



environmental affairs

Department:
Environmental Affairs
REPUBLIC OF SOUTH AFRICA

REVIEW AND UPDATE OF SOUTH AFRICA'S NATIONAL ACTION LIST FOR THE SCREENING OF DREDGED SEDIMENT PROPOSED FOR MARINE DISPOSAL

Table of Contents

	Page
1. Introduction	1
2. About this report	1
3. The London Convention and London Protocol	1
4. The Integrated Coastal Management Act	2
5. What is an Action List?	2
6. Contaminated sediment	3
6.1 What is contaminated sediment?	3
6.2 The significance of contaminated sediment	3
7. Environmental effects of dredging	5
8. Sediment quality guidelines	8
8.1 What are sediment quality guidelines?	8
8.2 Why develop sediment quality guidelines, and how should they be used for decision-making?	9
8.3 Sediment quality guidelines and uncertainty	10
9. Approaches for deriving sediment quality guidelines	11
9.1 Theoretical approaches	12
9.1.1 Sediment Background approach	12
9.1.2 Equilibrium Partitioning approach	13
9.1.3 Acid Volatile Sulphide approach	14
9.2 Empirical approaches	14
9.2.1 Apparent Effects Threshold approach	14
9.2.2 Screening Level Concentration approach	15
9.2.3 Spiked Sediment Toxicity Test approach	16
9.2.4 Effects Range approach	17
9.2.5 Effects Level approach	18
9.2.6 Logistic Regression Modelling approach	18
9.2.7 Consensus-based approach	19
9.2.8 Floating Percentile approach	19
10. Limitations of sediment quality guidelines	19
10.1 Sediment quality guidelines are protective of a limited number and type of potential receptors	19
10.2 Sediment quality guidelines are not available for all chemicals	20
10.3 Sediment quality guidelines do not address unanticipated chemicals	20
10.4 Sediment quality guidelines do not address the interaction of chemicals	20
10.5 Sediment quality guidelines do not adequately consider the exposure component of environmental risk	20
10.6 Sediment quality guidelines are often poor predictors of adverse biological effects	20
10.7 Sediment quality guidelines do not account for bioavailability	20
10.8 Sediment quality guidelines do not account for bioaccumulation/ biomagnification	21
11. The need for multiple lines of evidence when assessing sediment quality	21
11.1 Sediment chemistry	21
11.2 Biological assessment	22
11.2.1 Sediment toxicity tests	22
11.2.2 Benthic invertebrate community assessment	23
11.2.3 Bioaccumulation assessment	23

12.	Action Lists from various jurisdictions/countries	24
12.1	Europe	24
12.1.1	Belgium	24
12.1.2	Denmark	25
12.1.3	Finland	25
12.1.4	France	25
12.1.5	Germany	26
12.1.6	Ireland	26
12.1.7	Spain	26
12.1.8	United Kingdom (England and Wales)	27
12.2	North America	27
12.2.1	United States of America	27
12.2.2	Canada	28
12.3	Australasia	28
12.3.1	Australia	28
12.3.2	New Zealand	29
12.4	Asia	30
12.4.1	Hong Kong	30
13.	Existing South African National Action List	30
14.	Limitations of the existing National Action List	32
14.1	Limitation 1: Some guidelines are inappropriate for South African coastal waters	32
14.1.1	Basis, definition and practical application of baseline concentrations for metals in sediment	33
14.1.2	Metal baseline concentrations in sediment from South African coastal waters	36
14.1.3	Inappropriate sediment quality guidelines of the existing National Action List	36
14.2	Limitation 2: Some guidelines are over-protective and some possibly under-protective	38
14.3	Limitation 3: Additive effects	39
14.4	Limitation 4: Confusing terminology	39
14.5	Limitation 5: Biological testing	40
14.6	Limitation 6: Too much emphasis on metals	40
15.	Implications of the London Convention and London Protocol for revision of the National Action List	41
16.	Implications of the Integrated Coastal Management Act for revision of the National Action List	41
17.	Factors to consider and options for revising the sediment quality guidelines of the existing National Action List	41
17.1	Should sediment quality guidelines be defined for different management objectives?	42
17.2	Narrative intent of sediment quality guidelines	42
17.3	Derive sediment quality guidelines using empirical data	43
17.4	Derive sediment quality guidelines using the Sediment Background approach	44
17.5	Define sediment quality guidelines using the Equilibrium Partitioning approach	46
17.6	Adopt sediment quality guidelines from another jurisdiction/country	46
17.7	Remove or replace only the inappropriate guidelines of the existing National Action List	49
17.8	Formulate sediment quality guidelines using a hybrid approach	49
18.	Identification of candidate sediment quality guidelines for the revised National Action List	49
19.	Candidate sediment quality guidelines for the revised National Action List	51
20.	Sediment quality guideline example	20

21.	Other factors for consideration	59
21.1	Should sediment quality guidelines be formulated separately for different regions of South Africa?	59
21.2	Absence of sediment quality guidelines for other chemicals	60
22.	Decision-making for the revised National Action List	61
22.1	Narrative intent of the Level I and Level II guidelines	61
22.2	Should some metals be ‘weighted’ more than others in decision-making?	63
22.3	Chemical mixtures	64
22.4	Decision criteria	65
23.	Some factors that influence decision criteria	66
23.1	Nature of the dredged material	66
23.2	Nature of dredging site	66
23.3	Disposal site characteristics	66
23.4	Sampling design considerations	67
23.4.1	Consideration of dredged material management units	67
23.4.2	Number of samples, compositing of samples, and re-sampling	67
23.4.3	Depth of sediment sampled	68
24.	Options for further assessment	69
24.1	Chemical assessment	69
24.2	Biological assessment	70
25.	National Action List terminology	70
26.	‘Treatment’ considerations	71
27.	National dredging assessment framework	71
28.	References	72
29.	Appendices	78

1. Introduction

The South African government is a signatory to the London Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter (1972) (the London Convention) and to the 1996 Protocol to the London Convention (the London Protocol). The London Convention and London Protocol regulate the deliberate disposal of waste materials in the marine environment. In South Africa, the Integrated Coastal Management Act 2008 (Act 24 of 2008) (ICM Act) gives effect to the provisions of the London Convention and London Protocol. There are seven categories of waste and other material that are regulated under the ICM Act. Of these, the largest volume of material that requires disposal in offshore waters of South Africa is dredged material, derived predominantly from maintenance and capital dredging in ports.

The branch Oceans and Coasts of the Department of Environmental Affairs is mandated with the responsibility for regulating the disposal of materials in the marine environment off South Africa. As per the conditions of the London Convention and the London Protocol, Oceans and Coasts uses a National Action List to make decisions on whether sediment identified for dredging is of a suitable quality for unconfined, openwater disposal. The basis for the formulation and the manner in which the Action List is used for decision-making has not, however, been documented in sufficient detail. Furthermore, experience has shown that the existing National Action List is limited from several perspectives and needs to be revised. To this end Oceans and Coasts invited service providers to propose an approach for the revision of the National Action List, with the focus on the metal sediment quality guidelines and associated decision-making criteria. The focus of the revision is on the Action List *per se* and not on the formulation of a dredged material assessment framework, of which the National Action List will form a part. Through a competitive process, the Coastal and Marine Pollution research group of the CSIR was appointed to provide recommendations for the revision of the National Action List.

2. About this report

Sediment management is complex and multivariate, involving a careful balance of science, politics, and economics (Apitz et al. 2004). The National Action List will need to balance the sometimes conflicting needs of dredging proponents and the need to protect the ecology of the dredging and dredged spoil disposal environments. Because of conflicting needs and economic realities, the logical approach is to revise the

National Action List by consulting stakeholders. This report does not, therefore, present a revised National Action List, but rather provides stakeholders with information on factors that need to be considered and options for its revision. An invitation to a technical workshop hosted by the branch Oceans and Coasts of the Department of Environmental Affairs in Cape Town will be extended to certain stakeholders. The aim of the workshop is to develop an understanding of the needs, concerns and opinions of stakeholders on the revision of the National Action List.

To facilitate this process, the term Key Issue is used in the text of this report to highlight areas where stakeholder input is sought. The Key Issues are generally framed as questions. The Key Issues are also summarised in a separate Stakeholder Input report, which stakeholders are requested to complete and return to officials at the branch Oceans and Coasts of the Department of Environmental Affairs before the technical workshop. Stakeholders are encouraged to raise other issues that they feel have not been adequately addressed in this report. Stakeholder inputs will be consolidated and addressed to the extent practicable in the technical workshop. After the workshop, a draft technical report will be prepared, which will include the revised National Action List. The report will be circulated to a wider stakeholder audience for further comment before being finalised.

Although this report deals with technical issues, where possible these are described and discussed in a format that (hopefully) accommodates non-specialist audiences. The use of acronyms has been avoided wherever possible for this purpose.

3. The London Convention and London Protocol

As mentioned above, the deliberate disposal of waste and other matter at sea is regulated internationally by the London Convention and the London Protocol. Although other treaties and conventions have relevance in some areas of the world (e.g. the Oslo-Paris (OSPAR) Convention for the Protection of the North East Atlantic (1992)), none is relevant to South Africa. As a party to the London Convention and the London Protocol, the South African government has agreed to adhere to the pollution prevention principles encapsulated in these treaties, to enforce them under national law, and to report annually on disposal and monitoring activities.

The London Protocol entered into force on 24 March 2006. It modernises and will eventually replace the

London Convention, when all members of the London Convention are also parties to the protocol. For countries that are already parties to the London Protocol, the protocol supersedes the London Convention.

The London Protocol reflects the global trend towards precaution and prevention, with parties agreeing to move from controlled disposal at sea of a variety of land-generated wastes towards integrated land-based solutions for most, and controlled sea disposal of a few remaining categories of wastes or other matter. It introduces, amongst other improvements, the precautionary and polluter pays principles, and a new process for the assessment of wastes and other matter.

Under the London Protocol, all dumping of waste and other matter at sea is prohibited, except for possibly acceptable wastes on the so-called 'reverse list'. Whether these wastes are permitted for dumping is assessed using a procedure written into the London Protocol. The categories of waste include:

1. dredged material,
2. sewage sludge,
3. fish waste, or material resulting from industrial fish processing operations,
4. vessels and platforms or other man-made structures,
5. inert, inorganic geological material (e.g. mining wastes),
6. organic material of natural origin,
7. bulky items primarily comprising iron, steel, concrete and similarly unharmed materials for which the concern is physical impact, and limited to those circumstances where such wastes are generated at locations, such as small islands with isolated communities, having no practicable access to disposal options other than dumping.

The London Protocol states that 'Each Contracting Party shall develop a national Action List to provide a mechanism for screening candidate wastes and their constituents on the basis of their potential effects on human health and the marine environment.' Annex II of the London Protocol provides guidance on the assessment of wastes or other material that may be considered for dumping at sea. To provide guidance for implementing Annex II, Waste Assessment Guidelines were developed. The guidelines set out a procedure that should be followed by countries assessing proposals for sea disposal. Waste specific guidance has also been developed to assist in assessing each of the categories of materials that may be dumped under the protocol.

4. The Integrated Coastal Management Act

The ICM Act gives effect to South Africa's obligations to the London Convention and London Protocol. Section 73 of the Act states that (see Appendix 1):

- (1) The Minister must progressively and subject to available resources, develop a national action list to provide a mechanism for screening waste and other material on the basis of their potential effect on human health and the marine environment.
- (2) The national action list must—
 - (a) be developed in accordance with the Waste Assessment Guidelines set out in Schedule 2; and
 - (b) contain the prescribed information.

