sustainability of the method. Indeed, the new aerobic state of the mud promotes the growth of aerobic microorganisms that start producing large amounts of extracellular polymeric substances (EPS). EPS are compounds, mainly polysaccharides and proteins but also DNA, excreted by bacteria to form a gel-like matrix in which cells are aggregated and immobilized and which has a main role of protection but is also favourable to communication between cells or carbon storage (Costa et al., 2018; Wingender et al., 1999). The production of EPS allows the cells to grow in a community called biofilms, or flocs at smaller scale, as opposed to their free-floating life or planktonic form. After AND, without EPS production, the mud would rapidly go back to its initial non-navigable state but with EPS the particles are kept in suspension longer (Pang Qi Xiu et al., 2018) and the fluid remains navigable for weeks. The physical properties of EPS also permit the hulls of vessels to pass through with minimal friction, thus facilitating navigability through the fluid mud cloud (Kirby et al., 2008).

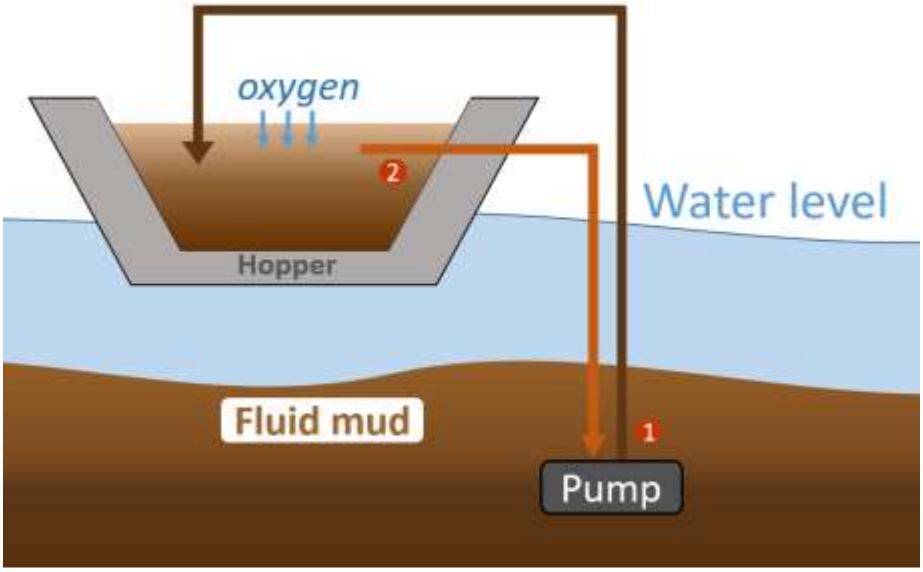

Figure 3: Active Nautical Depth Principle (as applied in Emden).
Muddy sediment is pumped into a hopper dredger (1) where it is aerated before it is pumped back to the see bottom (2).

1.2.2.2. AND current application and potential worldwide applicability

Emden port (Ems estuary, Germany) was the first to experiment with AND in 1990. The method has been successfully applied and is well described in the literature (Kirby, 2011; McAnally et al., 2016; Wurpts, 2005) In this case, mixing is achieved by pumping the fluid mud with a low-power submerged dredge pump into a hopper dredger (see Figure 3). The pumping initially alters the physical conditions by breaking the inter-particle bonds and fluidizing the mud. This mud goes in the hopper and is exposed to the atmosphere, thus passively becoming aerobic in a few minutes and ready to be placed back to the sea bottom. The fluid mud cloud remains in suspension for 3-4 months before the mixing episode has to be repeated (Kirby et al., 2008). In Emden’s port configuration, the fluid mud cloud maintained by AND prevents exterior sediment from re-entering the basin, consequently reducing the need for dredging to zero where previously 4 million m^3 of sediment was dredged each year. Finally, as a result of the reduced need for maintenance dredging, the overall cost of sediment management decreased from €12.5 million per year to €4 million per year (Kirby, 2013).

