Nordic Council of Ministers

Contaminated Sediments:

Review of solutions for protecting aquatic environments

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Preface

This report is written on behalf of the Nordic Council of Ministers, The Marine Group (HAV). The literature review, compilation of data and writing of the report is the result of a collaboration between The Norwegian Institute for Water Research (NIVA), The Department of Earth Sciences, Uppsala University in Sweden and The Finnish Environment Institute (SYKE), with the collaborators contributing information from their respective countries. NIVA has taken lead of the project with Research Manager Marianne Olsen as project manager. Scientists contributing to the report have been:

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  - Research scientist Dr. Karina Petersen, NIVA
  - Senior research scientist Morten Schaanning, NIVA
  - Research scientist Sigurd Øxnevad, NIVA
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  - Senior research scientist Matti Leppänen, SYKE.

A reference group with representatives from stakeholders has consisted of Bernt Malme, Hydro (Norway), Yvonne Ohlsson, Swedish Geotechnical Institute (Sweden), Jens Laugesen, DNV GL (Norway), Jesper H. Andersen, NIVA Denmark (Denmark), Jarkko Akkanen, University of Eastern Finland (Finland), Anders Karlsson-Drangsholt, Bellona (Norway) and Rebecca Gardner, Anchor GEA (USA).

The reference group has provided advice to the project, with comments on the draft outline and the draft of the final report. We thank the reference group for their contribution.

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Oslo, 23. December 2018
Marianne Olsen
Project manager

Summary

The Nordic Council of Ministers has recognized the need to compile the knowledge of sediment remediation strategies and to evaluate the different approaches in relation to cost, efficiency and adverse effects on the environment, to ensure that the best possible measures are taken in the future. The objective of this report is to give a status of regulatory framework and sediment management within selected Nordic countries (Norway, Sweden, Finland and Denmark). In addition, the report gives a short overview of international knowledge on remediation approaches, techniques and materials.

Contaminated sediment within the selected Nordic countries

Contaminated sediment has been recognized as a significant reservoir of legacy contaminants in industrial sites worldwide, potentially contributing with releases of contaminants for decades after industrial releases have been stopped. Further, contaminated sediments have been recognized as a potential source for transfer of contaminants into aquatic food chains. Worldwide, a number of sites have been identified and assigned priority in national programs for remediation. However, governance and regulatory instruments in dealing with contaminated sediments vary between countries. In Norway, the Norwegian environmental authorities have implemented a national strategy for remediation of contaminated marine sediments within harbors and fjords, which have resulted in the identification of prioritized sites, the development of action plans and the initiation of several remediation actions. The other Nordic countries do not have any corresponding national strategy.

Classification of sediment based on concentrations of defined contaminants or groups of contaminants is commonly used as a tool to identify sites of concern. The classification can be based on toxicity and potential environmental risk or on exceedance of defined thresholds or background levels. The Water Framework Directive (WFD) is implemented in EU member states and other European countries with the aim to achieve "good status" for all ground and surface waters in the EU, based on Environmental Quality Standards (EQS) developed and implemented under the WFD. In Norway, chemical status is primarily based on the national set EQSs for sediment (set for 28 EU priority substances) and biota (set for 23 EU priority substances), whereas in Finland the chemical status of water bodies is based on the EQS set for surface water and biota. Finland is not applying any sediment EQS values. In Denmark and Sweden, the definition of good chemical status is based on EQS for water, sediment and biota, though in Sweden EQS has been defined only for a few substances in biota and sediment.

Contaminated sediments in Norway are principally related to harbours and fjords with industrial activity in the form of process industries, pulp and paper industry and shipyards, as well as shipping. Today, industrial emissions are strictly regulated, while the significantly higher emissions of earlier times have led to contaminated sediment in many fjord recipients. Natural recovery through natural oversedimentation is usually recognized where clean particles settle on top of older polluted seabed, but the sedimentation rates vary and depends on site-specific conditions. Investigations up to the end of the last millennia revealed that sediments in more than 120 sites within the fjords have high concentrations of hazardous substances. Clean-up of contaminated seabed is a priority for Norwegian authorities and has been so since the late 1980s. For 17 prioritized fjords, regional action plans for contaminated seabed have been prepared. Following this action, the Norwegian Environment Agency has given orders to local industries on further investigations and development of site-specific action plans for their aquatic recipients, and in some cases also orders for implementation of measures. Clean-up measures undertaken as result of the regional action plans typically take place in close cooperation with the local, regional and/or national environmental authorities. In the years after 2000, governmental funds have been allocated annually for remediation of contaminated sediments and contaminated soil. To support the work on remediation of contaminated sediments, the authorities have prepared a set of guidelines, including Guidelines for handling sediments and Guidelines for risk assessment of contaminated sediments, where the process for assessing the need for measures are described.

In Sweden, a preliminary review of the type and occurrence of contaminated sediments identified in inland and/or coastal waters within each of Sweden's 21 counties in 2016, revealed that contaminated mineral-based (minerogenic) and/or cellulose-bearing ("fiberbank") sediments occur in at least 19 counties. At many sites, sediment contamination likely poses unacceptable risks to the environment and/or human health although less than a handful of management decisions have been taken. The Swedish Environmental Protection Agency (Narturvårdsverket) is the governmental agency responsible for environmental issues, and more precisely for coordinating, prioritizing and following up the work on environmental issues at the national level. They work together with other governmental agencies. The Swedish Geotechnical Institute has the national responsibility for research, technological development and knowledge building for remediation and restoration of contaminated sites. The Swedish EPA has decided that polluted areas without any responsible parties will be taken care of by government funds. In 2017, the government decided to distribute funds for soil and sediment remediation between 2018 and 2020. Sweden's most extensive contaminated sediment remediation project is Oskarshamn's harbour, which had a very active industrial history since the middle of the 1980s. A remediation effort started in 1996, and dredging operations began in 2016.

In Finland, the status of sediment contamination is only assessed for relocation purposes after dredging. There have not been systematic and nationwide surveys in Finland to identify contaminated sediment, though a preliminary national survey of contaminated sediments in inland waters lists 28 possible or known sites across the country. Only a few of them have been remediated. There is no guidance to assess environmental hazard and the only sediment related guide is for disposal of dredged sediment. The Ministry of Environment is the leading environmental administrative body, setting guidelines and policy for implementation of EU directives and national legislation. The environmental legislation is put into force via the practical work of the Regional Centres for Economic Development, Transport and the Environment (ELY) or even at city or municipality level. The chemical status of water bodies, as based on the WFD and amendment directives, is followed and classified by the regional ELY centres. The management of sediments by environmental authorities is most often related to dredging actions performed for keeping waterways open for navigation or constructions at harbours. Therefore, the use of waterways and the aquatic environment is the driving force for these actions, not the management of contaminated sediments i.e. environmental protection.

In Denmark, the coastal waters are heavily affected by anthropogenic activity, both from land- and ocean-based activities like aquaculture, shipping and industry. Dredging appears to be the main method for removal or handling of (contaminated) sediments in Denmark. The content of hazardous substances is included as an essential element in the evaluation of how sediments and dredging material can be handled. Hazardous substances in Danish marine waters have been monitored on a nationwide scale since 1998 through The Danish National Monitoring and Assessment Programme for the Aquatic and Terrestrial Environment, NOVANA. The Ministry of Environment and Food of Denmark is responsible for the water planning in Denmark, and for monitoring the condition of surface and ground water and the protected areas. The inner Danish waters are in general classified as problem areas in terms of chemical status. A review of Danish sediment data using thresholds commonly used in OSPAR and the countries of the study area revealed that 36% (28 sites) of the assessed areas had a high or good chemical status. Most of the assessed areas had a moderate chemical status (55%, 42 sites), and just 7 sites (9%) got the score bad or poor (NB: provisional calculations, not an official Danish assessment).

Short overview of remediation approaches

Historically, the most commonly used approach has been removal of the contaminated sediment by dredging and excavation and disposal/treatment off site. Isolation of contaminated sediment by capping with and without an active barrier has also been implemented in some countries. Other approaches including use of thin layer amendment with active materials, *in situ* stabilization, and monitored natural recovery (NMR) have been developed but to a lesser degree implemented. Recent advances in development of active materials and different types of capping materials are promising for further development of feasible low-impact and cost-efficient remediation approaches.

Remediation experiences

In Norway, the first sediment clean-up project was the *in situ* capping of Eitrheimsvågen in Odda carried out in 1992 at a cost of approximately NOK 40 million. During the period 2001/2002, five pilot projects were performed in Tromsø, Trondheim, Sandefjord, Kristiansand and Horten in order to gain new knowledge, learn how clean-ups of contaminated sediments could best be organized and carried out, as well as to gain more practical experience. In 2003, dredging and disposal at a near shore confined disposal site was carried out at Haakonsvern, Bergen. Later, several full-scale projects have been conducted, such as Kristiansand harbor, Oslo harbor and Tromsø harbor. One of the main knowledge needs identified by the Norwegian Council on Contaminated Sediments in 2006 was related to long term monitoring of remediation, to assess the efficiency of the remediation action and the potential recontamination, and to gain knowledge for future remediation projects.

In Sweden, some clean-up projects have been reported and reference projects can be found at http://atgardsportalen.se/. These involve mostly dredging and disposal, but also some capping projects. In Finland, only minor remediation actions have been motivated by clean-up, whereas several projects have been motivated by infrastructure developments.

How to decide upon remediation techniques

The risk-based approach to define clean-up levels requires assessment of site-specific conditions and bioavailability of contaminants. Norwegian guidelines for risk assessment of contaminated sediments are based on a tiered approach where Tier 1 equals the EQS and Tier 3 includes site-specific values for the parameters included in the risk assessment, such as partition coefficients and organic content in sediments. However, few project owners collect enough data to assess risk and develop Tier 3 clean-up levels based on bioavailability and site-specific conditions.

Decision criteria for remediation strategy should be used to make the decision-making process transparent. The decision criteria Efficiency, Benefits, Adverse effects and Cost have been used in the development of several remediation action plans in Norway. Multiple evaluation criteria to decide upon remediation strategy demonstrates the complexity and the need to do site-specific assessments prior to a decision. A climate change exposure assessment can be performed to reflect on a range of possible climate and weather scenarios.

Recommendations

- Of the Nordic countries included in the review, Finland needs comprehensive survey of suspected sites and national strategy for dealing with contaminated sediments.
- Risk based approach is beneficial to identify contaminated sediments, prioritize between sites and decide clean-up levels.
- Research and method developments are needed for the development of risk assessment tools for sites with multiple contaminants.
- Further research (both field and laboratory) is needed to develop new remediation techniques.
- The long-term effect of low impact strategies needs to be further investigated.
- Monitored Natural Recovery should be given more consideration as an alternative remediation strategy.
- Multi-criteria analyses to decide upon an efficient remediation strategy should be considered, taking into account criteria such as adverse effects in addition to benefit, cost and efficiency.
- A Nordic database of clean-up projects and long-term monitoring would improve the exchange of knowledge between the Nordic countries.
- The guidelines for contaminated sediments vary considerably between the Nordic countries. The Nordic countries should consider the possibility to have joint guidelines for contaminated sediments.

1. Introduction

1.1 Resume

Contaminated sediment has been recognized as a significant reservoir of legacy contaminants in industrial sites worldwide, potentially contributing with releases of contaminants for decades after industrial releases have been stopped and acting as a source for transfer of contaminants into aquatic food chains. The Nordic Council of Ministers have recognized the need to compile the knowledge of sediment remediation strategies and to evaluate the different approaches in relation to cost, efficiency and adverse effects on the environment. The objective of this report is to give a status of regulatory framework and sediment management within selected Nordic countries (Norway, Sweden, Finland and Denmark). In addition, the report gives a short overview of international knowledge on remediation approaches, techniques and materials.

1.2 Background

Following the increased awareness of pollution in the 1960–70s, primary emissions from industry, farmlands and households were significantly reduced during subsequent decades. Towards the turn of the millennia an increasing awareness of environmental problems caused by contaminated soils and sediments has occurred. Areas of interest are typically in the vicinity of industrial sites, cities or harbours. Industrial sites are often characterized by a single or a few specific contaminants, whereas areas accumulating contaminants from harbours or cities are often characterised by a large number of known and unknown toxic compounds. Contaminated sediments have been recognized as an environmental challenge and a source of pollution in aquatic food chains (Malins, Krahn *et al*. 1985, Varanasi, Reichert *et al*. 1985). The problem has attracted international attention, both scientific and political. Sediment has been recognized as a significant reservoir of legacy contaminants in industrial sites, potentially contributing with releases of contaminants for decades after industrial releases stopped. The concerns regarding the occurrence and extent of ecological and human health risks of contaminated sediment continue to grow (Spadaro 2011).

Worldwide, a number of sites have been identified and assigned priority in national programs for remediation. However, governance and regulatory instruments in dealing with contaminated sediments vary between countries. To date there has not been any global state-of-the-art compilation of remediation approaches or regulatory means, although Spadaro (2011) made an attempt to present a short worldwide status survey of regulation and technology. Spadaro's (2011) conclusion was that, as of March 2010, approximately 35 countries appeared to have some type of regulatory framework relating to contaminated sediment management, primarily in the form of quality standards related to dredging. Only a few of the frameworks appeared to be more than guidance, and about the same number (a few) appeared to have some type of technical framework available to evaluate risks from sediment contamination. Also, within the Nordic countries there is great variation in regulatory frameworks for management of contaminated sediment. In Norway, the Norwegian environmental authorities implemented a national strategy for remediation of marine contaminated sediments within harbours and fjords (White Paper 14 [2006–2007] "Working together towards a non-toxic environment and a safer future – Norway's chemicals policy"), which resulted in the identification of prioritized sites, the development of action plans and the initiation of several remediation actions. The other Nordic countries do not have any corresponding national strategy.

Historically, sediment actions like dredging were initiated to maintain sailing depth and harbour facilities, whereas remediation actions to improve the ecological quality of contaminated sediment are becoming more common throughout the world. Different approaches, such as dredging, capping, and monitored natural recovery (MNR) have been proposed, tested and applied worldwide. Lately, there has been an increasing awareness of the potential secondary effects of sediment remediation, and research has been conducted to gain knowledge of both the secondary effects of remediation and to develop new low-impact remediation approaches. However, the status given by Spadaro (2011) revealed that only a small minority of countries appeared to be intentionally employing techniques other than dredging, such as capping or MNR. Despite increasing effort, there appears to be no consensus on the best way to apply a scientifically sound, risk-based approach to the screening and clean-up of contaminated sediment sites.

The Nordic Council of Ministers have recognized the need to compile the knowledge of remediation strategies and to evaluate the different approaches in relation to cost, efficiency and adverse effects on the environment within the Nordic countries, to ensure that the best possible measures are taken in the future.

The objective of this report is to give a status of regulatory framework and sediment management in Norway, Sweden, Finland and Denmark. In addition, the report gives a short overview of international knowledge on remediation approaches, techniques and materials. This overview includes knowledge on materials and methods for removing contaminated sediments, capping of sediments and enhancing natural recovery, to prevent flux and uptake of contaminants to water and biota. Compilation of knowledge will also include the factors and mechanisms that might affect bioavailability and effects on biota, including ecotoxicological issues, as well as an evaluation of the ability of the different remediation approaches, techniques and materials to reduce the risk of environmental effects of single pollutants and chemical mixtures, with reference to the Technical Guidance Document on Risk Assessment (European Commission 2003) and the proposed framework for environmental risk assessment of chemical mixtures proposed by Backhaus and Faust (2012). As sediments usually are polluted by more than one source and one contaminant, the combined effects of contaminants will be taken into consideration as well as how measures deal with combinations of contaminants. Methods, techniques and materials are presented and evaluated in relation to feasibility, cost, efficiency and adverse effects on the marine environment.

2. Classification of contaminated sediments and quality standards

2.1 Resume

Classification of sediment based on concentrations of defined contaminants or groups of contaminants is commonly used as a tool to identify sites of concern. The Water Framework Directive (WFD) is implemented with the aim to achieve “good status” for all ground and surface waters in the EU, based on Environmental Quality Standards (EQS) developed and implemented under the WFD. In Norway, chemical status is primarily based on the national set EQS for sediment (set for 28 EU priority substances) and biota (set for 23 EU priority substances), whereas in Finland the chemical status of water bodies is based on the EQS set for surface water and biota. Finland is not applying any sediment EQS values. In Denmark and Sweden, the definition of good chemical status is based on EQS for water, sediment and biota, though in Sweden EQS has been defined only for a few substances in biota and sediment.

2.2 Environmental Quality Standards

The Water Framework Directive (WFD) (2000) is implemented in EU member states and other European countries. In Norway, the directive is implemented as Vannforskriften (2006) (https://lovdata.no/dokument/SF/forskrift/2006-12-15-1446/). The aim of the WFD is to achieve “good status” for all ground and surface waters in the EU. Surface waters include rivers, lakes, brackish water and coastal waters. Coastal waters are defined as reaching 1 nautical mile off the coast. EQS for water, sediment and biota have been developed and implemented under the WFD. EQS are set for annual average concentrations (AA-EQS) and/or for maximum admissible concentrations (MAC-EQS). In 2013, a new European *Directive*, 2013/39/EC (2013), amended the *Directives 2000/60/EC* (2000) and *2008/105/EC* (2008) as regards *priority substances* in the field of *water policy*. Newly identified substances were added, including the setting of *EQS*, and *EQS* of some existing substances were revised.