Schedule 2 referred to in part (2) above is a verbatim extract of Annex II of the London Protocol (see Appendix 2), that is, Guidelines for the Assessment of Wastes or Other Material that may be Considered for Dumping at Sea.

5. What is an Action List?

As mentioned above, the Waste Assessment Guidelines of the London Protocol state that 'Each Contracting Party shall develop a national Action List to provide a mechanism for screening candidate wastes and their constituents on the basis of their potential effects on human health and the marine environment.' As intimated by this statement, an Action List is a mechanism that allows managers and regulatory authorities to reach an informed decision on the suitability of waste material proposed for disposal in the marine environment. The Waste Assessment Guidelines state further that 'An Action List shall specify an upper level and may also specify a lower level. The Upper Level should be set so as to avoid acute or chronic effects on human health or on sensitive marine organisms representative of the marine ecosystem'. The upper and lower levels are, in the parlance of the Waste Assessment Guidelines, known as Action Levels. In other words, an Action List is comprised of one or more Action Levels. In the context of the disposal of dredged material, the Action Levels provide the decision criteria for determining whether dredged material (a) is suitable for unconfined, openwater disposal without further testing, (b) must undergo detailed testing before a decision can be made, or (c) is unacceptable for unconfined, openwater disposal and thus requires special management.

Action Levels may comprise one or more criteria that are used to identify whether the environmental concern is low or high for a particular characteristic of concern. The characteristics of concern in dredged material might include the concentrations of chemicals in the sediment, a biological response, or other characteristics that provide insight into the potential for the material to cause adverse biological effects in the marine environment.

An Action Level may, and often does, comprise criteria for a combination of physical, chemical and biological characteristics. In the context of dredged material, Action Levels should meet a number of general criteria including that they:

1. Should be meaningful for the dredged material characteristics and valued resources at issue,
2. Should focus on characteristics caused by anthropogenic impacts,
3. Should be sufficiently protective to minimise the probability of false negatives at the lower Action Level (i.e. reaching a conclusion that the dredged material poses no risk when in fact it does), and
4. Should be sufficiently accurate to minimise the probability of false positives at the upper Action Level (i.e. reaching the conclusion that a dredged material poses a risk when, in fact, it does not).

The Waste Assessment Guidelines state that 'In selecting substances for consideration in an Action List, priority shall be given to toxic, persistent and bio-accumulative substances from anthropogenic sources (e.g., cadmium, mercury, organohalogenes, petroleum hydrocarbons and, whenever relevant, arsenic, lead, copper, zinc, beryllium, chromium, nickel and vanadium, organosilicon compounds, cyanides, fluorides and pesticides or their by-products other than organohalogenes).'

The conclusions that are drawn from the use of Action Levels and the management decisions that follow are dependent on the technical basis of the Action Level derivation. Thus, Action Levels should be developed and applied with an understanding of what the Action Level is intended to protect and the technical argument linking the specific Action Level and protection objective. An upper Action Level is intended to provide a definitive decision point where the waste material under consideration may not be disposed unless further managed. Using an Action Level as the basis for reaching a conclusion that substances associated with the waste would likely cause adverse environmental effects if the waste were disposed requires that the Action Level be based on quantitative information about exposure (e.g.

concentrations of the substance organisms would contact) and effects (e.g. toxicology of the substance).

It is important at this stage to note that Action Lists and Action Levels, while analogous to screening level approaches such as sediment quality guidelines, have a different intent. The intent of an Action List is to provide a definitive decision point as part of a waste assessment process, whereas screening tools help to categorise wastes, but without specific regulatory intent. Thus, the upper Action Level is intended to provide a definitive decision point where the waste material under consideration may not be disposed, unless further managed. Screening approaches can be one of the tools used when making these decisions. Upper screening levels (e.g. Level II sediment quality guidelines) are, however, usually intended to indicate greater environmental concern and the requirement for further investigation, but each individual screening level may not have been developed with a specific regulatory intent. Exceeding an upper screening level increases the degree of scrutiny that would be undertaken before a decision is made, but the regulatory authority must also consider many other factors, including the total number of screening factors analysed, the toxicological importance of each screened contaminant, the volume of waste or other matter to be disposed, and other project specific items.

6. Contaminated sediment

6.1 What is contaminated sediment?

As far as the author could establish, no South African legislation provides a definition of contaminated sediment. There also appears to be no universally accepted definition of contaminated sediment. For the purposes of this report the definition of contaminated sediment provided in Section 501(b)(4) of the United States Water Resources Development Act of 1992 will suffice, namely that 'aquatic sediment which contains chemical substances in excess of appropriate geochemical, toxicological or sediment quality criteria or measures; or is otherwise considered by the Administrator [of EPA] to pose a threat to human health or the environment'.

Thus, there are two critical issues pertinent to contaminated sediment, namely that chemical substances are present at higher than 'normal' concentrations and that they pose a risk to human and/or ecological health. As discussed elsewhere in this report, contaminated sediment does not necessarily pose a risk to human and/or ecological health and deciding when it does is often a complicated process.

6.2 The significance of contaminated sediment

Sediment is an important component of coastal aquatic ecosystems since it provides essential habitat for communities of autotrophic and heterotrophic organisms, and bacteria (MacDonald et al. 2003). As such, sediment is essential to the functioning of 'healthy' coastal aquatic ecosystems (Burton 1991). Considering the role and importance of sediment in coastal aquatic ecosystems, it is clear that any form of anthropogenic disturbance to sediment will have an ecosystem level effect. One of the most significant forms of anthropogenic disturbance to sediment is chemical contamination. Coastal waters receive contaminants from local anthropogenic activities and through riverine inputs, amongst other factors. Within these waters the contaminants partition between aqueous (porewater (water between sediment grains) and overlying water) and solid phases (sediment, suspended particulate matter and organisms, Luoma 1983). Sediment is an extremely important sink for many contaminants that are anthropogenically introduced into surface waters. The reason is that many contaminants have low water solubility (i.e. are hydrophobic, or water repulsing) and/or are particle reactive. Once introduced into coastal waters, the contaminants adsorb onto suspended sediment and organic matter and are in this manner 'scavenged' from the water column through flocculation, coagulation and sedimentation (Förstner and Wittman 1979, Olsen et al. 1982, Huh et al. 1992, Honeyman and Santschi 1988, Mwanuzi and De Smedt 1999, Hatje et al. 2003). Sediment solids can hold up to a million times more metal than an equivalent volume of water. It is not surprising then that the concentrations of most contaminants in sediment and at the sediment-water interface usually exceed those in the water column by several orders of magnitude (Horowitz 1991, Daskalakis and O'Connor 1995).

In hydro-dynamically low energy environments, where there is relatively little physical disturbance of sediment (e.g. estuaries, ports and marinas), contaminants can accumulate in sediment to such high concentrations that they cause adverse effects to sediment-dwelling organisms through direct toxicity, or indirectly by altering community and foodweb structures (Chapman 1989, Burton 2002). Even in aquatic systems where the concentrations of chemicals in the water column are below water quality guidelines, contaminants in sediment may still be adversely affecting sediment-dwelling organisms (Chapman 1989, Baker and Kravitz 1992, McCauley et al. 2000). By the same token, many sediment-dwelling organisms inhabit burrows that are

relatively isolated from the surrounding sediment and may thrive even in contaminated sediment provided that overlying water quality is acceptable (Warren et al. 1998, Chapman et al. 1999b).

Sediment-dwelling organisms are in intimate contact with sediment and sediment porewater and, therefore, in intimate contact with contaminants that may be adsorbed onto the sediment particles or dissolved in the porewater. Many benthic invertebrates derive their food by consuming sediment and digesting particulate organic matter between the grains, or by consuming particulate organic matter on sediment. This serves as an important exposure route for these organisms to (particularly organic) contaminants that are partial to adsorption onto organic matter. At elevated concentrations, sediment-associated contaminants have been linked to a number of adverse effects to sediment-dwelling organisms. The testing of field-collected sediment in the laboratory has demonstrated that contaminants may be acutely and chronically toxic to benthic species (e.g. Swartz et al. 1986, 1991, Ferraro et al. 1991, McGee et al. 1999). Alterations to benthic invertebrate communities have been observed in sediment with elevated contaminant concentrations (e.g. Swartz et al. 1985, 1991, Thompson and Lowe 2004). The development of tumours and other abnormalities in benthic-feeding fish has also been attributed to sediment-associated contaminants (Couch and Harshbarger 1985, Malins et al. 1985, Goyette et al. 1988, Vethaak and Wester 1996, USEPA 1998, Jacob 2008).

Many substances that accumulate in sediment can also accumulate in the tissue of fish and shellfish, through dietary transfer. At elevated concentrations, these tissue-borne contaminants may represent significant risks to humans and wildlife that consume aquatic organisms. Fish consumption advisories or total bans on fishing are now a more or less routine feature in many countries, because of the risks that contaminants accumulated in fish and shellfish tissue pose to human consumers. Contaminated sediment also has social and economic implications, including the need to dispose of dredged sediment in expensive containment facilities rather than at unconfined, openwater sites, where there is a risk that the contaminants in the sediment may cause adverse effects. Contaminated sediment can reduce biodiversity and fish stocks and adversely affect the recreational value of a waterbody and human communities that are reliant on fishing for their economic wellbeing. Therefore, accumulation of toxic substances in sediments represents an important concern that needs to be addressed by environmental

managers. Environmental protection efforts in many countries now recognise sediment as a critical component of aquatic ecosystems and require its routine monitoring for environmental quality assessments and for such activities as dredging.

Sediment does not only act as a sink for contaminants, however, but it may also be a significant, long-term source of contaminants to the water column. The release or remobilisation (through desorption) of contaminants occurs when sediment is physically disturbed through natural processes (e.g. tides, storms, bioturbation (due to burrowing formation and movement of animals in an on sediment)) or through anthropogenic activities (e.g. dredging). This has special significance in the context of dredging projects and is hence discussed in more detail below.

7. Environmental effects of dredging

Since the purpose of the revised National Action List is to provide staff from the branch Oceans and Coasts of the Department of Environmental Affairs with a mechanism for reaching informed, transparent and consistent decisions on the suitability of dredged material for unconfined, openwater disposal in South Africa, it is worthwhile providing an overview of some environmental effects of dredging. Although this overview focuses predominantly on the adverse environmental effects of dredging and dredged material disposal, it is important to note that dredging may have environmental, social and/or economic benefits (e.g. Braden et al. 2004). The most obvious benefits are socio-economic, such as improved navigation and port facilities for the trade of cargo, but there are also environmental benefits associated with some dredging projects (e.g. remediation or clean-up dredging - see below).

It is also important to note that although there are adverse environmental effects associated with both the dredging and dredged material disposal processes, the act of dredging itself (and hence associated environmental effects) is not covered by the London Convention and the London Protocol. Nevertheless, a decision on whether dredged material is suitable for unconfined, openwater disposal is made using data generated from the laboratory analysis of sediment samples collected from in-place sediment identified for dredging. These analyses do, therefore, provide a mechanism for assessing the significance of certain adverse environmental effects associated with the dredging process.