Based on the successful results obtained following the implementation of AND in Emden port, an investigation of its potential to be up-scaled and used in other ports and harbours worldwide has been performed (Wurpts, 2005). There are some critical conditions necessary for AND to be successful and these include sediment particle size. A muddy substrate with low sand content is required in the targeted area. According to Wurpts (2005), AND should easily be applicable for a sand content of up to 10% with a particle size of between 60 and 200 μm. For sediment with a sand content exceeding 10%, however, the process can be refined. Indeed, the hopper dredger applied in Emden port has been designed in such a way that a sand extraction can be performed if needed.

Wurpts (2005) evaluated that these application conditions were viable for several ports in Europe, such as Bristol, Liverpool, Rotterdam, Brunsbuettel, Harwich, and Leer. In theory, many ports with muddy sediment could successfully use AND, feasibility studies must be performed on a site basis to evaluate the possibility of applying it as a sustainable method for sediment management (to replace or reduce dredging). McAnally *et al.* discusses in a review (2016) the possibility of applying PND and AND concepts in the U.S. waterways and concluded as well that these are theoretically applicable to many locations such as Gulfport, Mississippi, Atchafalaya, Louisiana, and Calcasieu, Louisiana but studies need to be undertaken to confirm it and bring it to application.

Apart from sediment particle size, other factors could be taken into account, such as nutrient quantity. Despite stating that a low nutrient level was optimal for the excretion of EPS in high amounts, Wurpts did not determine the extent of nutrient concentration influence on AND applicability (Wurpts, 2005). Bacteria secrete EPS and form biofilm communities in order to survive in harsh environmental conditions. Nevertheless, flocculation can still be observed in the case of nutrient-rich environments (Lai et al., 2018), indeed, biofilms confer many advantages to bacteria, they offer protection against predation, a better resistance to UV, to high concentrations of toxic compounds and to changes in salinity or pH (de Carvalho, 2018). More research could be done to determine if nutrient loads would influence AND as if EPS production is insufficient, the sedimentation of the fluid mud could happen too quickly after mud fluidization and reduce the sustainability of the method.

In addition to Emden AND was also applied in Delfzijl and Bramerhaven where the process was slightly modified. There, instead of pumping the mud into a hopper dredger, surface water is pumped into the mud to fluidize and aerate it (Nasner et al., 2007). These AND applications should not be confused with water injection dredging, which uses a similar concept but with a high-pressure water jet aimed at flushing the mud out of the location, whereas in the case of Delfzijl and Bramerhaven the aim is only to create a navigable fluid mud cloud through low power injection.

2. Port contamination and sediment remediation

2.1. Sediment contaminants

Port and harbour activities generate many types of pollution: petroleum and its derivatives, greenhouse gas emissions, release of compounds from antifouling paints, sewage, and wastewater. The multiple sources of contamination and the usual enclosed configuration of ports and harbours result in limited circulation leading to high levels of contaminants in sediments and subsequent negative impact to aquatic life due to their toxicity. The presence of contaminants usually damages the ecosystem locally by affecting the development, reproduction and survival of many indigenous species. There are countless examples of evidence for the toxicity of pollutants found in ports and harbours. Tributyltin (TBT), for example, previously used in antifouling paints, is well-known for its endocrine disruptive action, first discovered by the appearance of malformations leading to the decrease in oyster populations, which caused severe problems to the oyster production market of the Arcachon Bay in France in the 1970's (Alzieu, 2000). Since then, the knowledge on TBT's high toxicity has increased and it is commonly considered to be the most toxic substance deliberately delivered into the aquatic environment. Heavy metals also exert their toxicity in various organisms, they damage the tissues and DNA leading to numerous problems like growth inhibition, deformities or reduced fertility (Sharifuzzaman et al., 2016).

In addition to their local impact, several contaminants, like polycyclic aromatic hydrocarbons (PAHs), heavy metals and organotin compounds (OTCs), are known to be bioaccumulative, which means that they can be transported along the food chain, affecting a wide range of organisms and can potentially be toxic towards humans (de Carvalho Oliveira and Santelli, 2010; Nikolaou et al., 2009; Sharifuzzaman et al., 2016). Some of this pollution causes reversible damage; the contaminants degrade rapidly after introduction into the environment and are therefore defined as non-persistent, which is the case for fertilizers, domestic sewage, or non-persistent pesticides. On the contrary, other contaminants are called persistent, because the damage that they cause is either irreversible or they persist over a long time periods. The main contaminants persisting in sediment are OTCs, heavy metals, polychlorinated biphenyl (PCB) and PAHs.