Monitoring of sediment and biota is optional and it is the individual nation’s choice whether to include these matrices. In Norway, chemical status is primarily based on monitoring of pollutants in sediment and biota. In Finland, the chemical status of water bodies is based on the EQS set for surface water and biota (Ympäristöministeriö 2018). In Denmark, the definition of good chemical status is that the concentrations of pollutants do not exceed the national set EQS for water, sediment and biota

(MFVM 2017). The same concept applies in Sweden, though EQS have been defined only for a few substances in biota and sediment (Havs-och Vattenmyndigheten 2013).

Concentrations above the EQS are supposed to initiate further investigations, assessments and potentially also measures to enhance the environmental quality by reducing concentrations to levels below EQS.

There are no EQS set for sediment pore water.

2.3 National classification systems for contaminated sediments

2.3.1 Norway

The Norwegian Environment Agency has defined EQS in water for the 45 EU priority substances, in biota for 23 EU priority substances and in sediment for 28 EU priority substances. In addition, the Norwegian Environment Agency has defined EQS for water, biota (mainly fish but also a few other organisms/matrices) and sediment for water region-specific substances, as well as class limits in water and sediment in a five-class system (class I to class V) for both EU's priority environmental substances and for water region-specific substances (Miljødirektoratet 2016, Direktoratsguppen Vanndirektivet 2018). The quality standards and class limits for sediments are mainly defined for marine sediments, though some substances are also specified for freshwater sediments. No biota class limits have been developed based on toxicity, but abundance-based class limits for a few specific organisms can be found in older guidelines (Molvær *et al*. 1997). The classification system is meant to be a tool for various professional groups and administrators in management, counseling and research for assessment and determination of environmental status in different water bodies. The criteria for the determination of class limits are based on internationally established systems for environmental quality standards and risk assessment of chemicals in the EU, and the quality standards are prepared as described in the Technical Guidance Document for Deriving Environmental Quality Standards (TGD No. 27). In the classification system, class boundaries represent an expected increasing level of damage to the organisms in the water column and sediments. The limits are based on available information from laboratory tests, risk assessments and dossiers on acute and chronic toxicity on organisms.

The limit values and class limits (except Class I) have been established based on available information on the environmental toxicity of ecotoxicological laboratory tests. To ensure adequate protection where there are not enough data, assessment factors (AF) are used. Implementing AFs accounts for organisms that are more sensitive than those used in laboratory tests. The value of the AF depends on the data availability for the substance, with decreasing AFs with increasing amount of data.

The upper limit of Class II and III in the classification system complies with the WFD AA-EQS and MAC-EQS. Upper Limit for Class II is equivalent to AA-EQS, which is the limit of chronic effects on long-term exposure, and upper limit for Class III is equivalent to MAC-EQS, which is the limit of acute toxic effects in short-term exposure. The upper limit for Class I represents high background values for naturally occurring contaminants (e.g. some metals). For most of the anthropogenic contaminants without any natural source (such as persistent organic pollutants), the upper limit for Class I is set to zero. The upper limit for Class IV corresponds to the limit above which there is a risk for extensive acute toxic effects (risk assessed without safety factors), whereas the upper limit for class II represent the lowest level at which a risk for acute toxic effects is present (safety factors included). All class boundaries outside the upper limit of Class I are calculated from toxicity risk assessments.

The sediment classification system is intended for use in minerogenic fine grained sediment consisting of clay and/or silt. As environmental contaminants are mainly related to small particles and organic matter, minerogenic sediments with gravel or coarse sand will not be suitable for assessment through this system. Further, the limit values are adapted to Norwegian conditions where the organic carbon content in sediment is generally lower than in many EU countries. For 17 prioritized fjords (see chapter 4.1), regional action plans for contaminated seabed have been prepared.

2.3.2 Sweden

The Swedish EPA is responsible for setting up a classification system for pollution levels. The agency has compiled methods of inventories of contaminated sites (MIFO in Swedish, [Swedish EPA 2002]) taking into account the toxicity levels of different pollutants together with other environment factors that determine the environmental risk. EQS have been set up for soil and groundwater. However, only a few EQS are to date defined in Sweden for marine sediments (see table 1) (Havs-och Vattenmyndigheten 2013). The Norwegian threshold values are often used when a comparison with ecotoxicological effects is needed. Effects and safety threshold values from the USA (effect threshold levels set by the National Oceanic and Atmospheric Administration – NOAA), Canada (TEL, Threshold Effect Levels set by the Canadian EPA), the Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR convention), and the UK (WRC, Water Research Centre) are also used for reference (Swedish EPA 2002).

The Swedish EPA has determined, for each contaminant, five concentration classes as a function of their occurrence in all sediment samples collected between 1986 and 2014 (Josefsson 2017). The classes are not related to ecotoxicological effects, but only to the abundance in the considered samples. This approach aims to provide an overview of the levels of contaminants in the country, to highlight the biggest threats, characterise the level of contamination for future remediation, in monitoring of remediated sites, and to provide data for understanding the effect of various industries nationally and internationally (Eriksson 2017). The review of the levels of organic contaminants in marine sediments in Sweden has been revised in 2017 in collaboration with the Geological Survey of Sweden (SGU). The same kind of classification is used for metals in sediment. The currently used version is from 1999 (Naturvårdsverket 1999). It includes arsenic (As), cadmium (Cd), cobalt (Co), chromium (Cr), copper (Cu), mercury (Hg), nickel (Ni), lead (Pb), and zinc (Zn).

2.3.3 Finland

The WFD and Marine Strategy Framework Directive are implemented in Finland through national acts and decrees, and the chemical status of water bodies is assessed using EQS for surface waters and biota. The EQS values are based on priority substance directive (2013/39/EU) and national harmful substance list. However, as the WFD and the amendment directives do not contain EQS values for the sediments and national values have not been derived, Finland is not applying any sediment EQS values. Therefore, there is no classification system for contaminated sediments in Finland: the status of contamination is only assessed for relocation purposes after dredging (see 3.5.3).

Table 1: National EQS values for marine and freshwater sediments in the Nordic countries (mg/kg dry weight)

Compound	Denmark[1]	Norway[2]	Sweden[3]
Alachlor		0.0003 (CW)	
Anthracene	0.024 (IW) 0.0048 (OSW)	0.0046 (CW)	0.024[a] (U)
Arsenic (As)		18 (U)	
Bisphenol A		0.0011 (U)	
Brominated diphenylethers		0.062 (CW) 0.31 (FW)	
Lead (Pb)	163 (IW, OSW)	150 (CW) 66 (FW)	130 (IW) 120 (OSW)
Cadmium	3.8 (IW, OSW)	2.5 (CW)	2.3 (U)
C10-13 chloroalkanes		0.8 (CW)	
Chlorfenvifos		0.0005 (CW)	
Chlorpyrifos		0.0013 (CW)	
Chromium (Cr)		660 (U)	
Copper (Cu)		84 (U)	
DDT total		0.015 (U)	
Para-para-DDT		0.006 (U)	
Dekamethyl cyclo pentasiloxane (D5)		0.044 (U)	
Di-(2-ethylhexyl)phthalate (DEHP)		10 (CW)	
Diflubenzuron		0.000184 (U)	
Dioxin and dioxin-like PCBs[4]		8.6×10^{-7} TEQ (CW)	
Dodecylphenol with isomers		0.0044 (U)	
Endosulfan		0.00007 (CW)	
Ethinylestradiol	17.3×10^{-6} $3.428 \times 10^{-4} \times f_{OC}$ (IW, OSW)		
Fluoranthene		0.40 (CW)	2.0[a] (U)
Hexabromocyclododekan (HBCDD)[5]		0.034 (CW) 0.17 (FW)	
Hexachlorobenzene		0.017 (CW)	
Hexachlorobutadiene		0.049 (CW)	
Hexachlorocyclohexane		0.000074 (CW) 0.00074 (FW)	
Medium chain chlorinated paraffins		4.6 (U)	
Mercury and mercury compounds		0.52 (CW)	
Methylnaphthalenes (PAH): 1-methylnaphthalene 2-methylnaphthalene Dimethylnaphthalenes trimethylnaphthalenes	$\Sigma = 0.478 \times f_{OC}$ (IW, OSW)		
Methyl-tert-butylether (MTBE)	0.081 (IW, OSW)		
Naphthalene	0.138 (IW, OSW)	0.027 (CW)	
Nickel and nickel compounds		42 (CW)	
Nonylphenol	$25 \times f_{OC}$ (IW) $2.5 \times f_{OC}$ (OSW)	0.016 (CW)	
Octylphenol	$39.3 \times f_{OC}$ (IW) $3.93 \times f_{OC}$ (OSW)	0.0003 (CW) 0.003 (FW)	

Compound	Denmark[1]	Norway[2]	Sweden[3]
PAHs			
acenaphtylene		0.033 (U)	
Acenaphtene		0.10 (U)	
Benzo(a)anthracene		0.06 (U)	
Benzo(a)pyrene		0.18 (CW)	
Benzo(b)fluoranthene		0.14 (CW)	
Benzo(k)fluoranthene		0.14 (CW)	
Benzo(g,h,i)perylene		0.084 (CW)	
Chrysene		0.28 (U)	
Dibenzo(ah)antracene		0.027 (U)	
Fluorene		0.15 (U)	
Indeno(1,2,3-cd)pyrene		0.063 (CW)	
Phenanthrene		0.78 (U)	
Pyrene		0.084 (U)	
Pentachlorobenzene		0.4 (CW)	
Pentachlorophenol		0.014 (CW)	
Perfluoroctanic acid (PFOA)		0.071 (U)	
Perfluoroktylsulfonat and its derivatives (PFOS)		0.00023 (CW) 0.0023 (FW)	
PCB 7		0.0041 (U)	
Strontium	75 (IW)		
Silver (Ag)	1.5 (IW) 13 (OSW)		
1,2,4-triazol	5.5 x f_{OC} (IW) 0.55 x f_{OC} (OSW)		
Teflubenzuron		0.0000004 (U)	
Tetrabromobisphenol A (TBBPA)		0.108 (U)	
Tributyltin compounds (tributyltin cation)		0.000002 (CW)	0.0016 [a] (U)
Triphenyltin		$3.61E^{-05}$ (U)	
Trichlorobenzenes		0.0056 (CW)	
Triclosan		0.009 (U)	
Trifluralin		1.6 (CW)	
Tris(2-chloroethyl)phosphate, TCEP		0.0716 (U)	
Tris(2-cholr-1-methylethyl)phosphate (TCCP)	111 x f_{OC} (IW) 11.1 x f_{OC} (OSW)		
Vanadium (V)	23.6 (IW, OSW)		
Zinc (Zn)		139 (U)	

Note: EQS for sediment in Sweden (sediments with 5% of organic carbon, except for cadmium and lead compounds, for which the organic carbon content has been normalised for comparison). CW = coastal water, IW = inland water, OSW = other surface water, FW = fresh water, U = water type undefined.

[1] MFVM 2017. Bekendtgørelse om fastlæggelse af miljømål for vandløb, søer, overgangsvande, kystvande og grundvand. BEK nr 1625 af 19/12/2017. Miljø- og Fødevareministeriet, Miljøstyrelsen, j.nr. SVANA-400-00066. f_{OC} is the fraction of organic matter in the sediment.

[2] Miljødirektoratet 2016 (M608). EQS for priority substances and hazardous substances in sediments are given for coastal water and freshwater, EQS in Norway correspond to Class II in the national classification system of chemical condition. EQSs for other EU-chosen substances and water region specific substances are not discriminated between coastal and fresh water.

[3] HVMFS 2015:4. Havs- och vattenmyndighetens föreskrifter om ändring i Havs- och vattenmyndighetens föreskrifter (HVMFS 2013:19) om klassifisering och miljökvalitetsnormer avseende ytvatten. Havs- och vattenmyndighetens författningssamling.

[4] This includes the following substances: 7 polychlorinated dibenzo-p-dioxins (PCDDs): 2,3,7,8-T4CDD (CAS 1746-01-6), 1,2,3,7,8-P5CDD (CAS 40321-76-4), 1,2,3,4,7,8- H6CDD (CAS 39227-28-6), 1,2,3,6,7,8-H6CDD (CAS 57653-85-7), 1,2,3,7,8,9-H6CDD (CAS 19408-74-3), 1,2,3,4,6,7,8-H7CDD (CAS 35822-46-9), 1,2,3,4,6,7,8,9-O8CDD (CAS 3268-87-9). 10 polychlorinated dibenzofuranes (PCDFs): 2,3,7,8-T4CDF (CAS 51207-31-9), 1,2,3,7,8-P5CDF (CAS 57117-41-6), 2,3,4,7,8-P5CDF (CAS 57117-31-4), 1,2,3,4,7,8-H6CDF (CAS 70648-26-9), 1,2,3,6,7,8-H6CDF (CAS 57117-44-9), 1,2,3,7,8,9-H6CDF (CAS 72918- 21-9), 2,3,4,6,7,8-H6CDF (CAS 60851-34-5), 1,2,3,4,6,7,8-H7CDF (CAS 67562-

39-4), 1,2,3,4,7,8,9-H7CDF (CAS 55673-89-7), 1,2,3,4,6,7,8,9-O8CDF (CAS 39001-02-0). 12 dioxin-like polychlorinated biphenyls (PCB-DL): 3,3',4,4'-T4CB (PCB 77, CAS 32598-13-3), 3,3',4',5-T4CB (PCB 81, CAS 70362- 50-4), 2,3,3',4,4'-P5CB (PCB 105, CAS 32598-14-4), 2,3,4,4',5-P5CB (PCB 114, CAS 74472-37-0), 2,3',4,4',5-P5CB (PCB 118, CAS 31508-00-6), 2,3',4,4',5'-P5CB (PCB 123, CAS 65510-44-3), 3,3',4,4',5-P5CB (PCB 126, CAS 57465-28-8), 2,3,3',4,4',5-H6CB (PCB 156, CAS 38380-08-4), 2,3,3',4,4',5'-H6CB (PCB 157, CAS 69782-90-7), 2,3',4,4',5,5'-H6CB (PCB 167, CAS 52663-72-6), 3,3',4,4',5,5'-H6CB (PCB 169, CAS 32774-16-6), 2,3,3',4,4',5,5'-H7CB (PCB 189, CAS 39635-31-9).

[5] This includes 1,3,5,7,9,11-Hexabromcyclododekan (CAS 25637-99-4), 1,2,5,6,9,10-Hexabromcyclododekan (CAS 3194-55-6), α-Hexabromcyclododekan (CAS 134237-50-6), β-Hexabromcyclododekan (CAS 134237-51-7) and γ- Hexabromcyclododekan (CAS 134237-52-8).

[a] Sediment-EQS refers to 5% organic C. Normalisation is needed when sediment organic content is greater than 5%.

Source: Havs-och Vattenmyndigheten, 2013.

3. Contaminated sediments in the Nordic countries

3.1 Resume

In Norway, more than 120 sites within the fjords have been recognized with high concentrations of hazardous substances. For 17 prioritized fjords, regional sediment remediation action plans have been prepared. In Sweden, a preliminary review of contaminated sediments revealed that contaminated mineral-based and/or cellulose-bearing sediments occur in at least 19 of 21 counties. In Finland, the status of sediment contamination is only assessed for relocation purposes after dredging. A preliminary national survey of contaminated sediments in inland waters lists 28 possible or known sites across the country. In Denmark, the coastal waters are heavily affected by anthropogenic activity, both from land- and ocean-based activities like aquaculture, shipping and industry. Hazardous substances in Danish marine waters have been monitored on a nation-wide scale since 1998.

3.2 Norway

Norway has Europe's longest coastline. Including all islands, it is approximately 101,000 km. The coastal water covers an area which is about 5 times larger than the freshwater area.

Contaminated sediments in Norway are principally related to harbours and fjords with industrial activity in the form of process industries, pulp and paper industries and shipyards, as well as shipping. In addition, emissions from municipal wastewater treatment plants, urban drainage and diffuse discharges, e.g. from polluted soil, landfills and deposit sites, contribute to the complexity in contaminant sources and impact. Examples of hazardous substances that are common in sediments are TBT, PCB, PAH and heavy metals such as mercury (Hg), lead (Pb), copper (Cu) and cadmium (Cd), as well as chlorinated compounds like dioxins (PCDD) and furans (PCDF). The seabed in 120 areas in Norwegian fjords was examined for environmental pollutants in the 1990s. About 90 of these areas showed high concentrations of one or more hazardous substances.

A typical situation in Norway is a power-intensive process industry located in the inner parts of a fjord with easy access to hydroelectric power and access via waterway for shipment of raw materials and products. Some larger lakes and rivers can also be significantly affected by industrial activity, especially from the pulp and paper industry. Today, the industry's emissions are strictly regulated, while the significantly higher emissions of earlier times have led to contaminated sediment in the recipients. Natural recovery through natural oversedimentation is usually recognized where rivers bring clean particles that settle on top of older polluted seabed, but the sedimentation rates vary and depends on site-specific conditions.

Norwegian fjords usually have one or more sills that limit the circulation of bottom water, which leads to accummulation of contaminated particles behind the sills. However, transport of contaminants across a sill can occur either due to direct emissions to water masses above the sill level or by periodic water exchanges that bring bottom water and re-suspended sediment higher up in the water column and above the sill. In several large fjords, studies have shown impacts from previous industrial emissions that have spread over several tens of km^2 from the source.