Dredging activities can broadly be categorised as maintenance dredging, capital (sometimes called construction) dredging and environmental dredging. Maintenance dredging involves the routine, periodic removal of fine-grained material that accumulates in navigable waterways and ports as a result of natural processes. Maintenance dredging is performed annually in most South African ports. This type of dredging is performed to permit the safe movement of vessels in navigable waterways and ports and is by far the most common type of dredging throughout the world. Capital dredging is usually performed to support new developments, such as quays in ports, new navigation channels, and to increase the depth of already navigable waterways beyond their original design dimensions. This type of dredging is non-repetitive and often involves the dredging and disposal of large volumes of heterogeneous material over a relatively short period of time. A local example is the recent dredging performed in the entrance channel to the Port of Durban, which now has dimensions that permit new-generation Panamax vessels to enter the port. Environmental dredging is performed to create or restore habitats (e.g. nesting islands), and so called clean-up or remediation dredging, which involves the dredging and safe disposal or confinement of contaminated sediment. There are other types of dredging that do not fall into these categories, including dredging for resources (e.g. gravel, diamonds), dredging for shoreline nourishment/protection, and dredging associated with the development of flood control mechanisms. In some cases dredging will fall into two or more categories. For example, maintenance or capital dredged material is often used beneficially (e.g. shoreline nourishment) rather than disposed. There are several options to dispose of the dredged sediment (e.g. beach nourishment or creation of artificial wetlands in the case of clean sediments, or isolation and containment on land in the case of contaminated material), but for economic reasons most dredged material is disposed at unconfined, openwater sites.

The environmental effects of dredging fall into two broad categories, namely effects associated with the dredging process and effects associated with the dredged material disposal process. Although dredged sediment is often disposed in confined containment facilities or used for beneficial purposes, this discussion focuses on the unconfined, openwater disposal of dredged material. The most obvious direct and certain adverse effects associated with the dredging process are the removal of substrate and organisms in or on that substrate, and the re-suspension of sediment and the

associated increase in water column turbidity and suspended solids concentrations. This leads to the loss and/or disturbance of habitat and biodiversity. With the exception of deep-burrowing benthic or highly mobile epibenthic organisms that may survive a dredging event through avoidance, dredging can result in the complete removal of organisms from a dredging site. The benthic and epibenthic community will usually recover after the dredging event. The nature and period of recovery is, however, site-specific and may take from a few months to many years. In situations where there is frequent dredging, complete recovery may never occur.

In environments where the dredged substrate is comprised predominantly of sediment, benthic communities usually recover more quickly compared to areas where the dredged substrate is comprised of stable gravel habitat that is dominated by long-lived components with complex biological interactions controlling community structure. In the latter case, recovery may take many years and possibly decades (Boyd et al. 2003). It is important in this context to note that re-colonisation and recovery are not analogous. While re-colonisation may lead to recovery and may already begin within a few days after a dredging event, recovery refers to the development of a community that is the same or very similar to the community that existed prior to the dredging event. As stated above, recovery may take many years and if there is irreversible habitat change through dredging (e.g. replacement of one substrate type with another) may never occur. Scientists now recognise that the rate of recovery is site specific and there is no overall unifying trend in recovery. Obviously, the loss of a community will alter biodiversity within the dredged ecosystem and affect organisms in higher trophic levels through, for example, the loss of food sources for fish that feed predominantly on benthic organisms.

All methods of dredging lead to the re-suspension of sediment, both during the excavation process and during overflow from hoppers and barges. The extent of sediment re-suspension depends on many factors, including sediment properties (e.g. grain size) and site conditions (e.g. current strength; Palermo et al. 2008). In many cases, resuspension is usually near-field and generally only lasts during and for a short period after the dredging event (Herbich and Brahme 1991, Collins 1995). However, in cases where very fine-grained sediment (so-called powder mud) is dredged the sediment may remain suspended in the water column for many hours after the dredging event and extend over a large area (Herbich and Brahme 1991, Collins 1995,

Pennekamp et al. 1996, Johnson and Parchure 2000, Hayes and Wu 2001). Amongst other impacts, increases in suspended solids concentrations and turbidity reduce light penetration into the water column and may thus influence the photosynthetic ability of flora (water column and submerged forms), clog and damage the feeding apparatus of filter feeding organisms (Essink 1999), and reduce the prey capture ability of line-of-sight predators (e.g. piscivorous fish and birds). Resettling of suspended particulates can also impact bottom-dwelling organisms in habitats near the dredging site. For example, Lohrer et al. (2004) have experimentally shown that layers of sediment as thin as 3 mm can have a detrimental effect on sand flat infauna. The significance of these impacts depends on the nature of the waterbody. They will be more significant in a waterbody that is clear compared to, for example, a river that has a naturally high suspended solids load.

A less obvious, but no less important environmental effect of dredging is the release of contaminants from sediment into the water column. Short-term releases of contaminants from the sediment directly into the water column from dredging operations may be 1 to 3 orders of magnitude greater than pre-dredging releases for the same period of time (Sanchez et al. 2002). As discussed previously, sediment is a significant sink for many contaminants in aquatic ecosystems. Most contaminants partition onto particulate matter, such as clay minerals, iron and manganese oxides/hydroxide coatings on clay particles, and organic substances (Calmano et al. 1993). A proportion of the contaminants will also dissolve in porewater, or in water immediately overlying sediment (the sediment-water interface). However, the proportion of metals in sediment porewater is usually very low. For example, Van den Berg et al. (2001) reported that less than 0.1% of the amount of metal (cadmium, copper and zinc) in sediment from a study site in the Netherlands was in the dissolved phase. Iron and manganese oxides/hydroxides along with organic matter are important binding sites for metals in oxic sediment (Saulnier and Mucci 2000, Li et al. 2000, Zoumis et al. 2001, Fan et al. 2002), while the formation of metal sulphides dominates in anoxic sediments (Di Toro et al. 1990, Zhuang et al. 1994, Caetano et al. 2002). When partitioned to iron and manganese oxides/hydroxides and sulphides, metals are essentially immobilised because of the very low solubility of the metal-oxide/hydroxide and metal-sulphide complexes, and unless the sediment is ingested by an organism are usually not bioavailable and hence unable to exert toxicity.

During dredging, the physical disturbance of the sediment leads to changes in its chemistry. For example, the influx of dissolved oxygen into anoxic sediment leads to an increase in redox potential and a decrease in pH (mainly due to the oxidation of sulphide; Förstner 1989; Reible et al. 2002). These changes in chemistry lead to changes in particle-contaminant complexes which may in turn lead to the remobilisation and transfer of metals to the water column, principally from sulphide-bound complexes (iron sulphide/manganese sulphide; Calmano et al. 1993). The resuspension of anoxic sediment results in variable desorption rates of metals adsorbed to sulphides (Caille et al. 2003). Metals that co-precipitated with or adsorbed onto iron sulphide and manganese sulphide are rapidly oxidised following sediment resuspension, due to their relative solubility in oxic conditions (Allen et al. 1993, Simpson et al. 1998, Caetano et al. 2002). More stable, sulphide-bound metals are less susceptible to oxidation in the short-term due to their slower oxidation kinetics (Caetano et al. 2002). Due to differences in the partiality of metals for different sediment phases, alterations in the chemistry of sediment through physical disturbance may lead to variable releases of different metals. Once released, chemicals can remain in the dissolved (or free) form, which is the most bioavailable and hence most toxic form. However, iron and manganese are rapidly re-precipitated and deposited as insoluble oxides/hydroxides, to which newly released metals can become adsorbed at varying rates and extents (Di Toro et al. 1990, Saulnier and Mucci 2000, Caetano et al. 2002). Thus, dissolved concentrations of metals often peak in the immediate vicinity of a dredging area and decrease sharply away from the area as the metals are re-precipitated or otherwise scavenged from the water column (Goosens and Zwolsman 1996, Saulnier and Mucci 2000). Pieters et al. (2002) reported low contaminant mobilisation during dredging, although metal mobility changed during each dredging technique and was different for every contaminant examined. Van den Berg et al. (2001) and De Groote et al. (1998) also observed low mobilisation of metal contaminants into the dissolved phase during dredging, which they considered to be due to the rapid scavenging of sulphide liberated metals by newly formed iron and manganese oxides/hydroxides.

There are numerous adverse environmental impacts associated with unconfined, openwater disposal of dredged material. The type, extent, magnitude and duration of these impacts depends on the nature of the material that is disposed (e.g. volume, grain size composition, concentrations of contaminants) as well as

the characteristics of the disposal site (e.g. whether it is dispersive or non-dispersive, grain size composition; Bolam et al. 2006). During the disposal process, there is typically an increase in suspended solids concentrations and turbidity in the water column. However, only a small percentage of the material remains in suspension, the majority going directly to the bottom (Peddicord 1987). Whether the increase in suspended solids and turbidity adversely impacts on water column organisms depends on the nature of the material being disposed (e.g. sand or mud) and the dilution factor at the disposal site. During suspension and settling of the sediment, changes in physical and chemical conditions may lead to the desorption of particulate-bound contaminants into the water column, for generally similar reasons as during the dredging process. However, available data suggest that little of the contaminants adsorbed onto sediment are released into the water column during disposal, and peaks in concentrations of contaminants at disposal sites last only a few minutes. Based on the studies of Lee et al. (1977), the turbidity resulting from the hopper disposal of hydraulically dredged sediment is indistinguishable from ambient levels within an hour or two. Based on 35 years of experience with disposal of dredged sediment at sea in New England (USA), Fredette and French (2004) concluded that impacts of organisms via the water column are generally minimal.

At predominantly non-dispersive sites, most of the material will remain on the bottom following placement and may form mounds. At predominantly dispersive sites, in contrast, the material may be dispersed during placement or eroded from the bottom over time and transported off the disposal site by currents and/or wave action. Benthic communities at the disposal site will inevitably be smothered, buried and crushed by the disposed material (Whomersley 2005), irrespective of whether they are of a dispersive or non-dispersive type. If large volumes of sediment are disposed, benthic organisms might not be able to migrate through the material. However, many benthic organisms have capabilities to vertically migrate through substantial overburdens (Maurer et al. 1978, 1986, Essink 1999). If the disposed material has a very different grain size composition to sediment on the disposal site, then this may lead to a shift in the structure and composition of the benthic community due to differences in the grain size preference of benthic organisms (Stronkhorst and van Hattum 2003, Vivan et al. 2009). Maurer et al. (1978, 1986) have shown that overburdens of materials dissimilar to ambient sediments (e.g. mud on sand) have greater impact than deposition of like materials. Lohrer et al. (2004) have experimentally shown that layers of

terrestrial sediments as thin as 3 mm can have detrimental effects on sand flat infauna. These effects are not transient, but may persist for prolonged periods of time (Cummings and Thrush 2004). Clearly, these impacts will also impact on higher trophic levels, such as fish, through the removal or alteration in food resources. Although the recovery of benthic communities at dredged material disposal sites after disposal is often relatively quick (e.g. Newell et al. 1998, Roberts and Forrest 1999, Smith and Rule 2001, Cruz-Motta and Collins 2004), in some cases recovery may take many years (e.g. Newell et al. 1998, Boyd et al. 2003, Sheridan 2004). If dredged sediment has an elevated contaminant load, then the contaminants are obviously transferred to the disposal site should unconfined, openwater disposal of the material be permitted (e.g. Stronkhorst and van Hattum 2003). Maintenance dredged sediment is usually anoxic, and similar to the dredging process, changes in the physicochemical properties of the sediment at the disposal site can result in the release of contaminants (e.g. Francingues et al. 1985). This release may occur over a protracted period as currents erode sediment from the disposal site, exposing contaminant laden deeper layers of the sediment. These contaminants can exert a direct adverse impact on organisms at the disposal site through toxicity, or a secondary toxic effect through bioaccumulation. Erosion of sediment from the disposal site may also lead to a change in the grain size composition of the surroundings, leading to changes in benthic community structure and composition.