2.2. Sediment remediation

2.2.1. Traditional remediation

The vast majority of the methods designed for the remediation of contaminated sediment (Table 2) involve dredging and placement *ex-situ* followed by a designated treatment. Most of the available treatments are physical and chemical. Thermal treatment such as incineration, as an example of physical treatment, is often used because of its efficiency but it consumes a lot of energy and has a high cost (Du et al., 2014). A classic chemical treatment is chemical oxidation, which uses oxidants such as Fenton's reagent, potassium permanganate or hydrogen peroxide to break down contaminants. It has been raised however that incomplete reactions or side reactions may occur during chemical treatments, leading to the release of other potentially toxic compounds (Ferrarese et al., 2008; Finnegan et al., 2018).

2.2.2.Sediment bioremediation

Efforts have been made to find more environmentally friendly and cost-effective ways for the remediation of dredged contaminated sediment and bioremediation is an encouraging process in this regard. Bioremediation consists of the degradation of a contaminant as a result of the activity of a living organism. It usually involves contaminant breakdown by microorganisms (biodegradation) or by plants (phytoremediation).

Bioremediation has been applied successfully as an *ex-situ* treatment for contaminated sediment (Chikere et al., 2016; Novak and Trapp, 2005; Rocchetti et al., 2014; Wu et al., 2014). Used *ex-situ*, however, it is still associated with the negative effects of dredging described above (*e.g.* strong environmental impact, complex legislation, high cost) and remains unsustainable as the sediment is removed from its initial location. Consequently, developing *in-situ* solutions that do not require dredging for the remediation of contaminated sediment are most desirable.

Several options have been proposed for *in-situ* bioremediation of contaminated sediment, the simplest one being natural attenuation, which consists of leaving the environment to decontaminate itself and only monitoring the progress of degradation (Lofrano et al., 2017). Natural attenuation is usually a slow process and can be applied for low-risk contaminants. Biostimulation and bioaugmentation can therefore be used to boost the process of natural attenuation. Biostimulation involves the stimulation of the native degrading community by creating more favourable conditions for the growth and activity of the microorganisms. This can be achieved, for example, by the addition of nutrients or oxygen. For bioaugmentation, microorganisms identified to be efficient at degrading a targeted contaminant are added to the native community. Biodegradation has been widely studied at the laboratory scale. Studies have been assessing the biodegradability potential of sediment contaminants by a precise microorganism in pure culture or mixed culture (Dean-Ross et al., 2002; Harrabi et al., 2019; Khanolkar et al., 2015; Mulla et al., 2018; Y.-S. Wang et al., 2015). In an attempt to mimic more accurately the environmental conditions, microcosm experiments were set up using spiked or naturally contaminated sediment (Demirtepe and Imamoglu, 2019; Levi et al., 2014; Matturro et al., 2016; Peng et al., 2019; Wang et al., 2016; Z. Wang et al., 2015; Yang et al., 2015).

In both culture media and microcosms, biostimulation and bioaugmentation approaches have been tested to determine the optimal conditions of degradation. Bioaugmentation was sometimes shown to be efficient to enhance biodegradation regardless of the conditions (Dell'Anno et al., 2009; Li et al., 2015), while sometimes showing no effect on degradation rates (Demirtepe and Imamoglu, 2019). Wang *et al.* (2015) tested the effect of bioaugmentation using different strains isolated for their nonylphenol biodegradation ability and observed a positive impact on nonylphenol biodegradation in microcosms for only one of them. Another study assessing the impact of bioaugmentation on perchloroethylene (PCE) biodegradation in different microcosms using sediment from various sites also reported different levels in bioaugmentation efficiency (Schiffmacher et al., 2016). In the latter study, the authors explained the contrasting results by the presence of diverse co-contaminants in the different sites, leading to variable degradation pathways that do not necessarily lead to the complete elimination of the toxic compounds. Other factors, however, play a role in the success or failure of bioaugmentation attempts, such as the ability of the bioaugmented strain or population to adapt to the target environment and to compete with the indigenous microorganisms (Mrozik and Piotrowska-Seget, 2010).