Following the implementation of WFD, Norway has registered about 2,280 coastal water bodies that are classified according to ecological and chemical status. In Norway, the chemical status in coastal waters is mainly based on monitoring of pollutants in sediment and biota. For several water bodies, there are not enough data to perform a classification according to the WFD. Data sets and classification of state in all mapped waters in Norway are gathered in the database Vannmiljø and are available through www.vannportalen.no. Assessment of the risk of failing to achieve the environmental goals of the Water Framework Directive by 2021 shows that 29% of all coastal waters are at risk of failing, while 62% are likely to achieve environmental targets, and 9% are undefined. Most coastal waters are in good ecological condition, as it is essentially the chemical condition that is the cause of the risk.

Surveillance of sediment contamination in industrialized fjords and harbours in Norway was initiated in the 1980s (SFT 2000). Following this survey, an overview of 32 areas where serious contamination was recorded was presented in 1992 (SFT 1992), and in 1993–94 the National Program for Pollution Surveillance conducted surveys along the coast to get a better overview of the situation of contaminated sediments in Norway (Konieczny 1995, Konieczny 1995, Konieczny 1996). The problems with contaminated sediments were far more extensive than previously thought, and investigations up to the end of the millennia found that sediments in more than 120 larger and smaller sub-areas (locations) in the fjords have high concentrations of hazardous substances. Extensive surveys of a number of fjords have been carried out, and coastal areas and fjords have been monitored through state-wide pollution prevention programs and through surveillance programs imposed on industry and other actors in the individual fjord areas. Following the implementation of the WFD, there is greater responsibility for monitoring for the individual actors than before.

3.3 Sweden

Swedish marine waters are mainly part of the Baltic Sea, except for a small part of the coast in the Kattegat and Skagerrak in the southwest. The Baltic Sea is a unique environment, it is bordered by nine countries, with densely populated land areas in its southern part. It consists of a semi-enclosed sea with surface water salinity varying from 1.5–1.8% in the southern part to 0–0.2% in the northern parts (HELCOM 2018). Around

85 million people share its catchment area (Sobek, Sundqvist *et al*. 2015). About a third of the Baltic Sea is shallower than 30 meters and water exchange occurs only through the three narrow Danish straits; the Great Belt, the Little Belt and The Sound (Öresund/Øresund), connecting the Baltic Sea with the Kattegat and Skagerrak strait in the North Sea. Anthropogenic pressures and the lack of water exchange and vertical circulation create a very fragile and endangered environment. Extensive efforts have been done in the last few decades to lower pollution sources (nutrients and toxins) and curtail eutrophication but more actions are needed to reach a good status, and it will take a long time for the environment to recover (Kotilainen, Arppe *et al*. 2014, HELCOM 2018, Severin, Josefsson *et al*. 2018). These actions include the remediation of contaminated marine sediments (Severin, Josefsson *et al*. 2018).

In Sweden, knowledge about soil pollution and management of contaminated land-based sites is much more developed than knowledge about contaminated sediments and their management. Aware of this knowledge gap, and pressed by human health risks posed by the poor state of the Baltic Sea, the government decided in August 2017 to distribute funds for soil and sediment remediation between 2018 and 2020 (Severin, Josefsson *et al*. 2018).

Currently, known contaminated sediment sites in Sweden are historical consequences of past industrial activity. Due to recent regulations, releases of pollutants are either forbidden or more controlled, and therefore significantly lowered. However previous releases can still impact the environment because of their past accumulation in the environment, subsequent release and potential bioaccumulation and/or biomagnification. All data on recognized contaminated sites and their state of remediation are compiled on a specific portal (EBH-portal), where official reports are publicly accessible. In 2017 a special issue on contaminated sediment was published (Klas Köhler 2017). Each county board also publishes regional site-specific reports on their own websites.

Jersak *et al*. (2016) summarizes results of a preliminary review of the type and occurrence of contaminated sediments identified in inland and/or coastal waters within each of Sweden’s 21 counties. Contaminated mineral-based (minerogenic) and/or cellulose-bearing (“fiberbank”) sediments occur in at least 19 counties. At many sites, sediment contamination likely poses unacceptable risks to the environment and/or human health although less than a handful of management decisions have been taken.

Mercury was heavily present in water and sediments until the 1970s because of industrial releases (from chlor-alkali plants, metal production, producing chlorine and sodium hydroxide for the forest industry), waste incineration and diffuse sources (Elmgren 2001, Lindeström 2001). Once the toxicity of mercury (especially methyl mercury) had been recognized, such use of mercury was banned and a health maximum concentration of mercury in fish of 1 mg.kg–1 was set (Elmgren 2001). However, due to its persistence in the environment and in particular in organic matter, mercury can still be found in aquatic environments in Sweden. It has mainly been investigated biota, and the results show good status only in the Arkona basin and a few coastal areas (HELCOM 2018).

Many studies have reported high levels of dioxins and furans in the sediments from the Baltic Sea, in the aquatic environment and in the air (De Wit, Jansson *et al.* 1990, Nylund, Asplund *et al.* 1992, Rahmberg 2012, Sobek, Wiberg *et al.* 2012, Wiberg, Assefa *et al.* 2013). These studies show that the peak concentrations in sediments were reached in the mid-1960s to 1980s, and the concentrations have followed a general decrease since then (Sobek, Wiberg *et al.* 2012). The origins of this general pollution are unsure, but probably various. Atmospheric release, such as high temperature combustion (Sobek, Wiberg *et al.* 2012) from both industrial and public activities (Wiberg, Assefa *et al.* 2013) are dominant. Ramberg *et al.* (2012) pointed in particular to paper and pulp industry releases, contributing to the dioxins accumulated in sediments in coastal ecosystems of the Bothnian Bay.

The paper and pulp industry has been thought to make a significant contribution to the contamination of sediments and marine ecosystems in the Baltic Sea as well as in lakes in Sweden, Finland and Norway. Until the Swedish Environmental Protection Act (Miljöskyddslagen) was introduced in 1969, solid waste material from the factories used to be discharged directly into the sea (or lakes). Following this law, the Licensing Board of Environmental Protection decided on case-specific regulations and limits. Contracted by Regional County Administrative Boards, SGU surveyed selected potentially contaminated seafloor areas in the vicinity of known (past and present) pulp and paper factories (including sawmills). The deposited solid waste was identified to consist of relatively thick accumulations of cellulose-rich sediment, so-called "fiberbanks", which show high amounts of persistent organic pollutants (POPs) such as PCBs, HCB and DDT, and metals such as Hg, Pb, Cd, Cr, Cu, Ni, Zn and As (Apler, Nyberg *et al.* 2014, Apler 2018, Apler, Snowball *et al.* 2019). The concentration of each pollutant varies widely from one factory to another, due to the various industrial processes used for different types of product (pulp, paper, board). Concentrations in various sites have been found to range from Very high to Low according to the Swedish Environmental Criteria (Naturvårdsverket 1999, Josefsson 2017). The amounts of PCB in fiberbanks can reach a critical level corresponding to potential chronic effects on aquatic or sediment dwelling organisms (Apler 2018, Apler, Snowball *et al.* 2019). The high biological oxygen demand characteristic of the organic-rich fiberbanks causes anoxia within the sediment mass, which results in the production of methane, carbon dioxide and hydrogen sulphide. Evidence of this gas production is visible at some sites through pockmarks on the fiberbank surfaces. There are 315 areas identified as potentially contaminated because of the vicinity to saw mills, and paper and pulp industries. SGU has now surveyed 39 sites in lakes and coastal areas that are contaminated with fibres, and only 38% of them show signs of being covered by natural, minerogenic sediments (considered to be clean) since the discharge of waste ceased (Norrlin and Josefsson 2017).

Sweden's most extensive contaminated sediment remediation project is related to Oskarshamn's harbour, which had a very active industrial history since the middle of the 1980s. A copper processing plant and a battery producer (nickel and cadmium) used to reject heavily contaminated wastes into the harbour area until the Swedish Environmental Protection Act came into force in 1969 (Miljöskyddslagen). A sewage treatment plant released many different types of hazardous substances and an oil depot might have caused oil products and heavy metals leakage. The shipping activity is also very intense, with the main ferry connection to Gotland, inducing both contaminant release and resuspension of previously contaminated sediments. The seabed is contaminated with As, Pb, Cd, Cu, Hg, Ni, Zn, dioxins, PCBs and TBT. The remediation effort started in 1996, although the actual dredging operations did not start until 2016.

3.4 Finland

There have not been systematic and nationwide surveys in Finland to detect sediment contamination and assess the environmental hazard. Surely, there are locations known to be contaminated and assessed at least for monitoring campaigns but only a few of them have been remediated. In Lake Jämsänvesi sediment was capped in 1999 due to creosote contamination from a wood impregnation facility. The sediment was covered with polypropylene geotextile and capped with sand and gravel (Hyötyläinen, Karels *et al*. 2002). Another example of conducted sediment remediation is the site contaminated with dioxins, furans and PAHs in the vicinity of a past saw mill in Penttilä, Joensuu (2009–2011). A sediment volume of 35,000 m^3 was suction dredged and placed in geotextile tubes to dewater. The dewatering was aided by the use of polymers and the dewatered sediment was subsequently transported to landfill sites. The largest known contaminated case in Finland is sediment in River Kymijoki contaminated with Hg, dioxins and furans. The river is also leaching contaminants to the Gulf of Finland in the Baltic Sea. The total volume of contaminated sediment in the river is around 5 million m^3. This river site has received thorough assessments (Salo, Verta *et al*. 2008) and environmental impact assessment (Ramboll Finland 2010). The sediment was decided to be left untouched due to low risks for human health, difficult and costly remediation options, and the high risk of redistribution of contaminants during the dredging of the river sediment.

There are two major factors in Finland limiting the progress in sediment risk assessment. First, there is limited information on the possibly contaminated sites and, in general, the lack of understanding the role and threat of sediment contamination in aquatic ecosystems. A preliminary national survey of contaminated sediments in inland waters lists possible locations and known sites (Jaakkonen 2011). The typical sources are forest industry, mining, metal industry, waste water treatment plants, chemical industry and harbours. The suspected or known contaminants are heavy metals, PAHs, TBT and chlorinated organic contaminants. The survey lists 28 possible or known sites across the country. Most sites are found in the southern part of Finland. This list is not comprehensive and additional sites have been detected later and are currently investigated. The second major factor is the lack of EQS values derived for contaminants in sediments in Finland. Therefore, there is no guidance to assess environmental hazard and the only sediment related guide is for disposal of dredged sediment (Ympäristöministeriö 2015). Dredging is typically performed in waterways and in the Baltic Sea harbours for navigational and harbour construction purposes and there was a need for guidance for deciding relocation options for dredged sediments.

3.5 Denmark

The Danish coastline is approximately 7,000 km long and quite large compared to the size of the country. The Danish waters are generally shallow and are thus more easily impacted by transported water and materials. In addition, the Danish waters have a large variation in physical, chemical and biological conditions as they are positioned in the transition between the brackish Baltic Sea and the salty North Sea, and vary from small protected areas with stagnant water and low salinity, to open waters with high salinity and high water exchange.

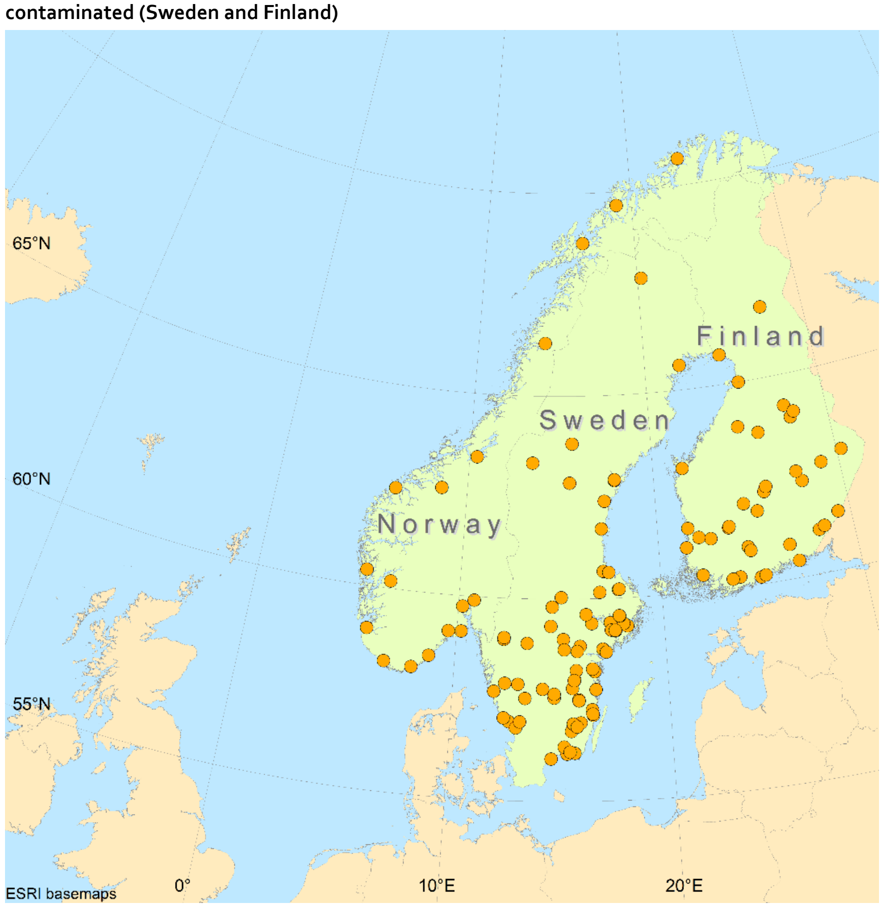
Dredging appears to be the main method for removal or handling of (contaminated) sediments in Denmark. Transport of sand due to the movement of water is common along the Danish coastline. A high portion of Danish harbours are constructed in areas affected by some degree of material movement, and the infrastructure are affecting the balance of the normal sand transport. The sand is usually captured at the harbours' upstream side and leads to erosion on the sheltered side. Eventually a balance is reached, which allows the sand to be transported past the harbour and settle in the fairway. Most harbours must over short or long intervals have sediment removed from them. Dredged material containing medium to coarse sand particles from navigable channels and fairways, can be used in exposed areas to protect against erosion as long as concentrations of organic pollutants are insignificant (Miljøstyrelsen 2015). A potential strategy to reduce erosion is to use sand capping with dredged materials from waterways on top of soft bottom sediments to facilitate establishment of eelgrass. This strategy depends on the availability of large amounts of high quality (low contamination) dredged sandy sediment (Strand, Larsen *et al*. 2018). An evaluation of materials approved for dredging in 2015 showed that most of the sediment was generally assessed to have concentrations of contaminants within background levels. However, measurements were rarely available in the database for assessment of the dredging materials (Strand, Larsen *et al*. 2018).

The Danish waters are heavily affected by anthropogenic activity, both from land- and ocean-based activities like aquaculture, shipping and industry, (Hansen 2016). The Danish environmental protection agency assesses the environmental conditions of harbours based on eleven themes (D1-11) described in Denmarks' Ocean Strategy, in which D8 states that concentrations of contaminants should be on a level that do not lead to environmental effects. The goal is to obtain good environmental condition for all 11 themes. In parallel, the WFD goals for good ecological and chemical status in fjords and coastal water should be fulfilled (Miljøstyrelsen 2018, Miljøministeriet Naturstyrelsen n.d.).

Hazardous substances in Danish marine waters have been monitored on a nation-wide scale since 1998 through The Danish National Monitoring and Assessment Programme for the Aquatic and Terrestrial Environment, NOVANA (Hansen 2016). The program is important for Denmark to fulfil its obligation in terms of national laws, EU-directives, and international conventions regarding monitoring in both the aquatic and terrestrial environment and the atmosphere (Hansen 2016). The inner Danish waters are in general classified as problem areas in terms of chemical status (Andersen, Kallenbach *et al*. 2016). A review of Danish sediment data using thresholds commonly used in OSPAR and the countries of the study area (Commission OSPAR 2009) revealed that 36% (28 sites) of the assessed areas had a high or good chemical status. Most of the assessed areas had a moderate chemical status (55%, 42 sites), and just 7 sites (9%) were given a bad or poor (NB: all calculations are provisional and the cited report should not be considered as an official Danish assessment) (Andersen, Kallenbach *et al*. 2016).

As in other Nordic countries, heavy metals are one of the main groups of contaminants in sediments. In Denmark, Hg, Pb, Cd, and Cu, typically account for 50–75% of the total toxic contribution from metals in dredged material. In addition, organic contaminants like PAHs and tributyltin (TBT) contribute substantially to the total toxicity of sediments (Stuer-Lauridsen, Geertz-Hansen *et al*. 2005). In sediments from coastal areas and inner waters, the concentration of several contaminants (e.g. Cd, Cu, Zn, Pb, Hg), reported through the NOVANA program, were higher than EUs EQS values, national set EQS or other comparable limit values (e.g. OSPARs Environmental Assessment Criteria; EAC or Effects Range Low; ERLvalues, (Commission OSPAR 2009) in some samples (Strand, Larsen *et al*. 2018).