As is the situation at the dredging site, benthic and epibenthic communities at dredged material disposal sites will recover after a disposal event. The nature and period until recovery is, however, also site-specific and may take from a few months to many years (e.g. Stronkhorst and van Hattum 2003, Bolam et al. 2006). There are thus a variety of potential effects of the disposal of dredged material in the sea but the effects on the seabed are usually of most significance (Fredette and French 2004, Bolam et al. 2006).

8. Sediment quality guidelines

Sediment quality guidelines comprise a key, and in some cases the sole component of Action Lists that are used by regulatory (permitting) authorities in many countries to reach a decision on whether sediment identified for dredging is of a suitable quality for unconfined, openwater disposal. Indeed, the existing South African National Action List places particularly strong emphasis on the use of sediment quality guidelines for decision-making, to the extent that decisions to date have been made largely on the basis of comparing contaminant

concentrations in sediment identified for dredging to the sediment quality guidelines of the existing National Action List. Before discussing options for the revision of the sediment quality guideline component of the National Action List it is necessary to develop an understanding of sediment quality guidelines, the rationale for their development, approaches for their derivation, and their limitations and appropriate use.

8.1 What are sediment quality guidelines?

The term 'sediment quality guidelines' is generally used to describe numeric chemical concentrations, or in some cases narrative statements, that are intended to be protective of biological resources, predictive of adverse effects to those resources, or both (CCME 2002, Wenning et al. 2002). However, the terminology for these numeric concentrations varies widely and includes sediment quality criteria, sediment action levels, sediment quality standards, sediment quality targets, sediment quality values, sediment quality benchmarks, sediment quality objectives, and sediment trigger values. Although the term sediment quality guideline is used forthwith in this report, Chapman and Wang (1999) have warned against the loose usage of terminology in this context. In particular, they and other workers (e.g. Burton 2002) warn against use of the terms criteria and standards, since these tend to have regulatory implications or connotations. This regulatory implication is counter to the generally accepted usage of sediment quality guidelines, which is as informal screening tools that are best utilised in a weight of evidence approach rather than a deterministic approach. The author is, however, in agreement with USACE (1998) that the term guideline is sufficiently broad and general in its meaning to include all approaches used to derive sediment quality guidelines, and has no regulatory connotation as a 'pass/fail' criterion or standard. Indeed, the term guideline inherently implies that it is a guide rather than a standard for decision-making.

Irrespective of the terminology used and the many approaches that have been followed for the derivation and formulation of sediment quality guidelines, they tend to have a similar objective, namely to identify chemical concentrations that correspond to the occurrence and/or absence of adverse biological effects (USEPA 2000). The underlying supposition in the derivation of effects-based sediment quality guidelines is that the guidelines can be used as a substitute for direct measures of potential adverse effects of contaminants in sediments on sediment-dwelling organisms (Wenning et al. 2002). Certain sediment quality guidelines represent chronic toxicity thresholds, below which adverse effects

to sediment-dwelling organisms are unlikely to occur. Other types of sediment quality guidelines are intended to identify the concentrations of sediment-associated contaminants above which adverse effects are likely to be frequently observed. Still other sediment quality guidelines, although considerably less well developed, are intended to identify concentrations of sediment-associated contaminants that are associated with unacceptable levels of bioaccumulation by wildlife.

8.2. Why develop sediment quality guidelines, and how should they be used for decision-making?

It is widely recognised that the while the concentrations of chemicals measured in sediment can be used to provide an indication of whether the sediment is contaminated, chemical concentrations by themselves cannot be used to determine whether pollution has occurred, that is, whether the contamination has caused an adverse biological effect. All pollutants are contaminants, but not all contaminants are pollutants because substances introduced into the environment may be more or less bioavailable to organisms depending on their chemical form, modifying factors in the environment, the environmental compartment they occupy, and the reactions (behavioural and physiological) of exposed organisms (Chapman et al. 2003).

In order to fully understand whether elevated chemical concentrations are causing adverse biological effects, biological assessment is required (Chapman et al. 1999). This may include, for example, toxicity testing and assessment of benthic invertebrate community composition. These studies are, however, often lengthy, complex and expensive, and are not always definitive. The derivation of sediment quality guidelines arose, in part, from a desire by managers and decision-makers to have a simple, consistent tool for discriminating between contaminated sediments that are of little concern and those predicted to have adverse biological effects, based only on the measurement of chemical concentrations in the sediment. Not only would this ensure that decisions on sediment quality and future actions required are made in a consistent, transparent and equitable manner, but would eliminate the need for an expensive suite of biological analyses to determine whether a particular site is or is not of concern. In essence, sediment quality guidelines were conceived as a shortcut to decision-making.

Sediment quality guidelines are advantageous in that they are easy to use by both specialists and non-

specialists, simplify decision-making, and provide a consistent benchmark for assessing sediment quality. Amongst other uses, sediment quality guidelines are now widely used as a tool to interpret sediment chemistry, to identify chemicals of concern, to rank and prioritise aquatic systems for further attention, and to assess the suitability of dredged material for unconfined, openwater disposal (Long and MacDonald 1998, Macdonald 1994, Wenning et al. 2002). Many workers consider sediment quality guidelines to be a useful tool for assessing sediment quality when used in combination with other assessment tools in a weight of evidence approach (MacDonald et al. 1992, Ingersoll et al. 1996, 1997, USEPA 1997, Long and Macdonald 1998, MacDonald et al. 2000).

However, many scientists have cautioned against and indeed criticised the use of sediment quality guidelines for assessing sediment quality, particularly as a standalone tool (e.g. O'Connor 2004, Jones-Lee and Lee 2005, Wetherington et al. 2005). They contend that sediment quality guidelines do not provide a sufficient basis for determining the potential for adverse biological effects. The reason is that despite the science underpinning the derivation of sediment quality guidelines being sound, no sediment quality guidelines have been shown to consistently and reliably predict toxicity (Wenning et al. 2002). Elevated contaminant concentrations do not always cause adverse biological effects, although adverse biological effects are usually associated with elevated contaminant concentrations. The reason that adverse effects are not always observed is that numerous physical and chemical characteristics of sediment control the bioavailability of chemicals, and hence their toxicity. Furthermore, aquatic organisms have evolved strategies to both conserve and regulate essential metals in sediments in certain elevated concentration ranges without adverse biological effects (Chapman et al. 1999). Sediment quality guidelines are also not applicable to mixtures of chemicals, or to chemicals that bioaccumulate (Chapman et al 1999).

Sediment quality guidelines primarily serve to identify hazard that is, the potential or not for risk. To actually determine whether that hazard may be realised, that is, whether there is a risk and the magnitude of such a risk, requires further assessment such as an ecological risk assessment, incorporating appropriate consideration of site- and situation-specific conditions (i.e. local chemical, physical, and biological conditions). Sediment quality guidelines are thus applicable in the first step of an environmental risk assessment, namely the problem formulation process (Chapman et al. 1999). Subsequent steps involve the use of other tools (e.g. toxicity testing,

benthic invertebrate community analyses, tissue burden analyses) in a weight of evidence approach to fully assess sediment quality. Ultimately, the ecological risk assessment output defines the probability of risk occurring or having occurred. Since sediment quality guidelines may vary in their incorporation of chronic toxicity information, they may lack the ability to predict or protect against long-term effects, including bioaccumulation (Chapman et al. 1999, Chapman 2007). Similarly, the current biological receptors-based sediment quality guidelines cannot be used directly for human health risk assessment.

If sediment quality guidelines are to be useful as decision tools, they should meet a number of criteria: 1) They should be sufficiently protective to minimise the probability of false negatives (type II error, i.e. they should not allow toxic sediments to pass), 2) they should be designed to minimise the probability of false positives (type I error, i.e. they should not fail too many nontoxic sediments), and 3) they should be meaningful for the sediments and toxicants to which they will be applied. Because of the broad range of potential toxicities of single contaminants and mixtures of contaminants in various sediment matrices and to different organisms, no single sediment quality guidelines can meet both criteria 1 and 2. For this reason, most international decision-making frameworks adopt a tiered approach based on multiple lines of evidence. Sediment quality guidelines are most appropriately used in the lowest or second lowest tier, in order to determine whether there is a need to enter further tiers. In some programmes, these lines of evidence are collected together; in other programs, failure of one test (tier) leads to subsequent tests. Any management framework that relies on just sediment quality guidelines to inform decisions must choose between a preference for type I and type II errors. Too many false positives in a decision framework can result in excessive costs (i.e. sediments requiring control, remediation, or management not truly necessary on a risk basis), although too many false negatives may result in insufficient protection of human health and the environment.

8.3 Sediment quality guidelines and uncertainty

Although sediment quality guidelines are widely used and in some cases viewed as a panacea for decision-making, as mentioned above it is necessary to recognise that sediment quality guidelines have limitations. The advantages and disadvantages of different approaches for deriving sediment quality guidelines and the implications for their use are discussed in a subsequent

section of this report. It is, however, necessary at this early stage to address the issue of uncertainty in the context of sediment quality guidelines. This was briefly alluded to above in the context of false positives and false negatives.

In a perfect world, sediment quality guidelines would be able to unequivocally discriminate between the concentrations of chemicals in sediment that cause adverse biological effects and those that do not. In other words, all sediments with chemicals at concentrations below a certain level would show no adverse effects, while all those with chemicals at concentrations above this level would show adverse effects. Furthermore, the magnitude of the adverse effects would increase as chemical concentrations increased until a threshold was reached, above which all organisms would be die through toxic effects (Figure 1a). Although this is borne out in some cases (e.g. MacDonald et al. 2000), the occurrence of adverse biological effects often does not show a clearly delineated relationship because there are a multitude of poorly understood process that influence the bioavailability and hence toxicity of chemicals in sediment (USACE 1998, O'Connor 2004, Batley et al. 2005, Wenning et al. 2002). Instead, the distribution of biological effects generally shows a relationship characterised by ranges of chemical concentrations where biological effects are rare, where cases of both effects and no effects are found (transition zone), and where biological effects essentially always occur (Figure 1b). Consequently, none of the sediment quality guidelines consistently and reliably predicts toxicity. Factors that cause overlap between effect and no-effect data are many, but include contributions of other chemicals to effects, unaccounted for differences in chemical availability, differences in response among organisms, and errors in measurement of either chemical or response (Wenning et al. 2002).