Biostimulation attempts also give varying results. Nutrient addition sometimes effectively enhances biodegradation (Demirtepe and Imamoglu, 2019; Tang et al., 2019; Ye et al., 2013) but can also inhibit it in some other cases (Z. Wang et al., 2015; Wong et al., 2002). Biostimulation normally aims at boosting the growth of microorganisms in order to obtain better degradation activity, but providing a source of carbon or energy that is more readily available can also result in its preferential use, to the detriment of the target toxic compound degradation (Wong et al., 2002). The other biostimulation approach, consisting of providing oxygen to favour aerobic metabolism, which holds degradation pathways of numerous contaminants, often has a positive impact on biodegradation rates. Several authors reported the aerobic biodegradation of contaminants such as pesticides (bentazone, dichlorprop, mecoprop, glyphosate), PAHs, alkanes, phthalate acid esters (PAEs), 2,4,6 trinitrotoluene (TNT), organotin compounds and nonylphenol in microcosm experiments involving sediment (Beolchini et al., 2014; Fahrenfeld et al., 2013; Levi et al., 2014; Li et al., 2015; Wald et al., 2015; Wang et al., 2016; Z. Wang et al., 2015). Other studies focussed on assessing the aerobic biodegradation of contaminants by specific microorganisms in pure culture, which is also useful in a potential bioaugmentation approach (Cruz et al., 2007; Mulla et al., 2018; Y.-S. Wang et al., 2015). Even more interestingly, the beneficial effect of resuspension on the biodegradation of heavy metals and phenanthrene was reported (LeBlanc et al., 2006; Pourabadehei and Mulligan, 2016). These studies are of particular interest for the purpose of this review, as they demonstrate a beneficial effect of the processes involved during AND (resuspension, aeration) on contaminants biodegradation.

2.3. Research needs in the field of sediment bioremediation

All of the studies described in the previous section have improved the knowledge on sediment contaminant biodegradation with the aim of developing novel bioremediation solutions, but after years of research at the laboratory scale, there is still a clear lack of pilot-scale studies in the field and actual applications (Majone et al., 2015; Perelo, 2010). Other innovative techniques have been proposed, often hybrids between physical, chemical and biological treatment, they include, for example, reactive capping, reactive barriers, or bioelectrochemical removal (Lofrano et al., 2017; Majone et al., 2015). Recently, a field trial reported the successful use of immobilised microbial activated beads for the *in-situ* remediation of river sediment aimed at reducing nitrogen and organic carbon pollution (Fu et al., 2018). This study, however, represents an exception, and reviews of *in-situ* bioremediation highlight the lack of application of the proposed methods, which are rarely brought to field trials, despite their promising potential (Lofrano et al., 2017; Majone et al., 2015). This lack of application is explained by several factors. There is a lack of consensus for the use of *in situ* bioremediation, due to uncertainty about the effectiveness, control and possible secondary effects. A need for the development of biomolecular tools for site investigation has also been emphasised (Majone et al., 2015). More research is consequently needed to overcome these barriers.

Table 2: Overview of remediation solutions for contaminated sediment.

Type of method	Advantage	Disadvantage	Examples of technology used
Physical	• Very effective • Suitable for high levels of contaminants • Fast remediation	• Expensive • Energy consuming • Mostly applied *ex situ** • *Strong perturbation of sediment biology and physico-chemistry*	Incineration
			Immobilization
Chemical	• Effective	• Can involve side reactions • Only applied *ex situ**	Solvent extraction
			Chemical oxidation
Biological	• Environmental-friendly • Cheaper • Can be applied *in situ**	• Involve long durations • More efficient for low/moderate contamination • Lack of full-scale application	Phytoremediation
			Biodegradation (natural attenuation, biostimulation, bioaugmentation)