Figure 1: Sediment sites prioritized for remediation (Norway), and sediment sites identified as contaminated (Sweden and Finland)



Note: The 17 prioritized sites in Norway is based upon the Norwegian White Paper No. 12 (2001–2002) "Protecting the Riches of the Seas". The Finnish sediment sites are based on suspected or known contamination (Jaakkonen 2011) as well as on expert knowledge and unpublished data. The Swedish sites are based on Jersak *et al* 2016.

4. Regulations of contaminated sediments in the Nordic countries

4.1 Resume

Clean-up of contaminated seabed is a priority for the Norwegian EPA and a national strategy has been implemented. The authorities have prepared a set of guidelines for contaminated sediments. In Sweden, the Swedish EPA has decided that polluted areas without any responsible parties will be taken care of by government funds. In Finland, The Ministry of Environment is the leading environmental administrative body. The only sediment related guide is for disposal of dredged sediment. In Finland, the only sediment related guide is for disposal of dredged sediments published by the leading environmental administrative body, The Ministry of Environment. In Denmark, The Ministry of Environment and Food of Denmark are responsible for the water planning, and for monitoring the condition of surface and ground water. The content of hazardous substances is included as an essential element in the evaluation of how sediments and dredging material can be handled.

4.2 Norway

The Norwegian Environment Agency is a government agency under the Ministry of Climate and Environment. The Environment Agency implements and gives advice on the development of climate and environmental policies, including the WFD and national strategies on contaminated sediments. The principal functions of the Environment Agency include collating and communicating environmental information, exercising regulatory authority, supervising and guiding regional and local government level, giving professional and technical advice, and participating in international environmental activities.

Clean-up of contaminated seabed is a priority for Norwegian authorities and has been so since the late 1980s. The work started with a White paper in 1989 followed by an action plan for cleanup of contaminated sites, prepared by the Norwegian Environment Agency (SFT 1992). The action plan stated that 32 fjord areas that where mapped and classified as highly contaminated should be further investigated and assessed for measures by 1995, and that methods and technology for measures should be tested in one or two pilot projects. As a continuation of this action plan, the Norwegian Environment Agency prepared a report during 1997 and 1998 on status and priority for action (SFT 1998). The report was based on a review of results from previous surveys, in particular surveys performed after 1992 that had revealed several areas with serious contamination. The report included a selection and prioritization for follow-up among approximately 120 areas/localities. Areas/sites where the sediments had one or more of the substances PCB, PAH, Cd, Hg, Pb and TBT in high concentrations were initially prioritized. Criteria for further prioritization were given to the type of pollution, user interests, feasibility for remediation and environmental benefits (e.g. reduction in area given dietary advice). A significant proportion of the highest priority areas were ports.

For 17 prioritized fjords, regional action plans for contaminated seabed have been prepared as a follow-up of the White Paper No. 12 (2001–2002) "Protecting the Riches of the Seas". The overall target for the sediment clean-up was defined in the White Paper as follows:

> "…to bring concentrations of environmentally hazardous substances from discharges in bygone times down to a level which will not have serious biological effects or serious effects on the ecosystem." The government also proposed the creation of "a special council to compile data on this area and provide advice on conducting investigations and implementing measures."

This White Paper resulted in the Norwegian Council on Contaminated Sediments being appointed by the Norwegian Ministry of the Environment on 1 October 2003, with a mandate until 30 June 2006. The mandate was extended several times until 2016 when the Council was closed down. In 2006, The Norwegian Council on Contaminated Sediments prepared a synthesis report reflecting the conclusions and recommendations based upon the Council's activities from 1 October 2003 to 30 June 2006.

Based on the 17 regional remediation action plans, the Government has prepared a national action plan for the clean-up of contaminated seabed, presented in the White Paper 14 (2006–2007) "Working together towards a non-toxic environment and a safer future – Norway's chemicals policy". Following this, the Norwegian Environment Agency has given orders to local industries on further investigations and development of site-specific action plans for their aquatic recipients, and in some cases also orders for implementation of measures. Clean-up measures undertaken as result of the regional action plans typically take place in close cooperation with the local, regional and/or national environmental authorities. In the years after 2000, governmental funds have been allocated annually for remediation of contaminated sediments and contaminated soil. To support the work on remediation of contaminated sediments, the authorities have prepared a set of guidelines (table 2), including Guidelines for handling sediments (Miljødirektoratet 2018) and Guidelines for risk assessment of contaminated sediments (Miljødirektoratet 2016), where the process for assessing the need for measures are described.

Table 2: A selection of Norwegian guidelines for handling contaminated sediments (not complete)

Norwegian title	English title	Report nr.
Veileder for Håndtering av sedimenter	Guidelines for handling of sediments	M-350/2015 (Miljødirektoratet 2018)
Grenseverdier for klassifisering av vann, sediment og biota	Quality standards for water, sediment and biota	M-608 /2016 (Miljødirektoratet 2016)
Veileder for risikovurdering av forurenset sediment	Guidelines for risk assessment of contaminated sediments	M-409/2016 (Miljødirektoratet 2016)
Bakgrunnsdokumenter til veiledere for risikovurdering av forurenset sediment og for klassifisering av miljøkvalitet i fjorder og kystfarvann	Background documents for risk assessment of contaminated sediments	TA 2803/2011 (Bakke, Oen *et al.* 2011)
Regneark til bruk ved risikovurdering	Excel-sheet for risk assessment	Regneark til M-409/2016 (Miljødirektoratet 2016)
Testprogram for tildekkingsmasser – Forurenset sjøbunn	Test program for capping materials – Contaminated seabed	M-411/2015 Eggen, Myhre *et al.* 2017)
Nøkkelindikator for det nasjonale arbeidet med forurenset sjøbunn	Key indicator for the national work with contaminated sea bed sediments	M-831 /2017 (Lone, Røysland *et al.* 2012)
Faktaark, Tiltaksplaner for opprydding i forurenset sediment	Fact sheet; action plans for sediment remediation	M-325 (Miljødirektoratet 2015)

Note: More guidelines and reports can be found at: http://www.miljodirektoratet.no/no/Tema/Forurenset_sjobunn/

Following the implementation of the WFD in Norway, sediment monitoring provides grounds for assessing concentrations in the sediments against established EQS for EU's priority substances and for the region-specific substances. This assessment coincides with the Step 1 for risk assessment, though the guidelines set specific criteria for the density of sediment sampling stations. If the EQS is exceeded for sediment, the next step is to carry out a more detailed risk assessment (Step 2). In Step 2, the risk of spreading, ecological risk and risk to human health is assessed based on all measured pollutant parameters and a number of standard parameters for risk calculation. With even more site-specific data, such as site-specific partition coefficients, a Step 3 risk assessment can be performed. The risk assessment is based on summarizing the risk from the individual contaminants and provides the basis for identifying the contaminants that make up the largest risk contribution. The risk assessment can also be used to compare risk in different sub-areas of the planned action area, and to prioritize efforts in the areas with the highest risk.

4.3 Sweden

The Swedish Environmental Protection Agency (Narturvårdsverket) is the governmental agency responsible for environmental issues, and more precisely for coordinating, prioritizing and following up the work on environmental issues at the national level. Founded in 1967, the agency publishes reports, called "Swedish Pollutants Release and Transfer Register", on the quantities of 70 chemical substances emitted by large industries. They work together with other governmental agencies: the Geological Survey of Sweden (SGU), the Swedish Geotechnical Institute (SGI), the County Administrative Board (Länsstyrelsen) of each region, and the Swedish Agency for Marine and Water Management (SwAM, or HaV in Swedish). SGI has the national responsibility for research, technological development and knowledge building for remediation and restoration of contaminated sites.

The Swedish Environmental Code is a legislation that was adopted in 1998 to promote sustainable development. It includes many important rules and, particularly relevant to this study, the "polluter pays" principle. In parallel, the Swedish Environmental Protection Agency has defined 16 environmental quality objectives (Naturvårdsverket 2018) and 28 step objectives with the overall aim of creating a completely sustainable environment as soon as possible. Progress towards the goals will be reviewed in 2020. One of the goals is to have a non-toxic environment, and more specifically contaminated areas must be remediated so that they do not represent any threat to human health and the environment. To achieve this, the Swedish EPA has decided on three major points. First, the polluted areas without any responsible parties will be taken care of by government funds. Then, an effort of technological innovation will be made together with SGU, SGI and the county boards in order to develop efficient and cost-effective remediation. Finally, the "polluter pays" principle will be applied to relevant areas, with the help of the County boards, the local authorities and SGI, to get more responsible parties involved and provide a short-term support.

4.4 Finland

The Ministry of Environment is the leading environmental administrative body in Finland and is setting guidelines and policy for implementation of EU directives and national legislation. The Finnish Environment Institute is a national expert body advising and guiding monitoring, survey and research programmes in multidisciplinary tasks. The environmental legislation is put into force via the practical work of the Regional Centres for Economic Development, Transport and the Environment (ELY) or even at city or municipality level. The chemical status of water bodies, as based on the WFD and amendment directives, is followed and classified by the regional ELY centres. The classification and actions thereafter is based on the EQS values in the surface water and biota only. In cases where high concentrations of chemicals in sediment are influencing biological responses (e.g. abundance of biota), the ecological status can be assessed lower even without existing sediment EQS values. The trends in sediment concentrations in monitoring programs can be surveyed but this does not necessarily lead to remedial actions.

In cases where the degree of contamination is assessed, quality standards in the dredging guide (Ympäristöministeriö 2015) or even in the Government Decree on "the assessment of soil contamination and remediation needs" are applied (Finnish Government 2007). In scientific publications or in national professional publications a reference to foreign sediment quality standards or guidelines is quite common (Wallin, Karjalainen *et al*. 2015, Vallius 2015).

The management of sediments by environmental authorities is most often related to dredging actions performed for keeping waterways open for navigation or constructions at harbours. Therefore, the use of waterways and the aquatic environment is the driving force for these actions, not the management of contaminated sediments i.e. environmental protection. Several Acts relate to management of dredging in Finland but the most important is the Water Act (587/2011) (Finnish Government 2011). The Acts allow the use of waterways but also require the protection of environment. The regional ELY centres are the supervising authorities and the Regional State Administrative Agencies (AVI) are responsible for permits of the dredging actions. The permits are required for almost all over 500 m^3 dredged volumes but also for lesser volumes if the dredging may have detrimental effects on environment. A simplified description of a permit process includes a start with a guidance of a local ELY centre or municipality followed by a submission of a permit application to the AVI. Statements are collected from interested parties and permit by AVI is finalized and has a legal force. The project is followed during the implementation and monitored after by an ELY center and/or a municipality.

The contamination level is one important factor in deciding the relocation and disposal of dredged sediments to open water sites. The dredging guide (Ympäristöministeriö 2015) gives the concentration levels for the most common contaminants and these levels are guiding the disposal options (table 3). These are applicable only for dredged sediments, not for assessing contamination level of sediment. The level 1 corresponds to natural conditions, 1A to concentrations not having an effect on disposal, 1B to concentrations allowing disposal to so called good and satisfying open water locations, 1C to concentrations allowing disposal to so called good open water locations and level 2 concentrations do not allow open water disposal but require landfill placement. The erosion and redistribution risk is low at good open water locations and a little bit elevated on the satisfying sites. The determined levels in table 3 are based on ecotoxicological data (European Copper Institute 2008) and guidance has been taken from the contaminant risk level calculations in the Netherlands (Lijzen, Baars *et al*. 2001) and Norway (Bakke, Oen *et al*. 2011). The water concentration based effective concentrations were translated to sediment pore water concentrations using partitioning coefficients (K_d) and further to sediment concentrations using organic carbon and clay fractions. The concentration levels are intentionally conservative and thus, mixture effects are not considered.

4.5 Denmark

The Ministry of Environment and Food of Denmark is responsible for the water planning in Denmark, and for monitoring the condition of surface and ground water and the protected areas. Denmark is divided into several water districts which in turn are divided into main water areas. The ministry of Environment and Food of Denmark sets rules and states specific environmental goals for the water districts groundwater and surface waters (watercourses, lakes, coastal water, transitional water), including rules for artificial and heavily modified surface water areas, deadlines for fulfilment of environmental goals and less stringent environmental goals (MFVM 2017).

In 2017, the Ministry of Environment and Food of Denmark issued a notification on determination of environmental goals for watercourses, lakes, transitional waters, coastal waters and ground water (MFVM 2017), which includes national EQS for sediment. Although the environmental goals are based on the EQS under the WFD, good chemical status are defined as having concentrations of contaminants below the national set EQS for water, sediment and biota (MFVM, 2017).

Similar to Finland, the content of hazardous substances is included as an essential element in the evaluation of how sediments and dredging material can be handled. The guidance document for dredging defines two action levels (concentrations) for the sediment which defines how the sediment should be handled (MST 2005) and are shown in table 3. If the concentrations are below the lower action level (class A) it can always be deposited at sea. Concentration between the two action levels (class B) can in principal be deposited in existing deposit areas but a closer evaluation of the sediment should be performed which includes amongst others the total amount of the relevant substances, choice of dredging area, and evaluation of other options for removal. Sediments with concentration above the upper action level (class C) might induce biological effects and should in principle be deposited on land, but can be deposited at sea under specific conditions (MST 2005). Sediments from navigable channels and harbours where the contents of hazardous substances are assessed to be of insignificant can also be placed along the coast to protect the coastline from erosion.

Interestingly, the national set EQS for substances in sediments (MFVM 2017) differ from the lower action levels in the guidance document for dredging (MST 2005), and national EQS are set for several substances which are not included in the assessment in the guidance document for dredged materials.

In the assessment performed by Strand (2018), it is pointed out that there is not a complete overlap in substances for which evaluation criteria have been developed between the guidance on dredging and the national EQS (MST 2005, Commission OSPAR 2009, MFVM 2017). For the overlapping substances, there appeared to be a good agreement between the lower action limits and OSPAR background concentrations. The national EQS (MFVM 2017) are generally higher than the lower action value and OSPAR background concentrations, with the exception of the EQS for anthracene which is lower than the OSPAR background concentration. Strand (2018) found that approximately 10% of the measured sediment concentrations in the NOVANA-program from 2007–2012 exceeded the lower action level for some of the substances that are evaluated for dredged materials. In addition, approximately 50% of the NOVANA data exceeded the national EQS values for anthracene, methyl-naphthalenes and nonylphenoles. Thus, exceedance of EQS in sediment do not necessarily affect how the sediment is classified and handled and exceedance of EQS in dredged materials might, therefore, occur without influencing classification and handling (Strand, Larsen *et al*. 2018).

4.6 Comparison of limit values for dredged materials in the Nordic countries

A comparison of limit concentrations in Denmark including action levels from MST (2005) and estimated background concentrations for navigable channels (MST 2005) was published by Strand (2018), and is shown in table 3 together with concentration levels used for deciding handling of dredged materials in Finland. Denmark and Finland group dredged materials into different classes based on concentrations of contaminants. Most of the assessed contaminants are common between the two countries, although the Finnish list appears to cover more substances than the Danish list. For the included metals, the lower and upper Danish action levels correspond very well with the class 1 and 2 Finnish levels. Norway and Sweden do not have such action levels or classification of dredged materials. In Norway, the concentrations in the sediment determines whether measures for clean-up will be initiated, and dredged material is considered as waste or trade waste and will have to be disposed of according to the Norwegian legislation on pollution. The concentration of pollutants determines whether the dredged material is classified as dangerous waste. In general, dredged material should be delivered to a waste disposal site. However, other types of disposal can be applied for based on evaluation of amongst others, the concentration of pollutants and amounts of material. Disposal of dredged materials at sea must be approved by a county administration.

Table 3: Danish action levels and Finnish classification of dredged sediments

Compound, unit	Danish action levels (MST 2005)		Background concentration	Finnish classification of dredged sediments (Ympäristöministeriö, 2015)			
	Lower action level	Upper action level	(Danish dredging database)	1	1A	1B	1C
Arsenic, mg/kg DW	20	60	4.0	< 15	15–50	50–70	
Lead, mg/kg DW	40	200	7.9	<40	40–80	80–100	100–200
Cadmium, mg/kg DW	0.4	2.5	0.09	< 0.5	0.5–2.5		
Chromium, mg/kg DW	50	270	14	<65	65–270		
Copper, mg/kg DW	20	90 (200 kg/year)	3.4	<35	35–50	50–70	70–90
Mercury, mg/kg DW	0.25	1	0.02	<0.1	0.1–0.6	0.6–0.8	0.8–1
Nickel, mg/kg DW	30	60	4.3	<45	45–50	50–60	
Zinc, mg/kg DW	130	500	17.9	<170	170–360	360–500	
Tributyltin, TBT, µg/kg DW	7	200 (1 kg/year)	3	<5	5–30	30–100	100–150
Triphenyltin, µg/kg DW				<2	2–10	10–20	20–30
PCB, sum of 7, µg/kg DW	20	200	0 b)	<2d	2–4 d	4–10 d	10–30 d
PAH, sum of 9, mg/kg DW	3	30	0 [b)]				
Anthracene, µg/kg DW				<20	20–500		
Naphthalene, µg/kg DW				<20	20–250	250–2,500	
Phenanthrene, µg/kg DW				<20	20–500	500–5,000	
Fluoranthene, µg/kg DW				<20	20–200	200–2,000	
Benzo(a)anthracene, µg/kg DW				<20	20–100	100–1,000	
Crysene, µg/kg DW				<20	20–300	300–3,000	
Pyrene, µg/kg DW				<20	20–280	280–2,800	
Benzo(k)fluoranthene, µg/kg DW				<20	20–250	250–2,500	
Benzo(a)pyrene, µg/kg DW				<20	20–450	450–4,500	
Benzo(g,h,i)perylene, µg/kg DW				<20	20–100	100–1,000	
Indeno(123–cd)pyrene, µg/kg DW				<20	20–100	100–1,000	
C10-C40 mg/kg DW				<100	100–300	300–1,500	
Dioxins and furans (PCDD and PCDF) WHO-TEQ/kg DW				<5	5–10	10–30	30–60

5. State of art: remediation methods and approaches

5.1 Resume

Historically, the most commonly used sediment remediation approach has been removal of the contaminated sediment by dredging and excavation and disposal/treatment off site. Isolation of the contaminated sediment by capping with and without an active barrier have recently been implemented in some countries. Other approaches including use of thin layer amendment with active materials, *in situ* stabilization, and monitored natural recovery have been developed but to a lesser degree implemented. The recent progress in development of active materials and different types of capping materials are promising for further development of feasible low-impact and cost-efficient remediation approaches, but the efficiency and potential adverse effects should be further investigated.