This generalised concentration-response model, with the probability of effects increasing with increasing chemical concentration can be used as a framework to consider different sediment quality guideline approaches (Figure 1c, Batley et al. 2005). The probability of effects is low until it reaches a threshold effect guideline. At the high end of the distribution is the probable effect guideline, above which effects almost always occur. Between threshold effect and probable effect lies a transition zone where adverse biological effects may or may not occur, but within which the probability of adverse effects increases as the concentration of the chemical in sediment increases. Sediment quality guidelines can be thought of as vertical lines across this concentration-response curve. Different sediment quality guideline

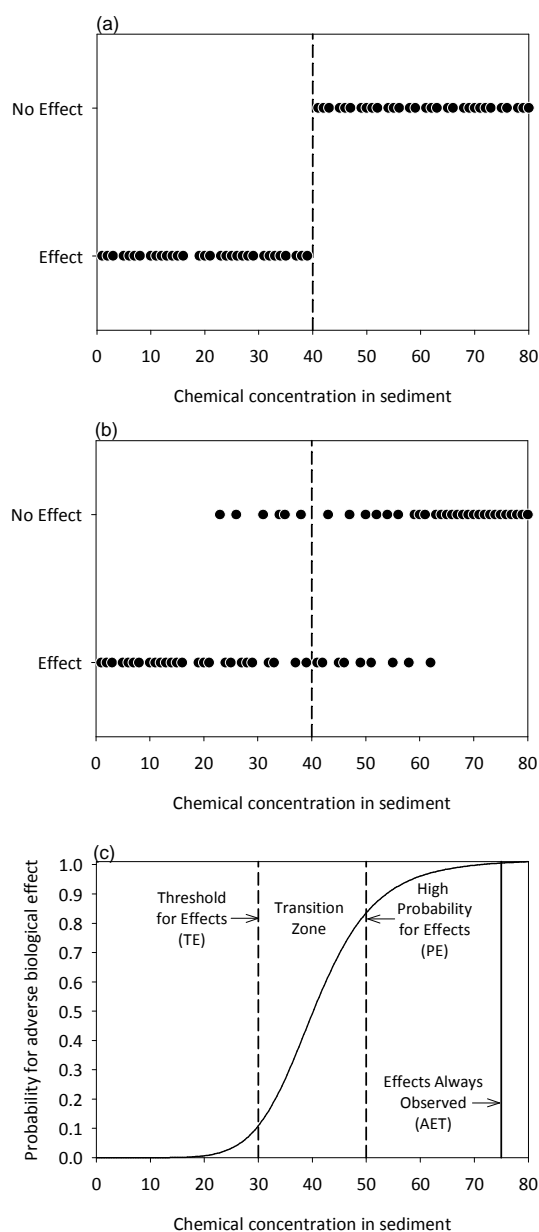


Figure 1. Examples of distributions of biological response data over a range of sediment contaminant concentrations. (a) Ideal case where there is no overlap between concentration ranges that do and do not show biological effects. (b) Typical actual distribution with overlap between these concentration ranges. (c) A generalised concentration-response model for contaminated sediments (from Batley et al. 2005).

derivation procedures are intended to represent threshold effect, probable effect, or mid-range effect guidelines. A threshold effect guideline is a contaminant concentration below which adverse biological effects are predicted to occur infrequently, whereas a probable effect guideline is intended to identify the contaminant concentration above which effects are predicted to frequently or always occur depending on the guidelines degree of conservativeness. Intermediate to these would be mid-range guidelines, which identify contaminant concentrations within the transition zone and wherein

the probability of effects is substantively above background, but above which effects are not always observed.

In this context, chemically based numeric sediment quality guidelines can be effective for identifying concentration ranges where adverse biological effects are unlikely, uncertain, and highly likely to occur. By definition, the existence of the transition zone in Figure 1c means that no guideline can unequivocally separate all sediments showing effects from those that do not. Nonetheless, sediment quality guidelines are often employed such that they are expected to separate 'toxic' and 'nontoxic' samples. In such applications, the interpretation of exceeding a guideline varies according to where the guideline lies along the concentration-response continuum. If one uses a threshold effect type sediment quality guideline to define 'toxic' (or unacceptable), then errors will tend to be characterized as false positives; that is, some sediments will be classified as toxic when in fact they are not. On the other hand, selecting a probable effect type guideline will result in predominately false negative errors, in which some sediments will be classified as nontoxic when in fact they are toxic. As one moves from a threshold effect guideline toward a probable effect guideline, the chances of false positives decrease and chances of false negatives increase. In some situations, it may be desirable to have a guideline that will identify almost all potentially toxic samples, even if many of those samples are not actually toxic (threshold effect guideline). In other instances, it may be desirable to have a guideline that predominately identifies sediments that have a high probability of being toxic, even if means that some sediments that may be toxic will be missed.

9. Approaches for deriving sediment quality guidelines

The approaches that have been used to derive sediment quality guidelines fall into two broad categories, namely theoretical (including mechanistic) and empirical (sometimes called correlative) approaches (Wenning et al. 2005). Mechanistic guidelines are derived from a theoretical understanding of the factors that govern the bioavailability of contaminants in sediment, and known relationships between chemical exposure or uptake and toxicity. Thus, mechanistic approaches provide at least a theoretical basis for assuming some cause and effect relationship between a contaminant of interest and a biological response. Equilibrium partitioning theory provides the basis for all existing mechanistic guidelines. Mechanistic guidelines account for bioavailability through normalisation to sediment characteristics that

affect bioavailability, primarily organic carbon for non-ionic organic chemicals, and simultaneously extracted metal-acid volatile sulphide, organic carbon, or other sediment fractions for metals (Wenning and Ingersoll 2005).

While mechanistic approaches are useful for describing potential contaminant availability and identifying the cause of toxicity, this approach cannot be used to derive sediment quality guidelines for many contaminants of interest. The more widely used type of sediment quality guidelines are those that are empirically derived. These guidelines are derived through the statistical analysis of large data bases of matching sediment chemistry and observed biological effects data (e.g. data from sediment toxicity tests and benthic invertebrate community composition). The data are arrayed on a continuum of increasing chemical concentration, and various statistical approaches are then used to define chemical concentrations that are associated with particular levels of effect or no effect (Wenning and Ingersoll 2002).

In addition to mechanistically-based and empirically-derived sediment quality guidelines, a third group of 'consensus-based' guidelines can also be considered. These guidelines are an evolution of empirically-derived guidelines, as they are formulated by taking the geometric mean of published sediment quality guidelines with a similar narrative intent. This reconciles sediment quality guidelines that have been derived using various approaches (MacDonald et al. 2000).

9.1 Theoretical approaches

9.1.1 Sediment Background approach

Background or baseline concentrations of chemicals in sediment are most commonly used for identifying whether sediment from an area of interest is contaminated. However, sediment quality guidelines in several jurisdictions/countries have been formulated using background or baseline concentrations. The sediment quality guidelines are formulated as a proportional exceedance of the background or baseline concentration that is deemed acceptable (e.g. 200% exceedance). An implicit assumption of this approach is that chemicals present in sediment at or above the background or baseline concentration will have an adverse effect on sediment-associated organisms, that is, the entire 'excess' concentration is in a bioavailable form.

Although the Sediment Background approach has most often been used to derive sediment quality guidelines for metals, the approach is suitable for deriving

guidelines for other chemicals. Both naturally occurring and synthetic organic compounds are present in marine sediment as a result of marine biological processes and geochemical transport from diverse sources or anthropogenic activities. Although the concentrations of organic chemicals that only have an anthropogenic origin should, in theory, be zero, the reality is that there is probably no place on earth that has not been contaminated. For example, numerous organic contaminants that only have an anthropogenic origin (e.g. polychlorinated biphenyls and organochlorine pesticides) have been detected in sediment and biological tissue from the Polar Regions, far from anthropogenic sources of these contaminants. Atmospheric transport is considered to an important vector for the transport of these contaminants to Polar Regions. Some organic chemicals that are important contaminants of surface waters due to anthropogenic releases also have natural sources. Polycyclic aromatic hydrocarbons are the best example. Polycyclic aromatic hydrocarbons are significant contaminants of sediment from urbanised and industrialised areas, where they are released from numerous sources. The incomplete combustion of fossil fuels in vehicles is amongst the most important sources, but they are also formed during forest fires for example. For these chemicals, their concentrations can be measured in areas where anthropogenic impacts are assumed to be minimal and some proportion of their distribution (e.g. 75th percentile) used to define the baseline concentration. Alternately, concentrations measured in pre-industrial sediment extracted from core samples can be used to define background concentrations.

Background/baseline concentrations have been used in one or another way to derive sediment quality guidelines for metals and in some cases also organic chemicals for the regulation of dredged sediment disposal in numerous countries, including Denmark, Finland, Germany, Spain and Sweden (see later for further information). In many jurisdictions/countries, the definition of background or baseline concentrations is strongly encouraged and comprises an important component of risk assessment frameworks, since it has been shown that sediment quality guidelines are often lower than background or baseline concentrations (e.g. Fletcher et al. 2008). The background/baseline concentrations thus comprise an important for screening sediment.

Advantages:

The major advantages of the Sediment Background approach are its simplicity and that while the data

requirements and costs of this approach may be significant, these are nevertheless well below the requirements of empirical approaches for deriving sediment quality guidelines. Background and baseline concentrations tend to be site specific and may need to be developed for each region of interest. However, once background or baseline concentrations have been defined these do not need to be redefined, but can simply be built upon as and when new data becomes available for the area of interest. This is different to empirical approaches, which become increasingly defensible as the database increases in size. The concept of background or baseline concentrations is easily understood and can be used by both specialists and non-specialists for assessment purposes. The approach also does not require quantitative toxicological data and thus avoids the need to seek explanations for contaminant behaviour or biological effects (Persaud et al. 1993, MacDonald 1994, Fletcher et al. 2008).

Limitations:

Since the Sediment Background approach is based exclusively on the concentrations of chemicals in sediment, sediment quality guidelines that are derived from these concentrations have no biological basis. Thus, guidelines based on a proportional exceedance of the background or baseline concentration are defined subjectively (i.e. what is considered acceptable or unacceptable) and are indefensible from a toxicological perspective. The implicit assumption that concentrations of chemicals that are in 'excess' of the background/baseline concentration are able to exert adverse effects assumes that the 'excess' concentrations are in a bioavailable form. This is often not true, however, since bioavailability is modified by numerous, poorly understood physical and chemical characteristics of sediment (Persaud et al. 1993).

Sediment characteristics, such as grain size and organic content, have been shown to be major factors that influence the composition and structure of benthic communities, but these are not taken into account by this approach. Rather, the assumption is that the chemicals alone are responsible for observed effects (i.e. differences; Persaud et al. 1993). Background/baseline concentrations tend to be highly site specific and should only be applied to the area for which they were developed.

9.1.2 Equilibrium Partitioning approach

The Equilibrium Partitioning approach has been used to formulate sediment quality guidelines for non-polar, hydrophobic organic chemicals, including organochlorine

pesticides and polycyclic aromatic hydrocarbons (Di Toro et al. 1990). The approach is based on the premise that the toxicity of sediment-associated non-polar organic chemicals (and divalent metals, see below) is dictated by the amount of the chemical that is un-complexed, or freely available, in interstitial (pore) water. When a non-polar organic contaminant enters sediment it will partition between the sediment and porewater in three compartments: a fraction will adsorb to the organic carbon in the sediment, another fraction will adsorb to dissolved organic carbon in the interstitial porewater, and a third fraction will dissolve in the porewater. An equilibrium will be established so that any change in the chemical concentration in one compartment will result in a corresponding change in concentration in the other compartments. Equilibrium partitioning methodology contends that sediment toxicity is attributable to the concentration of a chemical that is dissolved in the porewater and which is hence bioavailable (USEPA 1992). The assumption is, therefore, that porewater is the primary route of organism exposure. It can be inferred, then, that a water quality guideline developed to protect aquatic life from contaminants dissolved in the water column should also protect sediment-associated organisms from contaminant concentrations dissolved in porewater. For interpretive purposes then, the concentration of the contaminant that is predicted in the porewater, as calculated from bulk sediment chemistry measures, is compared to water quality guidelines.

The USEPA (2003, 2005), the New York State Department of Environmental Conservation (NYSDEC 1999) and the Netherlands (Crommentuijn et al. 2000 a,b) have developed sediment quality guidelines using the Equilibrium Partitioning approach.

Advantages:

The Equilibrium Partitioning approach does not require the collection of biological data, since it is based solely on sediment chemistry (USEPA 1992). The approach begins to address cause and effect, because the guidelines are based not on correlation but on strong theoretical foundations (Chapman and Mann 1999). The approach also relies on an existing toxicological rationale, which has been established during the development of the water quality criterion being used (Persaud et al. 1993).