**ex situ treatments of sediment involve dredging and all the detrimental issues associated with it.*

3. Potential coupling of sediment management and bioremediation

Sediment management techniques like AND, which use the resuspension and aeration of sediment without transportation, could serve a double objective. In fact, as a beneficial side effect, the aeration and resuspension of the mud may favour bioremediation of sediment pollutants while reducing the production of other pollutants such as methane, ammonia, or hydrogen sulphide by anaerobic microorganisms. Using AND for the bioremediation of contaminated sediment could be a good option since it would be applied *in situ* and therefore would not involve spreading of contamination or further pollution during transportation. Further research is necessary to evaluate specifically the potential applicability of AND for the remediation of contaminants found in ports and harbours. Ideally the aim would be to target a wide range of compounds to make AND a versatile method to manage and remediate sediment in multiple places around the world but a first step in the investigation is to understand the factors contributing to degradation of a single contaminant.

AND as it is used currently in Emden already caused major savings in the sediment management budget of the port, the bioremediation part of it could be a passive benefit of the method and would therefore involve no extra cost. If it was revealed as efficient as it could be by analysing the literature, this would make it a very attractive technique to solve two major issues in the port industry.

As promising as it looks, using AND or a derivative for the bioremediation of a harmful contaminant would nonetheless be subject to critical scrutiny. Resuspending sediment certainly constitutes a perturbation of the port ecosystem, mostly because of the turbidity caused by the fluid mud cloud. Note that this turbidity is more localised than the one observed during dredging excavations, the fluid mud is pumped back to the sea bottom where it forms a layer of navigable mud without mixing with the above water. It is, however, important to note that ports are by essence perturbated environments, with ship traffic, maintenance work and contamination, the ecosystem is often disturbed (Darbra et al., 2005). Such a method being used as a replacement for dredging could still mitigate the disturbance as all the issues linked with transportation and disposal are eliminated.

The use of AND for the bioremediation of strongly contaminated locations would need to be approached with caution as this could lead to the release of toxic compounds into the surrounding waters for a certain period of time before contaminant biodegradation. Indeed, it could be argued that causing a strong perturbation of the ecosystem in order to sustainably clean an area might be an acceptable compromise compared to leaving these highly contaminated locations as they are but facing regular resuspensions and perturbations caused by ship traffic or natural events. Some remediation methods such as capping are especially designed to tackle this kind of issue, but they are only suitable for contaminants that are degraded anaerobically, and therefore cannot be applied to a wide range of contaminants.

The fact that some contaminants are specifically degraded in different conditions of oxygenation also complicates the development of bioremediation solutions as they consequently must be adapted to the local ’cocktail of contaminants’. If resuspension-aeration techniques were to be applied to a site where aerobically degraded contaminants are present alongside anaerobically degraded contaminants in high quantity this would lead to the resuspension of the latter without any hope of future degradation, which would represent a bigger threat to the ecosystem and make the remediation effort counterproductive.

AND or equivalent techniques of resuspension-aeration could therefore find their best value when actually used routinely as management methods, eliminating moderate levels of contamination as they are introduced in ports and harbours through the inherent activities and preventing their accumulation to toxic levels, while preventing sediment accumulation in the navigable waterways.

Conclusion:

After using dredging for years to tackle siltation in ports and harbours it is widely acknowledged that this method of sediment management has many flaws with high environmental impact and significant costs, especially when dealing with contaminated sediment. Several methods have been proposed as alternatives to dredging, these have not replaced dredging which remains the most widely used technique. These alternative methods are based on different principles, preventing sediment from entering the target areas, resuspending it into a current, repeatedly pumping it out or making it navigable. Separately, a substantial research effort was made to improve the knowledge on bioremediation of contaminated sediment and these studies emphasize a strong contaminant biodegradation potential within the microbial community at the laboratory scale. Nevertheless, there is a clear need to advance the research to the next steps with field-scale pilot studies. More importantly, this review highlights the beneficial potential to rethink sediment management and bioremediation solutions in an integrated way, especially for contaminants that are biodegraded aerobically. Techniques such as AND could, in addition to reducing the need for dredging in muddy ports and harbours, biostimulate native microorganisms and could result in the elimination of harmful compounds such as PAHs, organotin compounds, various pesticides or herbicides.

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