Several different remediation approaches have been developed, tested, and applied worldwide. Remediation actions may depend both on the type of contaminants present and local bathymetric and distributional conditions, and control on active/primary sources is a prerequisite. Traditionally, the most commonly used approach has been removal of the contaminated sediment by dredging and excavation and disposal/treatment off site. Isolation of the contaminated sediment by capping with and without an active barrier have also been implemented in some countries. Other approaches including use of thin layer amendment with active materials, in situ stabilization, and monitored natural recovery (NMR) have been developed but to a lesser degree implemented. The recent progress in development of active materials and different types of capping materials are promising for further development of feasible low-impact and cost-efficient remediation approaches.

In the following, different methods and approaches are described with special emphasis on capping and capping materials.

5.2 Removing contaminated sediments

5.2.1 Environmental dredging

Dredging removes contaminated sediment from a water body without draining or diverting the water, and a certain amount of water is removed with the sediment. The sediments are usually dewatered on land and the water is usually treated before it is discharge back to the water body. The contaminated sediments are normally disposed of in a landfill or a confined disposal facility (upland, nearshore or subaquatic confined disposal facility). Highly contaminated sediment may be treated before disposal (United States EPA 2017), though this has only been done on a pilot scale within the Nordic countries as the costs are high. To protect against resuspension during dredging, the contaminated sediment area can be closed off or even fully enclosed with silt curtains that extend to the bottom. Flotation devices at the surface and anchors at the bottom ensure that silt curtains fully enclose the area and are in contact with the sediment surface. In harbours it is sometimes not possible to deploy silt curtains because they block ship traffic. In such cases it is possible to use bubble curtains made of air bubbles that creates a barrier that limits the sediments at the dredging site from dispersing into more sensitive areas. Real time monitoring for increased turbidity is used as an early warning sign that sediment may be spreading during the dredging. Environmental dredging can be accomplished using either mechanical (grab/backhoe), hydraulic devices (suction) or mixed mechanical and hydraulic dredgers. Mechanically dredged sediments contain about 10–20% more water compared to the in-situ sediment (Det Norske Veritas 2008, United States EPA 2017). Pure hydraulic dredging takes place by sucking the sediments directly from the seabed with a pump. Simplified, one can say that the method works in the same way as a household vacuum cleaner. Hydraulically dredged sediments are typically a thin slurry that contains 5 to 10 percent solids, and produces a much larger volume of water requiring treatment than mechanical dredging (Det Norske Veritas 2008, United States EPA 2017). On the other hand, mechanical dredging implies higher spill of particles to the water column (especially if open grabs/backhoes are used) and silt/bubble curtains do not always function as intended. A large amount of dredging equipment on the market is based on a combination of mechanical and hydraulic principles (Det Norske Veritas 2008). Mixed mechanical and hydraulic dredgers use a mechanical device (e.g. a cutterhead) to loosen the sediment which is then pumped hydraulically on board the dredging vessel or through a pipeline to land. The mixed mechanical and hydraulic dredged sediment contains typically 10 to 30% solids which is more than from pure mechanical dredging but less than from pure hydraulic dredging.

5.3 In-situ amendments

5.3.1 *Capping of contaminated sediments*

Capping of contaminated sediments has been in use since the 1970s–80s and is globally one of the most widely used measures for mitigating contaminated sediments, together with dredging. Capping involves placement of one or several layers of clean (inert or active) material over the contaminated sediments with the purpose to reduce the dispersal of contaminants to a level that is environmentally acceptable, and to reduce exposure of benthic fauna to an acceptable level (Laugesen, Eek *et al*. 2016). Capping of sediments is usually performed with an expectation of being a one-time measure with a long effective life. Different types of capping strategies are possible and are suitable for different types of contaminated sediments (table 4). Isolation capping is the capping strategy most commonly used. Thin-layer capping has been tried in several places whereas capping including an active layer has been used in some projects in the USA but only on a pilot scale within the Nordic countries. The following information is to a large degree based upon a state-of-the-art overview of documented Norwegian and international experience from capping projects (Laugesen, Eek *et al*. 2016).

Table 4: Suitability of selected capping strategies under different sediment conditions

Conditions	Isolation capping with mineral materials	Thin-layer capping with active material	Isolation capping including an active layer
Areas prone to erosion	Well suited with protection against erosion	Unsuitable	Well suited with protection against erosion
Unstable seabed conditions	Requires geotechnical evaluation/projecting	Often better suited than isolation capping (lower load)	Requires geotechnical evaluation/design
Free phase pollution (oil, etc.)	Requires conservative design (large thickness, tight solutions)	Unsuitable	Requires conservative design (large thickness, tight solutions). Active layer can bind oil (the capacity of the active layer is important)
Ground water outflow	Unsuitable (large outflow will not be stopped by capping)	Less suitable (due to low amount of active material and limited capacity)	Suitable if an active layer can reduce the flux of contaminants
Active sources on land	Unsuitable	More suitable; the active layer may have the capacity to bind some of the pollution from land	More suitable, if the active layer binds some of the pollution from land
Biological important habitat (rapid recolonization wanted)	Experiences suggest rapid recolonization, though the capping material may change the conditions and subsequently the ecosystem structure	Experiences suggests negative effects of active carbon on bottom fauna. Long-term effects are not known	Limited experience. Will depend on which active material is in the biological active layer

Note: The table is adapted, translated and slightly modified from Laugesen, Eek *et al.* 2016.

Covering the sediments (geotextiles etc.)

Geotextiles can be used to increase the bearing capacity of the sediment prior to capping, or to protect underlying cap material from erosion. They are normally not placed in areas with heavy ship traffic as they can be destroyed by anchors. Geotextiles can also be used to separate different materials with different grain size in the capping process to avoid mixing of the different layers, or to avoid mixing of the contaminated sediments with the clean capping materials. Geotextiles are flexible, porous fabrics that are made of biodegredation-resistent synthetic fibres. The most common functions of geotextiles are drainage, separation, reinforcement, and filtration (Bouthot, Wilson *et al*. 2004). Geotextiles have for example been used *in-situ* to avoid mixing of the contaminated sediments with the clean capping materials in Eitrheimsvågen in Norway. Textiles have also been used for capping of beaches (e.g. Vækerøstranda, Norway) and as foundation for filling in masses in the sea (e.g. Trondheim harbour, Norway) and to protect landfills on the seafloor (e.g. Kollevågen, Norway). Geotextile bags have been used *ex-situ* for temporary storage and dewatering (e.g. in Finland as reported above).

Reactive materials can be incapsulated between two sheets of geotextile mats (Meric *et al*, 2011, 2012, Perelo 2010). Such reactive core mats (RCMs) usually include a reactive layer containing one or more neutralizing or otherwise reactive materials (e.g. organoclay or activated carbon) to sequester organic contaminants (Olsta and Darlington 2010, Meric, Rad *et al*. 2011, Meric, Barbuto *et al*. 2012). The RCM is placed on the sediment and covered with overlying material for stability and protection. In addition to the traditional capping effects, the RCM has the potential to adsorb and/or neutralize target dissolved contaminants from the underlying sediment through diffusion and advection through the permeable RCM layer. The RCM can also prevent resuspension of contaminated sediments, and serve as a stable, protective foundation for new, overlying sediment that can be used to promote recolonization of biota (Meric, Rad *et al*. 2011, Meric, Barbuto *et al*. 2012).

Isolation capping

Isolation capping is typically capping of (contaminated) sediment with layers of different clean materials such as clay, sand, gravel and fine stone. For isolation capping to work properly, the layer(s) of protection must be deeper than the benthic zone impacted by successive bioturbating animals and plants, and the zone that might be influenced by erosion from waves, currents, or boat propellers. Typically, an isolation cap is 20–50 cm thick.

Thin-layer capping

Thin-layer capping can be seen as an Enhanced Monitored Natural Recovery (see chapter 5.3). Thin-layer capping is a relatively new technique. In Norway, this approach is until now only tested in pilot scale. Thin-layer capping may involve the use of chemically reactive materials that sequestrate and/or degrade sediment contaminants to reduce their mobility, toxicity, and bioavailability. Active materials are further discussed in 4.2.2.

Thin-layer capping and amendment with active materials are regarded as non-destructive and low-impact measures.

In the Norwegian Opticap project (Eek 2014, Schaanning, Beylich *et al*. 2014, Cornelissen, Schaanning *et al*. 2016, Samuelsson, Raymond *et al*. 2017) the effect of capping with different thicknesses and materials on the leaching and bioaccumulation from dioxin contaminated sediment were tested in large scale field and laboratory experiments. The effect of capping increased with layer thickness and in particular with addition of active materials (Josefsson, Schaanning *et al*. 2012, Schaanning, Beylich *et al*. 2014, Cornelissen, Schaanning *et al*. 2016). The Opticap project has also revealed some long-time persistent negative effects of active carbon on bottom fauna (Samuelsson *et al*. 2017), which is potentially important when considering remediation of large areas with viable benthic communities. Modelling of the transport of contaminants has been suggested to be a good tool for assessing the suitable thickness of the capping (Josefsson, Schaanning *et al*. 2012, Laugesen, Eek *et al*. 2016).

The Norwegian state of the art overview of capping experience (Laugesen, Eek *et al*. 2016) identified that the most important factors for succeeding with capping of contaminated sediments are:

- Control with active sources on land;
- Sequence: Coordination of capping projects with activities in adjacent contaminated areas in sea and on land (capping should be the last measure in the remediation sequence in order to avoid recontamination);
- Erosion: Mapping of currents, shipping traffic, waves etc.;
- Stability: Geotechnical evaluation of the planned capping should be performed;
- Thin layer caps should include an active material to significantly reduce flux of contaminants;
- Capping of large areas or areas with high ecological value should be assessed for negative effects on biota;
- Control of capping materials: The capping materials must be clean and chemically stable for the predicted effective working life of the cap.

5.3.2 Active materials (sorbents)

Due to some of the challenges that can arise with the traditional capping approaches, including resuspension of contaminated sediment in the water column, contaminated sediment residuals, contaminant transport through the cap and destruction of existing benthic ecosystems, new techniques offering greater flexibility in contaminated sediment management have been developed. One of these "new" techniques is the introduction of sorbents into contaminated sediments. These sorbents alter the sediment geochemistry, increase contaminant binding, and reduce contaminant exposure risks to humans and the environment (Ghosh, Luthy *et al*. 2011).
Sorbents are also used in active caps that are designed to use active materials (activated carbon, apatite, zeolite, organoclay, etc.) to strengthen their adsorption and

degradation capacity. The active capping technology promises to be a permanent and cost-efficient solution to contaminated sediments (Zhang, Zhu *et al*. 2016). Thin-layer capping with active carbon has shown promising results for reducing the bioavailability and leaching of organic contaminants (Cornelissen *et al*. 2017).

Several mineral-based materials have been studied for their ability to remediate metal-contaminated soil and sediment. These include zero-valent iron, hematite, ferrihydrite, apatite and clays (Qian, Chen *et al*. 2009, Su, Fang *et al*. 2016, Wang, Zhu *et al*. 2017). The natural calcium-rich clay minerals sepiolite and attapulgite can effectively reduce both the mobile metal fraction and the bioavailability to benthic organisms in Pb and Cd polluted sediments (Yin and Zhu 2016). A review of active capping technology was performed in 2016 (Zhang, Zhu *et al*. 2016) and a short overview of these results and new additional research is given below.

Activated carbon

Activated carbon (AC) is one of the most used active materials in *in situ* sorbent amendments. AC is produced from coal or biomass feedstock and treated at high temperatures to produce a highly porous structure with great sorption capacity. In general, there are two broad categories for AC application: 1) direct application of a thin layer onto the surface sediment with or without initial mixing, and 2) incorporating amendments into a pre-mixed, blended cover material of clean sand or sediment which is also applied onto the sediment surface (Patmont, Ghosh *et al*. 2015).

Four different test sites were established in the Norwegian Opticap project to test the efficiency of thin layer capping (1–5 cm) with different materials. Capping with AC mixed with clay reduced the leakage and bioavailability by more than 80% despite continuous sedimentation of contaminated material. Thin-layer capping with passive material (inert granular materials) had low or no positive effect regarding leaking and bioavailability (Eek 2014).

In situ treatment with AC is generally less disruptive and less expensive than traditional sediment clean-up technologies such as dredging or isolation capping. Tests with a range of field sediments showed that AC amendment in the range of 1–5% reduces equilibrium porewater concentration of PCBs, PAHs, DDT, dioxins and furans in the range of 70–99%, thus reducing the driving force for the diffusive flux of hydrophobic organic compounds (HOCs) into the water column and transfer into organisms (Ghosh, Luthy *et al*. 2011). Freely dissolved PCBs in porewater and overlying water measured by passive sampling were reduced by more than 95% upon amendment with 4.5% fine granular AC (Fadaei, Watson *et al*. 2015). The pore water concentration of Hg and dioxins can be reduced by 60–90% by application of AC (Eek 2015).

Most of the studies using benthic organisms show a reduction of biouptake of HOCs in the range of 70–90% compared to untreated control sediment. Reduced bioaccumulation of Cd and Hg/MeHg after amendment of AC and thiol-functionalised silica have also been demonstrated (reviewed by (Ghosh, Luthy *et al*. 2011). AC reduced the PCB uptake in fish by 87% after 90 days of exposure (Fadaei, Watson *et al*. 2015). A study looking at bioaccumulation and secondary effects in the non-biting midge *Chironomus riparius* found lower PCB concentrations in adult midges from AC amended compared to unamended sediments. AC may reduce transport from aquatic to terrestrial ecosystems by reducing bioavailability and bioaccumulation in vector species like the midge (Nybom, Abel *et al*. 2016). Reduced biouptake of PCBs was observed in *Lumbriculus variegatus* after remediation with AC mixed into sediment and with AC in a thin-layer cap (Abel *et al*. 2017).

Laboratory studies demonstrate that the effectiveness of sorbent amendment on lowering contaminant bioavailability increases with decreasing AC particle size, increasing dose of AC, greater mixing, and contact time (Ghosh, Luthy *et al*. 2011).

Several pilot fields and full-scale sites amended with AC were reviewed by Patmont *et al*. (2015), who concluded that AC can reduce pore water concentrations and biouptake significantly, often becoming more effective over time due to progressive mass transfer. When applied correctly, *in situ* treatment using sorbent material such as AC to sequester and immobilize contaminants is a reliable technology. Rapid benthic recolonization was observed at several sites, and in certain sites there were no changes in community structure and number of individuals relative to background. Reduced growth of aquatic plants in laboratory studies were observed at AC concentrations above 5% (dry wt), potentially due to nutrient dilution. At sites that were monitored over time, AC was found to remain in the sediments throughout the 3-year post-placement monitoring period. However, other projects have indicated that AC amendment slightly increases the erosion potential of sediments but still within historical data for natural sediments.

In a report reviewing potential measures for contaminated sediments in Gunneklevfjorden in Norway (Eek 2015), treatment with AC was considered to significantly reduce pore water concentrations of Hg (by 60%) and dioxins (by 90%). Thin-layer capping and amendment with active materials are regarded as non-destructive and low-impact measures. Non-woven fabric mats (NWFM) were placed on top of a layer of activated carbon (laboratory column incubation experiments) to investigate the efficiency of blocking nutrients (nitrogen, phosphorous) (Gu, Lee *et al*. 2017). The capping efficiencies for NH_4-N, T-N and PO_4-P with NWFM/AC capping was 66.0, 54.2 and 73.1% respectively. For T-P, capping efficiencies under all capping conditions were almost 100%. Overall, placing the NWFM above the AC can be successfully used for remediation of lake sediments with high amounts of nitrogen and phosphorous (Gu, Lee *et al*. 2017).

There is still ongoing research and developments related to AC amendments. Recently, *in situ* treatment employing the simultaneous application of anaerobic and aerobic microorganisms on AC was suggested to be an effective, environmentally sustainable strategy to reduce PCB levels in contaminated sediment. A 78% reduction of total PCB was observed using a 5*10^5 *Dehalobium chlorocoercia* and *Paraburkholderia xenovorans* cells/gram sediment with 1.5% AC as delivery system, and porewater concentrations of all PCBs were reduced by 94–97% for bioaugmented treatments (Payne, Ghosh *et al*. 2017).