Limitations:

The Equilibrium Partitioning approach assumes that the only route of exposure of sediment-dwelling organisms to contaminants is via porewater, ignoring important

sources of exposure through sediment ingestion and food. It has not yet been proven that the porewater is the only significant route of exposure (Persaud et al. 1993, Fletcher et al. 2008). In fact, several ionic contaminants are so hydrophobic that it is improbable that exposure through porewater is significant (O'Connor and Paul 2000). Equilibrium conditions, on which both the Equilibrium Partitioning approach depends, rarely occur in many environments. Port, estuary and nearshore marine systems are typically dynamic, with continuous sediment bioturbation and deposition/resuspension due to tides, wind and navigation traffic amongst other sources of disturbance. Therefore, an essential condition for the validity of the Equilibrium Partitioning approach does not exist, because equilibrium conditions do not occur (USACE 1998, O'Connor and Paul 2000). While the Equilibrium Partitioning approach is useful for describing potential contaminant availability and identifying the cause of toxicity, guidelines cannot be defined for many contaminants that are typically of interest in sediment quality assessment programmes (Vidal and Bay 2005).

9.1.3 Acid Volatile Sulphide approach

The Acid Volatile Sulphide approach is also based on equilibrium partitioning theory, but differs from the Equilibrium Partitioning approach in that it addresses the partitioning of ionic (divalent) metals between sulphides and water rather than partitioning of non-ionic organics between organic carbon and water. The Acid Volatile Sulphide approach assumes that divalent trace metals (cadmium, copper, lead, nickel and zinc) react with sulphide in sediment, specifically the sulphide fraction that is soluble in cold acid, known as acid volatile sulphide. Acid Volatile Sulphide is a reactive pool of solid-phase sulphide (predominantly iron sulphides) that is available to bind metals and render that portion relatively non-available and hence non-toxic to organisms (Di Toro et al. 1990, 1992, Ankley et al. 1996). The amount of Acid Volatile Sulphide present in sediment is thus a critical factor influencing the bioavailability of divalent metals. In the laboratory, metals that are extracted at the same time as sulphide are referred to as Simultaneously Extracted Metals. If the molar fraction of Acid Volatile Sulphide is present in sediment in excess of the reactive forms of the metals (i.e. Acid Volatile Sulphide > Simultaneously Extracted Metals), then the concentration of free (bioavailable) metals in porewater is expected to be low and unable to cause toxicity. In contrast, if Simultaneously Extracted Metals > Acid Volatile Sulphide, the excess fraction of the metals may potentially exist as free metal and could cause toxicity.

The USEPA (2003, 2005), the New York State Department of Environmental Conservation (NYSDEC 1999) and the Netherlands (Crommentuijn et al. 2000 a,b) have developed sediment quality guidelines using the Acid Volatile Sulphide approach.

Advantages:

As with the Equilibrium Partitioning approach, the Acid Volatile Sulphide approach does not require the collection of biological data since it is based solely on sediment chemistry, and begins to address cause and effect because the guidelines are based not on correlation but on strong theoretical foundations (Chapman and Mann 1999). The approach works comparatively as well as other approaches based on bulk sediment chemistry for correctly predicting trace metal contaminated sediment as toxic or nontoxic (Long et al. 1998).

Limitations:

Many of the same limitations that apply to the Equilibrium Partitioning approach also apply to the Acid Volatile Sulphide approach, since they are both based on equilibrium partitioning theory. Thus, the Acid Volatile Sulphide approach also assumes that the only route of exposure of organisms to contaminants is in porewater, ignoring important sources of exposure through sediment ingestion and food (Persaud et al. 1993). As stated above, equilibrium conditions upon which equilibrium partitioning approaches depend rarely occur in many environments. Only five metals (Cd, Cu, Pb, Ni, and Zn) can be evaluated using the Acid Volatile Sulphide approach (Jones et al. 1997). Acid volatile sulphide cannot exist in the presence of oxygen, and aquatic macrofaunal organisms can exist only in the presence of oxygen. Even though microscale changes in sediment oxidation are common in fine-grained sediment (thin-surface layers, lining of burrows, etc.), animals live only in the presence of oxygen and, therefore, in the absence of acid volatile sulphide. Lastly, acid volatile sulphide is not the only metal binding phase in sediments.

9.2 Empirical approaches

9.2.1 Apparent Effects Threshold approach

The Apparent Effects Threshold approach attempts to define the concentrations of contaminants in sediment above which statistically significant adverse effects for a specific endpoint (e.g. mortality) are always expected (Barrick et al. 1988). This approach was originally derived to help assess significantly contaminated sediments in largely urban settings (Chapman and Mann 1999). The adverse biological effects that have been considered in

this approach include toxicity to benthic and/or water column species (as measured using toxicity tests) and changes in benthic infaunal community structure. In Puget Sound, four marine Apparent Effects Threshold values have been generated, based on whole sediment toxicity tests using the amphipod *Rhepoxynius abronius*, toxicity tests on sediment elutriates using oyster larvae and Microtox® solid phase test, and biological effects in the field as measured through by the abundance of major taxa of benthic infauna (Batley et al. 2005). Inherent in this approach is the assumption that adverse biological effects above the level of contamination specified by the Apparent Effects Threshold are specifically related to the contaminant of interest, while below this level any effects observed could be due to other contaminants (Persaud et al. 1993, USACE 1998).

The state of Washington in the USA has used the Apparent Effects Threshold approach to establish sediment quality standards and minimum clean-up levels for contaminants of concern (WSDEC 1995). Ingersoll et al. (1996) utilised a similar approach to develop freshwater Apparent Effects Thresholds using the results of toxicity tests and chemical analyses conducted on sediments from various freshwater locations in the USA.

Advantages:

Because observed biological effects always occur above the Apparent Effects Threshold, this approach provides values based on non-contradictory evidence of biological impacts. The approach requires no assumptions regarding contaminant bioavailability (Persaud et al. 1993). Therefore, the effects on organisms can be due to exposure to contaminants through adsorption from sediments, interstitial water and ingested matter (Chapman 1989, Fletcher et al. 2008). The approach is considered sensitive and efficient, although the amount of data available has a marked effect on Apparent Effects Threshold uncertainty (Chapman 1989, USEPA 1992).

Limitations:

While inherently assuming a cause and effect relationship between a chemical concentration and a biological effect, the Apparent Effects Threshold approach cannot clearly demonstrate a cause and effect relationship for any single contaminant. This is because the biological effect is attributed to all chemicals measured in the sediment sample. Thus, there is no evidence that a contaminant that exceeds its sediment quality guideline will be associated with an effect in any sediment except the one that was used to derive the sediment quality guideline. The approach is thus site specific and should be used with caution in areas outside

that for which they were developed (MacDonald 1994). Furthermore, the approach assumes that the entire concentration of each chemical measured in the sediment sample is bioavailable. This is usually not the situation, especially for metals. A chemical may have been present in a non-bioavailable form in the sediment and had nothing to do with the biological effect, yet the effect was still attributed to the chemical. Thus, while definite ecotoxicological effects can be established, these cannot be attributed to any single contaminant (O'Connor et al. 1998, Crane et al. 2000, Jones-Lee and Lee 2005, Fletcher et al. 2008). Because the Apparent Effects Threshold is the sediment chemical concentration above which statistically significant biological effects always occur, they may be under-protective because biological effects have been observed at chemical concentrations below Apparent Effect Threshold values, but not 'always' (Persaud et al. 1993, Chapman 1999). If the data used consist of mixed species and endpoints, the least sensitive of these species and endpoints will always predominate and the guidelines may not be protective of other, more sensitive species (Persaud et al. 1993).

9.2.2 Screening Level Concentration approach

The Screening Level Concentration approach attempts to estimate the highest concentration of a contaminant that can be tolerated by a predefined proportion of benthic infaunal species (Neff et al. 1986). The Screening Level Concentration is determined from a database of the concentrations of contaminants in sediment and the co-occurrence of benthic organisms. For each organism for which adequate data are available, a species Screening Level Concentration is calculated by plotting the frequency distribution of the contaminant concentrations over all of the sites at which the species occurs. The 90th percentile of this distribution is taken as the Screening Level Concentration for the species investigated. The Screening Level Concentrations for all species are then compiled as a frequency distribution to determine the concentration that can be tolerated by a pre-defined proportion of the species. This concentration is then the Screening Level Concentration for the pre-defined proportion of species (Neff et al. 1986, Persaud et al. 1993). Originally, Neff et al. (1986) defined Screening Level Concentrations that were protective of 95% of species, but other workers have defined Screening Level Concentrations that are protective of fewer species (e.g. EC and MENVIQ 1992).

The Screening Level Concentration approach has been used to derive sediment quality guidelines for application in the Great Lakes (USA/Canada; Neff et al.

1986) and for the St. Lawrence River in Canada (EC and MENVIQ 1992, Fletcher and Welsh 2008). The approach for deriving sediment quality guidelines in Canada has changed (see CCME 2002) and guidelines derived using the Screening Level Concentration approach are no longer used for sediment quality assessment purposes.

Advantages:

Since the Screening Level Concentration approach does not make any assumptions about the absence of a species and considers only those species present, the approach does not require *a priori* assumptions concerning cause and effect relationships between sediment contaminant concentrations and the presence or absence of benthic species. As no relationship is assumed it is not necessary to take into account the wide variety of environmental factors that affect benthic communities, such as substrate type, temperature and depth. However, valid inferences can be drawn from this type of analysis regarding the range of sediment contaminant concentrations that can be tolerated by the infauna since field data on the co-occurrence of benthic infaunal species and sediment contaminant concentrations are used (Chapman 1989, Fletcher et al. 2008). Another advantage is that Screening Level Concentrations can be ranked to determine protective concentrations for any proportion of the species (McCauley et al. 2000).

Limitations:

As with other co-occurrence guidelines, although cause and effect is inherently assumed the Screening Level Concentration approach cannot clearly demonstrate a cause and effect relationship for any single contaminant (O'Connor et al. 1998, Crane et al. 2000, Jones-Lee and Lee 2005). As a result, the Screening Level Concentration for a particular contaminant will tend to be conservative (Chapman 1989, MacDonald 1994, Jones-Lee and Lee 2005). The approach assumes that the distribution of benthic organisms is related primarily to the levels of the contaminant measured in the sediments. The effects of other factors, including unmeasured contaminants, habitat composition (i.e. grain size), and interspecific interactions are not explicitly considered, but these are known to be important determinants of benthic invertebrate community composition (MacDonald 1994, Jones-Lee and Lee 2005). The approach is susceptible to skewed data, that is to either lightly or heavily contaminated areas. Therefore, care must be taken to include data taken over the full range of conditions since a database skewed to either lightly or heavily contaminated areas will yield guidelines that are either too conservative or do not provide adequate protection

(Persaud et al. 1993). Because it utilises benthic infaunal data only, this approach does not provide a weight of evidence for assessing sediment quality (MacDonald et al. 2003).

9.2.3 Spiked Sediment Toxicity Test approach

In the Spiked Sediment Toxicity Test approach, dose-response relationships are determined by exposing test organisms under controlled laboratory conditions to sediments that have been spiked with known amounts of contaminants (either individually or in mixtures). Thus, definitive cause and effect relationships between chemical concentrations and biological responses are established (e.g. mortality, reductions in growth; Persaud et al. 1993, MacDonald 1994). Sediment quality guidelines can then be determined using the sediment bioassay data in a manner similar to that in which aqueous bioassays are used to establish water quality criteria, although the sediment approach is considerably more complex since variables such as the grain size composition and total organic content of the sediment need to be considered. Where the concentration of contaminant-absorbing phases present in the sediment is high, as for example in sediment comprised predominantly of mud, the equilibrium pore water concentration of the contaminant spike will be lower than in the absence of such phases. Thus, the guidelines will be sediment-specific unless a normalising procedure is used.