Impacts on benthic fauna resulting from AC exposures were observed in 20% of 82 reviewed tests (primarily laboratory studies) (Janssen and Beckingham 2013, Patmont, Ghosh *et al*. 2015). For instance, increased reproduction, survival, larval growth and gut wall microvilli length in *Chironomus riparius* was found at low AC dose (0.5% sediment dw), whereas at higher doses of AC, adverse effects on emergence and larval development was observed (Nybom, Abel *et al*. 2016). Reduced growth and net loss of organism biomass of *Lumbriculus variegatus* were observed with a sediment concentration of AC of 0.1 and 1% respectively, whereas loss of biomass and mortality was observed when using AC in a thin layer cap (Abel *et al*. 2017). Effects on communities have been observed more rarely in AC field pilot demonstrations compared to laboratory tests and often diminish within 1 or 2 years following placement (Cornelissen, Elmquist Kruså *et al*. 2011, Kupryianchyk, Peeters *et al*. 2012), particularly where new (typically cleaner) sediment continues to deposit over time. Monitoring of a test site for thin layer capping with AC 49 months after remediation showed increased number of species in the test site, but the number of species were still lower than at the reference site and at the test sites with passive material, according to a review by Laugesen, Eek *et al*. (2016). This result indicates that a recolonization occurs at the site remediated by thin layer capping with AC, but that it takes a relatively long time. A similar negative effect of AC was also found in a test site in Trondheim harbour (Cornelissen, Elmquist Kruså *et al*. 2011).

AC can thus have both positive and negative effects, with the positive effects being reduced bioavailability of contaminants and the negative effects being reduced quality of bottom fauna. Higher dose and smaller grain size gave a larger reduction in bioavailability of contaminants, but also larger negative impact on the bottom fauna (Laugesen, Eek *et al*. 2016). Studies suggest that potential negative ecological effects can be minimized by maintaining finer-grained AC doses below approximately 5% dry wt basis (Patmont *et al*. 2015). A study by Abel and Akkanen (2018) using thin layer capping with AC in a test site and laboratory settings indicate that field sites dominated by shallow dwelling organisms could theoretically recover quicker after remediation (Abel and Akkanen, 2018).

One of the challenges related to AC (and other sorbents) amendment is that although the concentrations of bioavailable contaminants and their transport to surface- and ground-water are reduced, the total sediment concentrations of contaminants are not altered. Thus, the EQS for sediments will not be met by this remediation approach, even though bioavailability, for example, might be reduced. Regulatory confidence and comfort are building for consideration of bioavailability in assessments and remedial decision. However, there is still a bias against remedies other than removal. Full-scale experimental remediation at a few sites with long-term monitoring to evaluate effectiveness not only near the base of the food chain, but also looking into recovery of fish and higher animals might be necessary to gain acceptance for such amendments (Ghosh, Luthy *et al*. 2011).

The AC technology is especially attractive at locations where dredging is not feasible or appropriate such as under piers and around pilings, in sediment full of debris, in areas where over-dredging is not possible, and in ecologically sensitive sites such as wetlands. *In situ* amendments can also be used in combination with other remediation techniques (Ghosh, Luthy *et al*. 2011).

Although a few studies have reported negative effects of powdered AC amendment on specific benthic organisms, AC amendment could be the most appropriate measure in many places, such as in areas that are valuable habitats and biological resources, and where natural recovery is already in progress.

There is variation in application equipment for AC, and several trademarked, licensed and registered AC products are available, as well as other sorbents of which a few are presented below.

Biochar

Biochar is a carbon-neutral or carbon-negative material produced through thermal decomposition of plant- and animal-based biomass under oxygen-limited conditions (International Biochar Initiative, 2012). Biochar has been successfully used to reduce metal bioavailability in soil and resulting metal concentration in rice (Wang *et al*. 2018) and it is suggested that biochar could also be a suitable option for *in situ* capping for remediation of metal contaminated sediments (Wang, Zhu *et al*. 2017). Biochars derived from different biomass sources have different properties (Wang, Zhu *et al*. 2017), potentially making it possible to choose the most efficient biochar for the specific sediment contamination to be remediate. So far (to the best of our knowledge) no studies applying biochar as an active cap to remediate contaminated sediments have been performed. A conceptual model for use of biochar and in combination with other materials to reduce risk of metal pollutants for environmental systems was presented by Wang *et al*. (2017).

A new interesting development is the use of non-toxic elemental S-modified rice husk biochar as a green method for the remediation of Hg contaminated soil. Sulphur-modification of sorbents can greatly enhance Hg sorption capacity. The rice husk biochar is used as an alternative to granulated active carbon (O'Connor, Peng *et al*. 2018). More research is needed before biochar can be used in sediment remediation projects.

Apatite

Apatite is a group of phosphate minerals with high concentrations of OH-, F- and Cl- ions depending on the type of apatite. Apatite is the most abundant mineral of all phosphate bearing minerals, is readily available and has a low extraction cost. Apatite possesses a high cation exchange capacity suggesting it could be a binding agent to immobilize heavy metals (Singh, Ma *et al*. 2001, Zhang, Zhu *et al*. 2016). Hydroxyapatite has been shown to immobilize heavy metals such as Pb, Zn, Cd, Cu, Ni, and uranium (U) in water, soil and sediment (reviewed by Zhang *et al*. 2016). The breakthrough of As, Cd, Co, Se, Pb, and Zn when using apatite as capping material in column studies was significantly delayed by apatite (Dixon and Knox 2012). However, due to its nature, apatite can contain high concentrations of impurities (As, Cr, U) that may cause additional contamination when used for environmental remediation. Biologically formed apatite, like fish bone, has lower concentrations of impurities and higher solubility than mined and refined apatite (Zhang, Zhu *et al*. 2016).

Zeolite

Zeolites are crystalline, microporous, hydrated aluminosilicate minerals of alkali and alkaline earth elements. Zeolites have been used for filtration of drinking water, gas purification systems, purification of effluents, etc. Numerous studies have investigated the capacity of natural zeolites to stabilize or remove heavy metals like Pb, Fe, Cd, Zn, Co, Cu, and Mn in soil and water (reviewed by Zhang *et al*. 2016). Besides the effective removal of metals, zeolites have shown great potential to adsorb N released in eutrophic water bodies. By pre-treating the zeolite surface with cationic surfactants, they are also capable of demobilizing non-polar organics and anionic contaminants (Jacobs and Förstner 1999, Zhang, Zhu *et al*. 2016). By using modified zeolite (Z2G1) as a thin-layer capping material one could completely block the release of phosphorous from the sediment under aerobic and anoxic conditions, making it the only sediment capping material to actively remove both P and N (Gibbs and Özkundakci 2011, Zhang, Zhu *et al*. 2016).

Zero-valent iron

Zero-valent iron (ZVI, Fe^0) has been extensively studied for its reducing capacity of organic and inorganic pollutants by reducing the contaminants to non-toxic or easily degradable low-toxic forms coupled with the oxidation of Fe^0 to Fe(II) and Fe(III). ZVI has shown effective reduction of inorganic metals like Cr and As, and organic contaminants like nitro-aromatic compounds and chlorinated organic compounds. Nanoscale zero-valent iron (nZVI) particles have been designed. Compared to ZVI, nZVI particles have a larger specific surface area and higher surface reactivity and are thus more reactive to reduce contaminants. Although promising, there are concerns related to the long-term fate and ecotoxicity of nanoparticles in the environment (reviewed by Zhang *et al*. 2016).

Organoclay

Organoclays are organically modified clays that possess unique adsorption behaviour towards aromatic organic compounds, phenols, pesticides, and herbicides etc. (Park, Ayoko *et al*. 2011). Organoclays have been shown to remove benzene, dichlorobenzene, perchloroethene, As, Cr2O72-, Cr3+, Pb, Cd, Zn, and mechanically emulsified oil and grease from water (reviewed by Zhang *et al*. 2016). Since organoclay can effectively adsorb both nonpolar organic pollutants and heavy metals, it can be an appropriate option for mixed-contaminated sediment systems (Reviewed by Zhang *et al*. 2016).

Carbon nanotubes (CNTs)

Multi-walled carbon nanotube (MWCNT)-textured sand particles have been synthesized and been shown to effectively remove a wide variety of chlorinated aliphatic contaminants (Ma, Anand *et al*. 2010).

Other active materials

Other active materials mentioned in the review by Zhang *et al*. (2016) are calcite and biopolymers (cellulose, alginates, carrageenan, lignins, proteins, chitin, and chitin derivatives).

Mixtures of active materials

It is evident that the different active materials are efficient for immobilizing different kinds of contaminants. Thus, using mixtures of materials as a capping layer could enhance the sorption of different kinds of contaminants and efficiently address the co-occurrence of pollutants with different properties. A pilot-scale study (Knox, Paller *et al*. 2012) used multiple amendment active caps to treat metal-contaminated sediment in Steel Creek near Aiken, South Carolina. The caps consisted of apatite, organoclay, and biopolymers that had been initially investigated in the laboratory, and these materials were very effective in removal and retention of metals (Knox *et al*.2008).

Future research needs

Although remediation using sorbents shows great promise, some future research needs were identified by Gosh *et al*. (2011) and are listed below:

- Development of sorbents that can actively bind contaminants other than HOCs;
- Improved fundamental understanding of mechanisms of HOC binding to AC;
- Development of efficient, low impact delivery methods for amendments into sediments;
- Pilot-scale studies at various hydrodynamic and ecological environments to understand where the technology is best suited;
- Assessment of ecosystem recovery;
- Potential for microbial processes to degrade adsorbed contaminants;
- Full scale demonstrations;
- Development of modelling tools to interpret field results, understand food web transfer, predict long-term performance, and optimize AC dose and engineering methods of application;
- Life cycle analysis including carbon footprints of different sediment remediation technologies.

5.3.3 Stabilization

Stabilization with cement

Addition of cement or other solidifying materials to sediment will increase the physical strength of the sediment, reduce the permeability and reduce the leakage of some contaminants (Eek *et al*. 2015). Stabilization with cement is normally done for nearshore deposit sites (CDFs) to gain new land and the stabilized sediment is then isolated from the water and sediment in the recipient. Sediment stabilization of the sediments *in situ* may lead to biota on the seafloor being killed by the reactive and active components of the cement (Eek *et al*. 2015). With regard to Hg, tests have shown that this method is unable to reduce the bioavailability of mercury in several types of sediment. Stabilization of sediments is a very costly method, but in certain cases where you can gain new land it can be profitable to stabilize the sediments.

Redox stabilization to prevent methylmercury formation

Redox stabilization by adding a redox buffer (FeOOH, MnO2 or NO3-) can be used to prevent formation of methylmercury in mercury contaminated sediments (Eek *et al*. 2015). This method has not been sufficiently assessed for efficiency and potential negative effects and have therefore not yet been used in sediment remediation projects (Eek *et al*. 2015).

5.3.4 Bioremediation

Microbial degradation

Anaerobic bacteria can dechlorinate PCBs, whereas lightly chlorinated PCBs can be substrates for aerobic bacteria (Gomes, Dias-Ferreira *et al*. 2013). Swedish studies indicate bacterial decomposition of naphthalene, though no quantification has been made yet (Ian Snowball, pers.comm.).

Amendment combined with bioremediation

Some studies have investigated approaches combining amendment with bioremediation. Micoorganisms that have the ability to biotransform contaminants as well as have the ability to biotransform contaminants, can potentially increase the effectivenes of the remediation (Zhang, Zhu *et al*. 2016). Capping systems containing different sorbents and microorganisms have been developed in several studies (Sun, Xu *et al*. 2010, Huang, Zhou *et al*. 2013, Wang, Li *et al*. 2014).

5.3.5 Phytoremediation

The theory of phytoremediation is based on the use of plants to extract, sequester and/or detoxify pollutants from contaminated sediment. It is more commonly applied to contaminated soils, but it is under investigation for dredged sediments disposed in landfills for treatments and sediments lying in shallow waters (Perelo 2010).

Phytoremediation is still an emerging technology that has to prove its sustainability at field scale.

5.4 Monitored Natural Recovery

Contaminated sediments may naturally become less of a risk over time. During natural attenuation, pollutants are transformed to less harmful forms or immobilized by a wide range of processes that include biodegradation, dispersion, dilution, sorption, volatilization, radioactive decay, and chemical or biological stabilization, transformation, or destruction of contaminants (Gomes, Dias-Ferreira *et al*. 2013). Natural sedimentation of clean particles may also lead to reduced surface sediment concentrations of contaminants. Monitored Natural Recovery (MNR) is a remediation practice that relies on these natural processes to protect the environment and receptors from unacceptable exposures to contaminants. This remedial approach depends on natural processes to decrease chemical contaminants in sediment to acceptable levels within a reasonable timeframe. Enhanced MNR (EMNR) applies material or amendments to enhance these natural recovery processes, such as the addition of a thin-layer cap or a carbon amendment; see section 4.2.1 and 4.2.2. Parallel natural or enhanced processes, taken together with observed and predicted reductions of contaminant concentrations in fish tissue, sediments, and water, provide multiple lines of evidence to support the selection of MNR/EMNR. Extensive monitoring is therefore crucial to document the development in the environmental condition, spreading and risk in surface sediments (Eek 2015). The success of MNR/EMNR also depends on adequate control of contributing sources of contamination so that the recovery processes can be effective.

6. Experience gained from selected Nordic clean-up projects

6.1 Resume

In Norway, the first sediment clean-up project was the *in situ* capping of Eitrheimsvågen in Odda carried out in 1992. During the period 2001/2002, five pilot projects were performed in Tromsø, Trondheim, Sandefjord, Kristiansand and Horten. Several full-scale projects have been subsequently conducted. One of the main knowledge needs identified was related to long term monitoring of remediation, to assess the efficiency of the remediation action and the potential recontamination. In Sweden, some clean-up projects have been reported. Reference projects involve mostly dredging and disposal, but also some capping projects. In Finland, only minor remediation actions have been motivated by clean-up, whereas several projects have been motivated by infrastructure developments.

In Sweden, some clean-up projects gave been reported and reference projects can be found at http://atgardsportalen.se/. There are mostly dredging and disposal but also some capping projects. In the ongoing clean-up in Oskarshamn (see chapter 3.2), sediments are dredged, dewatered on shore, and stored in a landfill, while the polluted water is treated. Hydraulic dredging was first used, but proved difficult due to the presence of large rocks which hindered progress. More than 15,000 tons of rocks had to be moved by divers. Mechanical- and suction-dedging were then combined to remove most of the polluted sediment.

In Finland, only minor remediation actions have been motivated by clean-up, whereas several projects have been motivated by infrastructure developments, such as in the harbours of Helsinki (Vuosaari) and Turku. In Turku, the River Aurajoki transports sedimenting solids into the harbor area amounting to 100 000 cubic metres each year. Maintenance dredging involves an almost equal amount every year. A major part of these dredged sediments are considered contaminated and need to be treated in an environmentally safe way before deposition on the shore. The TBT contaminated dredged sediment was mixed ex-situ with binding materials (cement, blast furnace slag and coal fly ash) for stabilization and deposited on the pool in the harbor of Pansio.

In Norway, one of the main knowledge gaps identified by the Norwegian Council on Contaminated Sediments (2006) was related to long term monitoring. Monitoring of remediation effects are vital, to assess the efficiency of the remediation action and the potential recontamination, and to gain knowledge for future remediation projects. Until 2006, only two full-scale sediment remediation projects had been carried out in Norway (Nasjonalt råd for forurensede sedimenter 2006). These were the *in situ* capping of Eitrheimsvågen in Odda carried out in 1992 at a cost of approximately

NOK 40 million, and the dredging and disposal at a near shore confined disposal site at Haakonsvern, Bergen, performed in 2003 at a cost of approximately NOK 185 million. Later, several other projects have been conducted. The two first projects are briefly presented below, together with a few more recent projects that have contributed significantly to the knowledge of contaminated sediments and remediation measures.

6.1.1 Eitrheimsvågen

In Eitrheimsvågen, previous deposits of production waste in the littoral zone had led to high pollution of heavy metals, especially in the sediment in the Inner part of the Sørfjorden. In 1986, a 740 m long sealing barrier was placed around the waste deposit site, and in 1992 further measures to reduce the leaching to the sea was carried out. In 1992 the bay Eitrheimsvågen was capped with 95,000 m^2 geotextile and a 30 cm thick layer of sand. In 2001, sampling of the sediment surface showed that the sand capping had become re-contaminated by discharges and run-off from land (Walday 2002). The capping in Eitrheimsvågen was again very polluted with concentrations of Zn, Pb and Hg to almost the same levels as before the remediation action (Skei, Pedersen *et al*. 1987, Walday 2002). The capping itself was successful, but due to insufficient control of discharges and run-off, the cap was re-contaminated.

6.1.2 Haakonsvern, Bergen

The sediment outside Haakonsvern naval base was contaminated with PCB, PAHs, heavy metals (Hg, Pb, Cu, and Zn) and TBT, and the Norwegian Defense Estates Agency initiated clean up in sea and at land. In 2003 contaminated sediment was dredged and nearshore confined disposals (CDFs) were built with natural dewatering to the sea through filters in the CDFs (Lone 2003). Remediation actions have also been done on shore to stop further contamination. The aim of the remediation actions was to remove as much as possible of the contaminants to achieve reduced concentrations of PCB in the sediment (not higher than 50 µg/kg). The dredging at Haakonsvern with the establishment of near shore confined disposal sites was evaluated to be technically successful.

6.1.3 Five pilot projects

Five pilot projects were performed in Tromsø, Trondheim, Sandefjord, Kristiansand and Horten during the period 2001/2002 in order to gain new knowledge, learn how clean-ups of contaminated sediments could best be organized and carried out, as well as to gain more practical experience. About NOK 100 million was spent on the pilot projects, of which the Environment Agency contributed NOK 60 million.

The focuses of the different pilot projects were as follows:

- Tromsø: Biological effect measurements and risk assessment connected with contaminated sediments;
- Trondheim: Remediation techniques, in particular in association with the stabilization of sediments in land disposal sites (mixing in of cement and fly ash) which created new areas for harbour activities;
- Sandefjord: Testing of different dredging techniques as well as deposition in shallow water disposal sites and use of geotextile bags;
- Kristiansand: Effects of capping contaminated sediment with sand;
- Horten: Release and toxicity of TBT with respect to land disposal of contaminated sediments.

The pilot projects provided new knowledge and important experiences, and contributed to increasing the understanding of contaminated sediments as an environmental problem. The most important lessons learned from the pilot projects were listed in the report from the Norwegian Council on Contaminated Sediments (Nasjonalt råd for forurensede sedimenter 2006) and are repeated here:

- Sediment remediation will seldom be 100% solutions. Residual contamination following dredging is a well-recognized problem. Recently, in large clean-ups like in Oslo harbour (see below), Trondheim and Sandefjord this has been solved by placing a protecting capping layer on top after dredging to avoid residual contamination. How clean a sediment is after remediation is related to the scope of the measure. The scope is largely determined by the costs as compared to the benefits of the measure;
- The effects of clean-up measures are complicated to document due to the influences of many different factors in the sea. It is important to do accurate and detailed site monitoring both before and after the clean-up, and to realize that complete recovery may be a slow process;
- Planning, project management and organization of measures are central elements in obtaining a successful result. This also involves a professional and open information strategy in order to avoid unfounded fears and misunderstandings;
- As a part of the decision-making basis for a remediation project, an assessment must be performed of the risk associated with the bioavailability of the sediment bound contaminants and their toxicity. Increased knowledge of environmental effects and experience with ecotoxicological studies are desirable. In particular, there is a great need for increased knowledge about the environmental risks associated with TBT, which are still subject to considerable uncertainty. TBT is extremely common in harbour sediments. For many practical purposes, toxicity based EQS for TBT is substituted with a much higher operational EQS value;

- The pilot projects have shown that it is important that sediment remediation should not be over-focused on hot spot areas (often small areas with a high level of contamination in the sediments). The clean-up within a small area that is surrounded by contaminated sediments will be of little value unless the area can be designated with certainty as a source area (large danger of dispersal). The interrelationship between a local remediation site and the environmental situation in the surrounding area must be understood and clarified so that realistic assessments can be made when evaluating the measure with respect to the environmental targets for the whole area;
- The pilot projects have also shown that it is possible to carry out sediment remediation at a somewhat lower cost than previously expected.

Extended remediation actions have later been conducted in all the pilot sites.

6.1.4 Kristiansand harbour area, Norway

Since 2001 there have been many sediment remediation projects in Kristiansand harbour area, based on the overall remediation action plan developed by the County Council. Metal industry, shipyard industry, and urban run-off have contaminated the sediment with PAHs, PCB, TBT, HCB, dioxins, and heavy metals. The local industry has implemented clean-up measures and reduced the discharges of contaminants to the sea. Contaminated sediment has been removed by dredging and excavation, and areas have been capped with sand, gravel, and some places geotextile and gravel. Concrete mattresses have also been used as capping on some steep slopes on the seabed. Waterside deposits were established to deposit contaminated sediment. In some areas erosion has damaged the capping layer, and there has also been re-contamination of the cap due to nearby dredging (Næs and Håvardstun 2010, Næs and Håvardstun 2013). The quality of the benthic fauna has improved significantly over the years, and might have reached a status as good as can be expected in a multi-source urbanized area (Næs, Håvardstun *et al*. 2014).

6.1.5 Oslo harbour area, Norway

In the spring of 2006, a major remediation operation was started in Oslo harbour. It involved the dredging of 650,000 m^3 of contaminated harbour sediment and the capping of approximately 1 million m^2 of contaminated sediment outside the dredging area. The project was finished in 2014. The sediment in the inner Oslofjord has been contaminated with heavy metals, PAHs, PCBs, TBT and various organic contaminants due to decades of discharges and run-off from industry and various urban activities. It is estimated that 95–99% of the total volume of contaminated sediment was removed from the planned clean-up area within the Oslo harbour area (Pettersen 2014). The dredged contaminated sediment was deposited in a deep deposit area (70 m) and capped with a 40 cm layer of sand. In 2015, a survey verified that the concentration of contaminants was much lower than before the remediation actions were done, but occasionally higher than in 2013 (Slinde 2015). Thus, the survey indicated some re-contamination. Run-off from the city and resuspension from sediment outside the remediated areas are likely sources for the re-contamination. In one harbour area the capping layer was severely eroded, probably due to erosion caused by ship propellers. The capping layer of the deep deposit site has not been affected by erosion and recolonization to good biodiversity indices occurred within a few years after the operation. There has been some recolonization of soft bottom fauna at the sediment in the harbour areas.

6.1.6 Ormefjorden and Eidangerfjorden in the Grenlandfjord area, Norway

The Grenlandfjord area in South-Eastern Norway is a large area of several fjords suffering from severe pollution including dioxins/furans from an earlier magnesium plant. In 2009, a pilot study on thin-layer capping with activated carbon based on a large field experiment was conducted in the Ormefjorden and Eidangerfjorden (Cornelissen, Amstaetter *et al*. 2012, Schaanning and Allan 2012), financed by the Norwegian Research Council (project Opticap), Norsk Hydro (project Thinc), the Norwegian Environment Agency (project BEST) and with contributions also from the Swedish Formas VINNOVA project (Carbocap). The thin-layer capping was tested in order to reduce transfer of dioxin/furans from sediment to biota. The County Council of Telemark did an evaluation of thin-layer capping and AC amendment to enhance the natural recovery of the fjords (Olsen 2012). Based on a semi-quantitative assessment (see chapter 7), the County Council concluded that the knowledge, experience and results from all studies included in the projects Opticap, Thinc, Carbocap and BEST indicated support of thin-layer capping with AC amendment as a remediation approach, though more knowledge was needed on the long-term efficiency of the method and the adverse effects of AC amendment.

At two occasions in the following years, the long-term effect of capping on benthic habitat quality and biodiversity has been monitored with SPI-images and macrofauna investigations, and the effect on bioavailability of dioxins/furans has been investigated from chemical analyses of sediment and flux measurements in situ and in mesocosms. Reduced leakage of dioxins/furans to the overlying water was observed on fields capped with clay and activated carbon (Schaanning and Allan 2012, Eek *et al*. 2014, Schaanning, Beylich *et al*. 2014, Cornelissen, Schaanning *et al*. 2016, Samuelsson, Raymond *et al*. 2017). Reduced bioaccumulation in Hinia reticulata and Nereis diversicolor was found in sediments capped with clay and activated carbon (Schaanning and Allan 2012, Schaanning, Beylich *et al*. 2014). However, monitoring has shown loss of individuals, species and diversity compared to the reference fields (Schaanning, Beylich *et al*. 2011), with up to 90% reduction in abundance, biomass, and number of species (Samuelsson, Raymond *et al*. 2017). A nine-year follow-up study of cap-efficiency and recolonization has been funded by The Norwegian Environment Agency in 2018–19.

7. How to decide upon remediation strategies

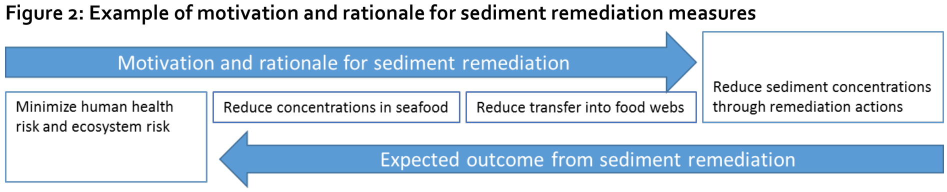
7.1 Resume

The risk-based approach to define clean-up levels opens for assessment of site-specific conditions and bioavailability of contaminants. Norwegian guidelines for risk assessment of contaminated sediments is based on a tiered approach where Tier 1 equals the EQS and Tier 3 includes site-specific data. However, few project owners collect data enough to assess risk and develop Tier 3 clean-up levels based on bioavailability and site-specific conditions. Multiple decision criteria for remediation strategy should be used to make the decision-making process transparent. The decision criteria Efficiency, Benefits, Adverse effects and Cost have been used in the development of several remediation action plans in Norway.

7.2 Defining clean-up levels

The clean-up of contaminated sediments is often motivated by the need to reduce risks to human health and the environment, especially from contaminated seafood (fish, etc.). Often, there is an expectation that remediation actions will lead to the lift of consumption bans. Accordingly, remediation actions may be based on a rationale that reduced contaminant concentrations in fish and seafood will follow from reduced sediment concentrations (figure 2). However, there are contradicting results on the importance of legacy contaminated sediments to fish tissue levels of certain contaminants such as Hg, and the association between levels of contaminants in sediments and consumption advice is not straight forward.

Figure 2: Example of motivation and rationale for sediment remediation measures



Traditionally, clean-up strategies have focused on removal or capping contaminated sediments in order to reduce the sediment concentrations within the biologically active zone, even though the risk is more related to exposure to porewater concentrations than to total sediment concentrations. In Norway, remedial actions typically include dredging to depths that expose pre-industrial (clean) sediments or installation of thick isolation caps to reduce the surface sediment concentrations to near natural background conditions. With such approaches, recontamination due to resuspension and horizontal transport of contaminated sediments from adjacent areas, diffuse or permitted industrial discharges or atmospheric input has become a growing concern. Therefore, clean-up goals based on natural conditions may be impossible to achieve in urban areas, but risk reduction measures may still be attainable.

Recently, activated carbon and other active materials have been shown to reduce the bioavailability of various contaminants in aquatic sediment, measured as reduced porewater concentrations and reduced flux from sediment to water. The use of active materials may reduce the thickness of the capping layer to a few centimetres (thin-layer capping), thus also potentially reducing remedial action costs. However, the use of active materials and thin-layer capping may not meet standard sediment clean-up goals (such as EQS) because these materials do not destroy contaminants, they simply sequester them. The introduction of low-impact remediation strategies like activated carbon amendment demonstrates the importance of a risk-based approach as well as the need to include multiple assessment criteria. This approach includes bioavailability (pore water concentrations) and ecological risk reduction rather than total sediment concentrations (EQS), as well as the potential adverse effects of remediation.

The risk-based approach to define clean-up levels opens for assessment of site-specific conditions and bioavailability of contaminants. In Norway, guidelines for risk assessment of contaminated sediments (Guideline M-409/2016) is based on a tiered approach. Tier 1 sediment clean-up levels established in Norway are based on screening levels ranging from natural background to severe impairment – the goal is often to achieve sediment concentrations below the lowest two levels (see chapter 2.2), which is harmonized with EU's EQS. Higher clean-up levels can be accepted if Tier 2 and 3 studies are performed, including site-specific values for the parameters included in the risk assessment, like partition coefficients and organic content in sediments. However, few project owners collect data enough to assess risk and develop Tier 3 clean-up levels based on bioavailability and site-specific conditions.

Identification of contaminated sediments should take into account the potential additive effects of pollutants and not just the concentration of single pollutants. This is because combined effects have been shown to occur even when the pollutants are present in concentrations below their individual No Observed Effect Concentrations (NOECs) (Faust *et al*. 2001, Kortenkamp *et al*. 2009). Additive effects are believed to be the default of mixtures of contaminants affecting the same endpoint in the same direction and this has now been acknowledged by regulatory authorities with the publication of a draft guidance on harmonized methodologies for human health, animal health and ecological risk assessment of combined exposure to multiple chemicals (Hardy *et al*. 2018).

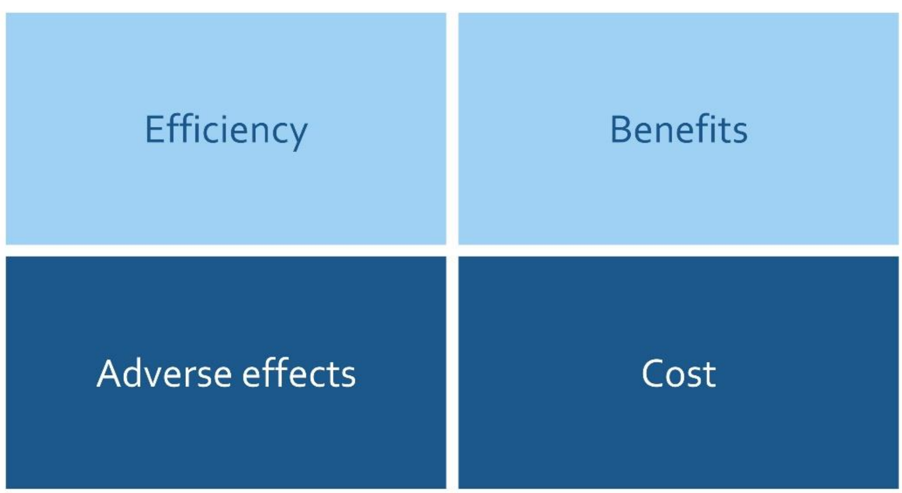
7.3 Decision criteria for remediation strategy

The report issued by the Norwegian council of Contaminated Sediments in 2006 (Nasjonalt råd for forurensede sedimenter 2006) concluded that there was a need for suitable tools for quantifying uncertainty in goal attainments and the risk of failure in clean-up projects, to be able to make the decision-making process transparent. The decision-making process can become difficult if it cannot be demonstrated that the environmental benefit can be quantified and documented.

Traditionally, dredging and isolation capping have aimed at reducing concentrations of hazardous substances in the sediments without taking into consideration the potential adverse effects of the remediation action. A more balanced evaluation of less invasive remedial measures such as *in situ* treatments, can be achieved by broadening the decision context to include all relevant factors such as short- and long-term ecological impacts and benefits, residual impacts, and performance. Such an assessment could also involve a comparative life cycle assessment (Ghosh, Luthy *et al*. 2011).

In the thin-layer capping pilot study in the Grenlandfjords in Norway, a semi-quantitative assessment was conducted (Olsen 2012) to decide upon the feasibility of the remediation method tested, based upon the criteria Efficiency, Benefits, Adverse effects and Cost (figure 3).

Figure 3: Examples of criteria for assessment of feasibility of remediation approach



Source: After Olsen (2012).

These criteria have later been included in the development of remediation actions plans for contaminated sediment sites developed by NIVA for Gunneklevfjorden (Grenlandfjords) and Elkembukta (Kristiansand) in Norway. The suggested criteria are presented in brief below:

7.3.1 *Efficiency*

The efficiency of a remediation method will reflect the methods' ability to reduce the sediment concentrations and the risk of environmental effects of single pollutants and chemical mixtures, including the uptake in biota and aquatic food webs. Several factors may influence on the efficiency, such as the chosen sediment remediation measure, the size of the area, bioturbation, resuspension of sediment due to disturbance such as erosion and in particular propeller activity, and recontamination of the area due to active sources.

In general, most sediment remediation measures will be effective on a broad range of contaminants, or groups of contaminants. Measures involving removal of contaminated sediment by dredging will effectively remove all types of contaminants (residual concentrations are expected). In situ amendments like capping creates a physical barrier between the polluted sediment and the new sediment layer and will be able to target a wide range of pollutants. Different types of pollutants will have different abilities to leak out from the cap depending on the capping material used. The same will be the case for geotextiles. In situ amendments using sorbents can be targeted to more specific chemical groups. For instance, AC are very efficient for retaining organic compounds, whereas apatite is more efficient for immobilizing heavy metals.

7.3.2 *Benefits*

The benefits from sediment clean-up measures will be closely linked to the objectives defined for the measure. In addition, cleanup measures can provide positive effects that are not directly measurable immediately after the remediation, for example related to recreation.

7.3.3 *Adverse effects*

In the summary of experience from the early Norwegian pilot projects (Nasjonalt råd for forurensede sedimenter 2006), no comments are made to the possible adverse secondary effects of remediation and the time to recovery/recolonization of the site after destructive measures like dredging and isolation capping. However, this aspect has gained increasing attention in recent years, and potential habitat destruction and recovery as well as effects on biodiversity are relevant to consider. Also, according to the Water Framework Directive there is a goal to achieve Good Ecological Status. Following dredging operations there are several challenges related to water treatment and deposition of dredged material, which may give adverse effects to the environment. The pilot study in the Grenlandfjords area considered the adverse effects of thin-layer capping including AC amendment and found negative effects on benthos (Schaanning, Beylich *et al*. 2014, Samuelsson, Raymond *et al*. 2017). There are indications that the potential effects of AC amendment depend on the quality as well as the particle size of AC, in addition to depend on the various feeding strategies of benthic organisms.