The Spiked Sediment Toxicity Test approach has been adopted as an approach for deriving sediment quality guidelines in Canada, although there is at present insufficient data available at this time for its use. Data from spiked sediment toxicity tests have also been incorporated into Effects Range guidelines derived by Long et al. (1995; see further below).

Advantages:

The major advantage is that a direct cause and effect relationship is established (O'Connor et al. 1998) with a high level of accuracy (USEPA 1992). The approach is suitable for all classes of chemicals and most types of sediments (MacDonald 1994).

Limitations:

Spiked-sediment bioassays have only been conducted on a few species of aquatic organisms and with only a limited number of substances (MacDonald 1994). Therefore, existing databases support the derivation of numerical sediment quality guidelines for only a few contaminants. Toxicity tests performed under controlled conditions in the laboratory may not be directly

applicable to field situations where conditions may vary considerably from those encountered in the laboratory. In order to derive realistic guidelines from this approach, different sediments with various chemical mixtures in differing proportions and using different organisms, as would exist in field situations is required. This requires considerable effort and expense (USEPA 1992). Because sediments vary widely in their geochemical makeup from region to region, the application of spiked sediment results may not be as transferable from location to location as is true for spiked water results and this variability needs to be considered when establishing sediment quality guidelines using this approach.

9.2.4 Effects Range approach

The Effects Range approach was originally developed by the National Oceanic and Atmospheric Association (NOAA) for correlating sediment chemical concentrations with biological responses as part of the National Status and Trends Program in the USA (Long and Morgan 1991). The sediment quality guidelines derived by Long and Morgan (1991) were subsequently updated by Long et al. (1995), by removing freshwater data that were originally included and adding more estuarine and marine data. These sediment quality guidelines have become perhaps the best known and most frequently applied in the world. Thus, workers from within and beyond North America have used the guidelines for assessing sediment quality (e.g. Torres et al. 2009). The sediment quality guidelines have also been adopted, with minor modifications to reflect local conditions, by the Hong Kong administrative area of China (ETWB 2002), and by Australia (Commonwealth of Australia 2009) and New Zealand (Maritime Safety Authority of New Zealand 1999).

The sediment quality guidelines were derived by Long and Morgan (1991) from an extensive database of chemical and co-occurring biological effects data for both freshwater and saltwater habitats in North America. All information in the database was weighted equally, regardless of the method that was used to develop it. Concentrations that were not associated with an adverse (toxic) effect were assumed to represent background conditions and thus excluded from further consideration. Two guidelines were derived, namely the Effects Range Low (lower 10th percentile of the 'effects' concentrations), below which adverse effects on sensitive life stages and/or species of sediment-associated organisms occurred infrequently, and the Effects Range Median (50th percentile of the 'effects' concentrations, that is, the median value), above which adverse effects were frequently observed (Long and

Morgan 1991). As stated previously, Long et al. (1995) refined the guidelines but did not change the basic approach, only excluding data for freshwater systems and data of marginal quality used in earlier and including a considerable amount of higher quality estuarine and marine data.

The Effects Range Low and Effects Range Median are informal screening tools, and are best applied when accompanied by measures of effects such as laboratory toxicity tests, benthic community analyses and bioaccumulation tests, which lead to the preparation of a weight-of-evidence (Long et al. 1995, NOAA 1999). By considering matching sediment chemistry and biological effects data from studies conducted in the field, the influence of mixtures of chemicals in sediments is incorporated into the sediment quality guidelines. This increases the degree of environmental realism and, thus, applicability of the guidelines. Long et al. (1998) also introduced Effects Range Median quotients, a calculation that emphasises sediments that have many Effects Range Median exceedances or a few extreme exceedances. The approach involves dividing the concentrations of each chemical by its Effects Range Median, adding up all the ratios, and then dividing by the number of ratios. If the resulting Effects Range Median Quotient for a sediment sample is greater than one the sediment is considered to have a greater chance of being toxic than if there were simply just an Effects Range Median exceedance (O'Connor and Paul 2000). The major limitation of this approach is that it needs to be optimised at a site specific level.

The Effects Range approach has been used to derive sediment quality guidelines for numerous chemicals in sediment from freshwater and marine habitats in the USA (Long et al. 1991, 1995, Ingersoll et al. 1996). MacDonald (1997) used this approach to derive Sediment Effect Concentrations for PCBs and DDT in the Southern California Bight.

Advantages:

Data used to derive the sediment quality guidelines is highly relevant to the guideline derivation process because it applies to a wide range of biological organisms and endpoints, incorporating a large number of direct measurements on organisms that are normally associated with sediments. In addition, the data is representative of a wide range of sediment conditions and incorporates mixtures of chemicals. These attributes are likely to give the sediment quality guidelines broad applicability, increasing the probability that they will be appropriate in different areas (Crane et al. 2000).

Limitations:

As with other co-occurrence guidelines, although cause and effect is inherently assumed between a chemical concentration and a biological effect, the approach cannot clearly demonstrate a cause and effect relationship for any single contaminant (O'Connor et al. 1998, Crane et al. 2000, Jones-Lee and Lee 2005). Although the approach incorporates chemical mixtures, the approach is unable to separate biological effects that may be due to a combination of contaminants (Chapman 1989, Jones-Lee and Lee 2005). The sediment quality guidelines were neither calculated nor intended as toxicological thresholds, but rather as informal screening tools. Therefore, there is no certainty that they will always correctly predict either non-toxicity or toxicity (NOAA 1999).

9.2.5 Effects Level approach

The Effects Level approach is fundamentally similar to the Effects Range approach. Here, however, both 'effect' and 'no effect' data are used to derive sediment quality guidelines. For each chemical, a Threshold Effects Level is derived as the geometric mean of the 15th percentile of the effects data set and the 50th percentile of the no effects data set. A Probable Effects Level is derived by determining the geometric mean of the 50th percentile of the effects data set and the 85th percentile of the no effects data set. The Threshold Effects Level is an estimate of the concentration of a chemical below which adverse effects only rarely occur, and the Probable Effects Level an estimate of the concentration above which adverse effects frequently occur (MacDonald et al. 1996). Essentially, the Threshold Effects Level corresponds to the Effects Range Low and the Probable Effects Level to the Effects Range Median of the sediment quality guidelines derived by Long et al. (1995).

The Effects Level approach has been used to derive sediment quality guidelines for numerous chemicals in sediment from freshwater and marine habitats in the United States (MacDonald 1994, Ingersoll et al. 1996, MacDonald et al. 2003) and Canada (Smith et al. 1996, CCME 2002).

Advantages and Limitations:

Advantages (and limitations) for the Effects Range approach also apply to the Effects Level approach (MacDonald et al. 1996, Crane et al. 2000). The geometric mean is used rather than the arithmetic mean as in the Effects Range approach, in recognition that the data are not normally distributed. Also the inclusion of no-effects data ensures representative statistics (MacDonald 1994, Crane et al. 2000).

9.2.6 Logistic Regression Modelling approach

The Logistic Regression Modelling approach is based on statistical analysis of matching chemistry and biological effects for a single endpoint (e.g. amphipod toxicity), and was developed as an alternate approach to the threshold approach for the derivation of sediment quality guidelines (Field et al. 1999, Field et al. 2002). A large database of marine/estuarine sediment samples with matching chemistry and toxicity test results (two species of marine/estuarine amphipods) was assembled. Data for individual sediment samples were sorted by ascending concentrations for each particular contaminant. The data were screened to reduce the influence of samples that did not contribute to the toxic effects associated with the specific contaminant of interest. A logistic regression model was then applied to the screened data that described the relationship between the concentration of a selected contaminant and the probability of observed toxicity. The logistic model can be simplified and described by the following equation:

$$p = \frac{\exp(B_0 + B_1(x))}{1 + \exp(B_0 + B_1(x))}$$

Where:

p = probability of observing a toxic effect based on a single chemical,

B_0 = intercept parameter,

B_1 = slope parameter, and

x = concentration or log concentration of the chemical.

Individual chemical regression models were combined into a single mixture effects model based on the maximum probability of effects, or P_{\max} (Field et al. 2002). The maximum probability obtained from the individual chemical models is selected to represent the chemical mixture present in a sample.

The Logistic Regression Modelling approach does not yield specific sediment quality guideline values for each chemical, but rather describes the relationship between contaminant concentrations and the probability of toxicity. The utility of this approach is that it can be used to determine the concentration of a contaminant that corresponds to any probability of observing toxicity. A manager can thus identify an acceptable probability of observing sediment toxicity at a site (e.g. 25%) and determine the corresponding chemical concentrations. The calculated value can then be used as the sediment quality guideline for the site.

The Logistic Regression Modelling approach has been used in California as a line of evidence for evaluating the quality of sediment in enclosed bays and estuaries. The approach has been calibrated to this state, using locally derived data.

Advantages:

The probability that a given sediment sample is toxic can be taken directly from the logistic regression curve. The probability of toxicity at pre-defined levels (e.g. 25%) can also be determined. This is useful when developing sediment quality objectives (i.e. the level of protection desired in terms of adverse (toxic) effects; Field et al. 1999).

Limitations:

The approach is data intensive and has a high false-positive rate (Wetherington et al. 2005).

9.2.7 Consensus-based approach

As the term implies, Consensus-based sediment quality guidelines are intended to reflect the agreement among various sediment quality guidelines with the same narrative intent, by providing an estimate of their central tendency. Consensus-based sediment quality guidelines are, therefore, considered to provide a unifying synthesis of the existing sediment quality guidelines and to account for the effects of contaminant mixtures in sediment (Swartz 1999, MacDonald et al. 2000).

The consensus stems from the idea that if different approaches for deriving sediment quality guidelines result in a quantitatively similar concentration, then the validity of the result is greatly enhanced (Batley et al. 2005). The geometric rather than the arithmetic mean is calculated because it provides an estimate of central tendency that is not unduly affected by outliers, and because the distributions of sediment quality guidelines is not known.

The consensus approach has been used to derive sediment quality guidelines for freshwater and marine habitats in the USA (e.g. Swartz 1999, Ingersoll and Macdonald et al. 1996, MacDonald et al. 2000, Crane et al. 2002).

Advantages and Limitations:

Since the Consensus approach incorporates guidelines developed through different approaches, the advantages and disadvantages inherent in these approaches are inherent in consensus-based sediment quality guidelines. (MacDonald et al. 2000).

9.2.8 Floating Percentile approach

The most recent approach to the derivation of empirical sediment quality guidelines is the Floating Percentile approach. A significant percentage of the error in empirical approaches to sediment quality guideline derivation is related to the use of a single percentile of the distribution to set the criterion for all chemicals. Because all chemicals do not contribute equally to toxicity in a data set, this oversimplification results in substantial mathematical error. Unlike many other sediment quality guidelines, this approach does not require the sediment quality guidelines for all chemicals to be based on the same percentile distribution (e.g. the Effects Range approach). The basic concept behind the Floating Percentile approach is to select an optimal percentile of the data set that provides a low false negative rate and then adjust individual chemical concentrations upward until false positive rates are decreased to their lowest possible level while retaining the same false negative rate. Most chemicals should be at or near their actual toxicity range, rather than a level arbitrarily assigned by a fixed percentile. In this manner, optimized criteria sets can be developed for a number of different target false negative rates, allowing the trade-offs between false negatives and false positive to be evaluated and a final set of sediment quality guidelines to be selected.

The Floating Percentile approach has recently been used to update freshwater sediment quality guidelines for Washington, Oregon and Idaho in the USA (Avocet 2003).