In a broader life cycle perspective, various capping strategies may have global impact depending on the capping material in use, including aspects like use of resources (e.g. mining of apatite), energy consumption and discharges related to production and transport of capping material. Hence, MNR is a way to avoid the adverse effects from remediation.

7.3.4 Costs

The costs of remediation have traditionally been the primary criteria for deciding upon remediation actions. The size of the area and the solutions for deposits of dredged material or the capping material available will strongly influence the costs. Often, the combination of infrastructure projects can give opportunities for synergies that may lower costs. This has been the case in several remediation projects in Norway, where material from road construction has been used as capping material (Kristiansandfjorden), or dredged material has been stabilized and used for land extension (e.g. at quay 2 in Trondheim harbour) or utilized for harbour extensions (Tromsø, Sandvika). The latter is examples of shoreline deposits which are beneficial due to controlled (safe?) storage of contaminated sediments, short transportation (low risk, low cost) and gain of valuable building plots. In Sandvika, stabilization of dredged, contaminated sediments was used to expand a quay-front area for recreational purposes.

7.4 Multiple Criteria Analyses

The discussion of multiple evaluation criteria to decide upon remediation strategy, demonstrates the complexity and the need to do site-specific assessments prior to a decision. There are tools developed to support multiple criteria decisions, such as Multiple Criteria Analyses (MCA). However, the quality of data available to describe and assess each decision criteria is essential.

7.5 Climate change impacts

An overview of climate change impacts that can potentially affect remediation vulnerability was prepared by US EPA in 2015 and lists factors related to precipitation, temperature, sea level rise, wind and wildfires. The listed climate change impacts include increased heavy precipitation events, increased flood frequency and intensity, decreased precipitation and increasing drought, increased landslides, increased occurrence of extreme temperatures, sustained changes in average temperatures, sea level rise, increased intensity of wind, and increased frequency and intensity of wildfires (United States EPA 2015, 2017). Several of these climate related changes can increase the risk for more discharge of contaminants to the sea. A climate-change exposure assessment can be performed to reflect on a range of possible climate and weather scenarios.

8. Recommendations

8.1 Resume

Recommendations are given on the need for future monitoring, research and method development, as well as improved decision tools for remediation strategy. A Nordic database and joint guidelines and harmonization between the Nordic countries should be considered.

8.2 List of recommendations

- Finland needs comprehensive survey of suspected sites and national strategy for dealing with contaminated sediments;
- Risk based approach is beneficial to identify contaminated sediments, prioritize between sites and decide clean-up levels;
- Research and method developments are needed for the development of risk assessment tools for multiple contaminants;
- Further research (both field and laboratory) is needed for new and promising techniques;
- The long-term effect of low impact strategies need to be further investigated;
- Consider multi-criteria analyses to decide upon remediation strategy, taking into account criteria such as adverse effects in addition to benefit, cost and efficiency;
- A Nordic database of clean-up projects and long term monitoring would improve the exchange of knowledge between the Nordic countries;
- Monitored Natural Recovery should be given more consideration as an alternative remediation strategy;
- The guidelines for contaminated sediments are varying a lot between the Nordic countries. The Nordic countries should consider the possibility to have joint guidelines for contaminated sediments.

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Sammendrag

Nordisk Ministerråd har sett behovet for å samle kunnskap om håndtering av forurensede sedimenter, inkludert ulike tiltaksalternativer i forhold til kostnadseffektivitet og effekter på miljøet for å sikre at best mulig tiltak gjøres i fremtiden. Hensikten med denne rapporten er å gi en oversikt over forekomst, regelverk og håndtering av forurensede sedimenter i utvalgte nordiske land (Norge, Sverige, Finland og Danmark). I tillegg gir rapporten en kort oversikt over ulike tiltaksalternativer, teknikker og materialer.

Forurenset sediment i utvalgte nordiske land

Forurensede sedimenter er globalt ansett som en betydelig kilde til utlekking av tidligere industriforurensning som følge av historiske utslipp av miljøgifter. Forurensede sedimenter bidrar potensielt med utslipp av forurensninger i flere tiår etter at regulære industrielle utslipp har opphørt. Forurensede sedimenter er også ansett som en potensiell kilde til opptak av miljøgifter i akvatiske næringskjeder. Over hele verden har en rekke forurensede lokaliteter blitt identifisert og gitt prioritet i nasjonale tiltaksprogram. Myndighetenes regulering og håndtering av forurensede sedimenter varierer mellom land. I Norge har de norske miljømyndighetene iverksatt en nasjonal strategi for håndtering av marine forurensede sedimenter i havner og fjorder. Dette har ført til identifisering av prioriterte områder, utvikling av handlingsplaner og igangsetting av oppryddingsprosjekter. De andre nordiske landene har ingen tilsvarende nasjonal strategi for opprydding i forurensede sedimenter.

Klassifisering av sedimenter basert på konsentrasjoner av definerte stoffer eller stoffgrupper, blir ofte brukt som et verktøy for å identifisere potensielle tiltakslokaliteter. Klassifiseringen kan baseres på toksisitet og potensiell miljørisiko eller på overskridelse av definerte terskel- eller bakgrunnsnivåer. Vannrammedirektivet (WFD) er implementert i EU-land og andre europeiske land med sikte på å oppnå "god status" for alle grunnvann og overflatevann i EU basert på miljøkvalitetsstandarder (EQS) utviklet og implementert under WFD. I Norge er kjemisk status hovedsakelig basert på nasjonalt definerte EQSer for sediment (fastsatt for 28 EU-prioriterte stoffer) og biota (fastsatt for 23 EU-prioriterte stoffer), mens i Finland er kjemisk status for vannlokaliteter basert på EQSer definert for overflatevann og biota. Finland har ikke etablert nasjonale EQS-verdier for sediment. I Danmark og Sverige er god kjemisk status basert på EQS for vann, sediment og biota, men Sverige har fastsatt EQS kun for noen få stoffer i biota og sediment.

Forurensede sedimenter i Norge er hovedsakelig relatert til havner og fjorder med industriell aktivitet i form av prosessindustri, treforedling og papirindustri, og skipsverft, samt skipsfart. I dag er industriens utslipp strengt regulert, mens de betydelig høyere utslippene fra tidligere tider har ført til forurensede sedimenter i mange fjorder. Naturlig forbedring gjennom naturlig tildekking skjer vanligvis i fjordområder der elver tilfører rene partikler som legger seg oppå eldre forurenset sjøbunn, men sedimenteringshastigheten varierer og avhenger av stedsspesifikke forhold. Undersøkelser i perioden fram til tusenårsskiftet viste at mer enn 120 lokaliteter har høye sedimentkonsentrasjoner av miljøfarlige stoffer. Opprydding av forurenset sjøbunn har vært et prioritert satsningsområde for norske myndigheter helt siden slutten av 1980-tallet. For 17 prioriterte fjorder er det utarbeidet regionale handlingsplaner for forurenset sjøbunn. På bakgrunn av dette har Miljødirektoratet gitt pålegg til lokale virksomheter om videre undersøkelser og utvikling av områdespesifikke tiltaksplaner for sine vannresipienter, og i noen tilfeller også pålegg om gjennomføring av tiltak. Oppryddingstiltak som følge av regionale handlingsplaner finner typisk sted i nært samarbeid mellom lokale, regionale og/eller nasjonale miljømyndigheter. Etter år 2000 har det vært avsatt statlige midler til opprydding av forurensede sedimenter og forurenset jord. For å støtte arbeidet med opprydding av forurensede sedimenter har myndighetene utarbeidet et sett med veiledere, inkludert veileder for håndtering av sediment og veileder for risikovurdering av forurensede sedimenter, der prosessen for å vurdere behovet for tiltak er beskrevet.

I Sverige viste en foreløpig gjennomgang i 2016 av type og forekomst av forurensede sedimenter i innlandet og/eller kystvann i hvert av Sveriges 21 fylker, at forurensede mineralbaserte (minerogene) og/eller cellulosebærende ("fiberbank") sedimenter forekommer i minst 19 fylker. På mange lokaliteter utgjør forurensede sedimenter sannsynligvis en uakseptabel risiko for miljøet og/eller menneskers helse selv om svært få forvaltningsbeslutninger er tatt. Det svenske Narturvårdsverket er det statlige organet som er ansvarlig for miljøspørsmål, og mer presist for å koordinere, prioritere og følge opp arbeidet med miljøspørsmål på nasjonalt nivå i samarbeid med andre myndigheter. Det svenske geotekniske instituttet har det nasjonale ansvaret for forskning, teknologisk utvikling og kunnskapsbygging for opprydding og restaurering av forurensede lokaliteter. Narturvårdsverket har besluttet at forurensede områder der det ikke finnes en tydelig ansvarlig problemeier, vil bli ivaretatt av statlige midler. I 2017 besluttet regjeringen å fordele midler til opprydding av forurenset jord og sedimenter i perioden 2018 til 2020. Sveriges mest omfattende oppryddingsprosjekt for forurensede sedimenter er Oskarshamns havn, som har hatt en meget aktiv industrihistorie siden midten av 1980-tallet. Oppryddingsarbeidet startet i 1996, og mudringsvirksomheten i 2016.

I Finland vurderes forurensningstilstanden for sediment kun i sammenheng med deponering av mudrede masser. Det har ikke vært systematiske og landsomfattende undersøkelser i Finland for å identifisere forurensede sedimenter, selv om en innledende nasjonal undersøkelse av forurensede sedimenter i innlandsvann identifiserte 28 mulige eller allerede avklarte lokaliteter over hele landet. Bare noen få har blitt fulgt opp i form av tiltak. Det foreligger ingen veiledningsdokumenter for å vurdere miljøfare, og det eneste sedimentrelaterte veiledningsdokumentet gjelder for håndtering av mudrede masser. Det finske Miljøverndepartementet fastsetter prinsipper og retningslinjer for gjennomføring av EU-direktiver og nasjonal lovgivning. Miljølovgivningen settes i kraft via det praktiske arbeidet til de regionale sentrene for økonomisk utvikling, transport og miljø (ELY), eller på by- eller kommunenivå. Den kjemiske statusen til vannlokaliteter, basert på WFD, overvåkes og klassifiseres av de regionale ELY-sentrene. Miljømyndighetenes forvaltning av sedimenter er oftest knyttet til mudringsaksjoner utført for å holde vannveier åpne for navigasjon, eller konstruksjoner i havner. Bruk av vannveier og vannmiljøet er derfor den primære drivkraften for sedimenttiltak og ikke miljøhensyn isolert sett.

I Danmark er kystvannet sterkt påvirket av menneskeskapt aktivitet, både fra landbaserte og havbaserte aktiviteter som akvakultur, skipsfart og industri. Mudring synes å være den viktigste metoden for fjerning eller håndtering av (forurensede) sedimenter i Danmark. Innholdet av farlige stoffer er inkludert som et viktig element i evalueringen av hvordan sedimenter og mudringsmaterialer kan håndteres. Miljøfarlige stoffer i danske kystområder har blitt overvåket i hele landet siden 1998 gjennom Danmarks nasjonale overvåkningsprogram for vannmiljø og natur, NOVANA. Det danske Miljø- og Fødevareministeriet er ansvarlig for vannplanleggingen i Danmark, og for å overvåke tilstanden til overflatevann, grunnvann og beskyttede områder. De indre danske farvannene er generelt klassifisert som problemområder med hensyn til kjemisk status. En foreløpig gjennomgang av danske sediment-data og sammenligning med terskelverdier som vanligvis brukes i OSPAR og landene i studieområdet, viste at 36% (28 lokaliteter) av de vurderte områdene hadde høy eller god kjemisk status. De fleste av de vurderte lokalitetene hadde en moderat kjemisk status (55%, 42 steder), og bare 7 steder (9%) ble vurdert som dårlig eller svært dårlig.

Kort oversikt over tiltaksløsninger

Historisk sett har den vanligste strategien for opprydding av forurensede sedimenter vært fjerning av det forurensede sedimentet ved mudring, og avhending/behandling på deponi. Isolering av forurensede sedimenter ved tildekking med og uten en aktiv barriere har også blitt benyttet i enkelte land. Andre tilnærminger, inkludert bruk av tynnsjikt-tildekking med aktive materialer, stabilisering på stedet og overvåket naturlig forbedring (NMR) er utviklet, men i mindre grad tatt i bruk. Resultater fra utprøving med aktive materialer er lovende for videreutvikling av kostnadseffektive tiltak med liten innvirkning på akvatiske organismer.

Erfaringer fra oppryddingsprosjekter

Det første oppryddingsprosjektet av forurensede sedimenter i Norge var *in situ*-tildekking av Eitrheimsvågen i Odda utført i 1992 til en kostnad av ca. 40 millioner kroner. I perioden 2001/2002 ble det gjennomført fem pilotprosjekter i Tromsø, Trondheim, Sandefjord, Kristiansand og Horten for å bygge kunnskap og få praktisk erfaring og lære hvordan opprydding av forurensede sedimenter best kunne organiseres og gjennomføres. I 2003 ble mudring og deponering utført ved Haakonsvern, Bergen. Senere er det gjennomført flere fullskalaprosjekter som for eksempel Kristiansand havn, Oslo havn og Tromsø havn.

Et av de viktigste kunnskapsbehovene som ble identifisert av *Nasjonalt råd for forurensede sedimenter* i 2006 var relatert til langsiktig overvåking av lokalitetene etter at tiltak var gjennomført for å vurdere effektiviteten av tiltakene og potensiell rekontaminering, og for å bygge kunnskap til fremtidige oppryddingsprosjekter.

I Sverige er det rapportert noen oppryddingsprosjekter, og referanseprosjekter finnes på http://atgardsportalen.se/. Disse involverer for det meste mudring og deponering, men også noen tildekkingsprosjekter. I Finland er det kun noen få oppryddingsprosjekter som er motivert av hensyn til miljøet, mens flere prosjekter er motivert av infrastrukturutvikling.

Hvordan beslutte oppryddingsstrategi

Den risikobaserte tilnærmingen for å definere tiltaksmål åpner for vurdering av stedsspesifikke forhold og biotilgjengelighet av stoffene. Den norske veilederen for risikovurdering av forurensede sedimenter er basert på en trinnvis prosess hvor Trinn 1 tilsvarer EQS, og Trinn 3 inkluderer stedsspesifikke verdier for parameterne som inngår i risikovurderingen, slik som partisjonskoeffisienter og organisk innhold i sedimenter. Ofte er det imidlertid ikke innhentet tilstrekkelig med data til å kunne vurdere stedsspesifikk risiko etter Trinn 3 eller etablere risikobaserte tiltaksmål med utgangspunkt i biotilgjengelighet av de aktuelle miljøgiftene.

Beslutningskriterier for valg av tiltak bør synliggjøres for å gjøre beslutningsprosessen mest mulig gjennomsiktig. Beslutningskriteriene Effektivitet, Nytte, Ulemper/konsekvenser og Kostnader har vært brukt i utviklingen av flere tiltaksplaner i Norge. Multikriterieanalyser for å avgjøre oppryddingsstrategi får fram kompleksiteten og behovet for å gjøre stedsspesifikke vurderinger før en beslutning om tiltak tas. En vurdering av klimaendringer kan utføres for å reflektere over en rekke mulige klima- og værscenarier og mulig innvirkning på forurensningssituasjonen.

Anbefalinger

- Av de nordiske landene som inngår i vurderingen, er det særlig for Finland behov for omfattende undersøkelse av antatte lokaliteter med forurensede sedimenter og en nasjonal strategi for håndtering av forurensede sedimenter.
- Risikobasert tilnærming er gunstig for å identifisere forurensede sedimenter, prioritere mellom steder og fastsette tiltaksmål.
- Forskning og metodeutvikling er nødvendig for utvikling av risikovurderingsverktøy som tar hensyn til potensiell additiv risiko ved samtidig tilstedeværelse av flere miljøfarlige stoffer.
- Videre forskning (både felt og laboratorium) er nødvendig for utvikling av nye og lovende tiltaksløsninger.

- Den langsiktige effekten av tiltaksløsninger med aktive materialer må undersøkes nærmere.
- Overvåket naturlig forbedring bør gis mer aksept som en alternativ tiltaksstrategi.
- Multikriterieanalyser for å avgjøre oppryddingsstrategi bør vurderes og bør inkludere mulige negative effekter av tiltak i tillegg til nytte, kostnad og effektivitet.
- En nordisk database med oppryddingsprosjekter og resultater fra langsiktig overvåking vil styrke kunnskapsutvekslingen mellom de nordiske landene.
- Retningslinjer for forurensede sedimenter varierer mellom de nordiske landene. De nordiske landene bør vurdere muligheten for å ha felles retningslinjer for forurensede sedimenter.

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Contaminated Sediments: Review of solutions for protecting aquatic environments

The Nordic Council of Ministers have recognized the need to compile the knowledge of sediment remediation strategies and to evaluate different remediation approaches. The objective of this report is to give a status of regulatory framework and sediment management within selected Nordic countries (Norway, Sweden, Finland and Denmark). In addition, the report gives a short overview of international knowledge on remediation approaches, techniques, and materials. Governance and regulatory instruments in dealing with contaminated sediments vary between countries. Of the countries presented, only Norwegian environmental authorities have implemented a national strategy for remediation of marine contaminated sediments. A Nordic database of clean-up projects and long-term monitoring would improve the exchange of knowledge between the Nordic countries.

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