10. Limitations of sediment quality guidelines

Although the limitations of sediment quality guidelines derived through different approaches were discussed above, it is worthwhile summarising the major limitations common to all sediment quality guidelines since their implications will need to be considered when formulating sediment quality guidelines for the revised National Action List.

10.1 Sediment quality guidelines are protective of a limited number and type of potential receptors

Sediment quality guidelines are usually derived to be protective of sediment-dwelling organisms. Very few jurisdictions/countries have derived sediment quality guidelines that are protective of higher trophic levels, such as fish, which are susceptible to exposure through

bioaccumulation and biomagnification. By implication, sediment quality guidelines are not protective of human health, by exposure through the consumption of contaminated fish and shellfish (USACE 1998).

10.2 Sediment quality guidelines are not available for all chemicals

There is an extremely wide range of chemicals for which sediment quality guidelines have not been derived. There are literally thousands of chemicals and their breakdown products that have the potential to cause adverse biological effects, but about which little is known. Peters and Gandrass (2001) estimated that less than 0.5% of registered compounds are usually covered by monitoring programmes. The sediment quality guidelines defined by Long et al. (1995) provide guidelines for less than one-third of the chemicals that have been measured in sediment from the Port of Los Angeles in California. Assessing the quality of sediment using sediment quality guidelines thus bases management decisions on a limited number of chemicals, and discounts the many other chemicals that might be causing adverse biological effects even if all chemicals for which there are guidelines are present at concentrations below the guidelines.

10.3 Sediment quality guidelines do not address unanticipated chemicals

Sediment quality guidelines provide no means for evaluating chemicals that may be present in sediment but are unknown to the assessor, yet are perhaps of environmental importance (USACE 1998, Ingersoll et al. 2005; Long et al. 2006). As mentioned above, Peters and Gandrass (2001) estimated that less than 0.5% of registered compounds are usually covered by monitoring programmes. The adage that 'you can only find what you are looking for' has relevance, but in this case should read 'you can only find what you measure for'. Thus, concentrations of all measured chemicals in sediment could conceivably be below sediment quality guidelines, yet the sediment could still pose an environmental risk due to other chemicals that were not measured or were unknown.

10.4 Sediment quality guidelines do not address the interaction of chemicals

While the data used to derive sediment quality guidelines often include field sediments that may contain multiple contaminants, the concentrations of individual contaminants are considered separately in the derivation process. No approach has been proposed to develop sediment quality guidelines that identify the

potential interactions of chemicals present together in the sediment (USACE 1998, Chapman and Mann 1999, Wenning et al. 2002). Such interactions could be additive, synergistic or antagonistic. Contaminated sediments often contain a variety of metallic and organic contaminants and the potential combinations and permutations of these contaminants are almost infinite.

10.5 Sediment quality guidelines do not adequately consider the exposure component of environmental risk

At a fundamental level, an environmental risk can exist only if there is (1) an effect due to some stressor(s) and (2) exposure of some receptor or organism to the stressor(s). One of the reasons that exceedance of sediment quality guidelines cannot predict adverse biological effects is that sediment quality guidelines are derived primarily on the basis of effect-related considerations and inadequately address exposure-related considerations (Chapman 2007).

10.6 Sediment quality guidelines are often poor predictors of adverse biological effects

Many workers consider sediment quality guidelines to be reasonably 'predictive' of adverse biological effects (Chapman and Mann 1999, Wenning et al. 2002). These conclusions have been reached by evaluating the incidence of adverse biological effects in field collected sediment against the sediment quality guidelines, which have shown that the guidelines provide a reliable basis for assessing sediment quality conditions (e.g. MacDonald et al. 2000). At the same time, other workers have shown that sediment quality guidelines may be poor predictors of adverse effects in other situations (e.g. O'Connor and Paul 2000, Bay et al. 2004, Lee et al. 2004, O'Connor 2004, Vidal and Bay 2005). Reliance solely on sediment quality guidelines that are derived from co-contaminated sediments such that the toxic effect is wrongly attributed to all contaminants tends to produce a high proportion of false positives, which can result in contaminated sites being incorrectly identified as polluted (Wetherington et al. 2005).

10.7 Sediment quality guidelines do not account for bioavailability

Site-specific sediment characteristics (e.g. grain size, organic carbon, pH, redox potential, acid volatile sulphides) and biotic factors (e.g., bioturbation, bioirrigation) can significantly influence the bioavailability (and hence toxicity) of contaminants.

While sediment quality guidelines are typically derived from data from many locations representing a range of sediment conditions, the resulting contaminant concentration values cannot possibly be applied to all sediment conditions (Chapman and Mann 1999).

10.8 Sediment quality guidelines do not account for bioaccumulation/biomagnification¹

Sediment quality guidelines are intended for the protection of sediment associated benthic invertebrates, through consideration of direct toxicity (acute or chronic). They are not designed to be protective of higher trophic levels, such as fish, either through direct contact or through dietary transfer (Chapman and Mann 1999, CCME 2002, Word et al. 2002). Bioaccumulation and the rarer phenomenon of biomagnification (increasing concentrations up three or more trophic levels, Chapman et al. 2003) need to be assessed separately (Chapman and Mann 1999), using dedicated testing procedures and through an ecological risk assessment. Biomagnification is only a significant concern for mercury in its methylated form (Chan et al. 2003).

11. The need for multiple lines of evidence when assessing sediment quality

As discussed above, there is growing consensus amongst the scientific community that sediment quality guidelines should not be used as standalone tools for decision-making, but rather should comprise one of several lines of evidence in a weight of evidence decision-making framework. Weight of evidence frameworks use evidence from several, separate lines of evidence, such as physical, chemical, and biological information. These lines of evidence evaluate effects at different levels of biological organisation.

11.1 Sediment chemistry

Sediment chemistry is determined by extracting and measuring the concentrations of chemicals from the sediment matrix through various analytical procedures and techniques in the laboratory. The sediment matrix

includes the sediment itself as well as the interstitial porewater. Sediment chemistry measurements can provide information on the spatial and temporal extent of contamination at a site, depending on the type of sampling design that is implemented.

One of the principal strengths of sediment chemistry is that it provides direct information on the presence of contaminants in sediments, relative to anthropogenic background or reference conditions. If background or baseline concentrations have been defined, then sediment chemistry provides a valuable early warning tool of contamination, and may be able to detect contamination before it reaches a level where adverse biological effects start to manifest. In addition, standard methods have been established for determining the concentrations of many analytes in sediment. Because measurements of sediment chemistry can be both accurate and precise, they provide a reliable basis for discriminating between contaminated and uncontaminated sites. Furthermore, analytical methods have been developed which may provide an indication of the potential bioavailability of certain substances (e.g. metals), although the applicability of these methods (such as weak acid digestions) has not been fully established.

The main limitation of sediment chemistry data is that while the concentrations of chemicals can be used to provide an indication of whether the sediment is contaminated, chemical concentrations by themselves cannot be used to determine whether pollution has occurred, that is, whether the contamination has caused an adverse biological effect. Such measurements provide information on contamination (substances present where they would not normally occur, or at concentrations above natural background) and exposure (i.e. concentrations to which sediment-associated organisms are exposed), but they do not provide information on pollution (contamination that causes adverse biological effects in the natural environment; Chapman 2007). All pollutants are contaminants, but not all contaminants are pollutants because substances introduced into the environment may be more or less bioavailable to organisms depending on their chemical form, modifying factors in the environment, the environmental compartment they occupy, and the reactions (behavioural and physiological) of exposed organisms (Chapman et al. 2003). Elevated contaminant concentrations do not always cause adverse biological effects (e.g. Burton 1991, Paine et al. 1996, McPherson et al. 2008), although adverse biological effects are usually associated with elevated contaminant

¹ The terms bioaccumulation and biomagnification are not meant to be used here interchangeably, but rather in the senses defined in USEPA (2000). Bioaccumulation is the accumulation of contaminants in the tissues of organisms through any pathway, including respiration, ingestion, or direct contact with water, sediment and pore water. Biomagnification is recognized as the result of the process of bioaccumulation where the concentration of the contaminant increases as it is transferred through more than two trophic levels.

concentrations. The reason that adverse effects are not always observed is that numerous physicochemical characteristics of sediment control the bioavailability, and hence toxicity of metals. Furthermore, aquatic organisms have evolved strategies to both conserve and regulate essential metals in sediments in certain elevated concentration ranges without adverse biological effects (Chapman et al. 1999). It is for these reasons that sediment chemistry is used for screening purposes, to determine whether further assessment is warranted.

11.2 Biological assessment

As stated above, if the chemical analysis of sediment demonstrates that chemicals are present this does not imply that they are having an adverse biological effect since the chemicals may not be present in a bioavailable form. Toxicity can only occur if chemicals are in a bioavailable form, that is, in a form that can be taken up (i.e. can cross biological membranes) by organisms. Because bioavailability depends on the chemical and the environmental matrix, sediments with the same contaminant concentrations may exhibit very different toxicities (Newman and Unger 2003). In addition to causing direct effects on aquatic organisms (e.g. toxicity), contaminants can accumulate in the tissues of sediment-dwelling and sediment-associated organisms. Because many benthic and epibenthic organisms represent important components of the food web, sediment-associated contaminants can be transferred to higher trophic levels. In this way, contaminated sediments represent a potential hazard to aquatic predators (e.g. fish), to wildlife species that consume aquatic organisms (e.g. otters), and to humans that consume contaminated fish and shellfish.

Determining when contamination has resulted in pollution requires not only chemical but also biological measurements (Chapman 2007). Appropriate biological studies are the ultimate measure of whether environmental damage has occurred (Chapman et al. 1999). Biological tests provide a means for assessing contaminant bioavailability as organisms are only expected to respond to that fraction of sediment-associated contaminants that can be desorbed from sediment particles and brought into contact with organism tissues (PIANC 2005). A brief overview of some biological assessment techniques for sediment is provided below.

11.2.1 Sediment toxicity tests

Sediment toxicity tests are an important component of sediment quality assessments in many jurisdictions/

countries. Sediment toxicity testing entails the exposure of test organisms to field-collected sediments in the laboratory. The laboratory environment allows for the control of confounding variables that may vary in the field, such as salinity, temperature or dissolved oxygen, thus permitting a distinction between toxic effects and effects due to natural habitat variability. Tests have been developed to assess lethal (acute) and sub-lethal (chronic) endpoints. The tests may be as simple as a short-term (relative to organisms lifecycle) test on a single contaminant using a single species, or as complex as mesocosm studies in which the long-term effects of mixtures of contaminants on ecosystem dynamics are investigated. To measure toxicity, a specific biological endpoint (e.g. mortality, reductions in growth or reproduction) is used to assess the response of the organisms to contaminants in the sediment.

Whole sediment (also sometimes called bulk or solid-phase) toxicity tests are the most relevant for assessing the effects of contaminants that are associated with sediments. However, tests have also been developed to assess the toxicity of suspended sediments, sediment elutriates and extracts, and porewater. Many of the latter tests have specific application to assessing the potential environmental impacts associated with the dredging process and the openwater disposal of dredged material. Usually, regulatory authorities require that the assessment of whole sediment be performed using a suite of tests that measure for both acute and chronic endpoints in organisms from a range of trophic levels.

Toxicity tests have a number of advantages that make them relevant for evaluating the effects of contaminated sediments on aquatic organisms. Standard methods have been established to support the generation of reliable and comparable data, as well as to minimise the confounding effects of the physical and chemical characteristics of sediments on data interpretation. The results of many of the tests are ecologically-relevant because they commonly use resident species and provide a means to compare the sensitivities of different organisms. They provide quantitative information on sediment toxicity that provides a basis for discriminating between impacted and un-impacted sites. They measure the effects of unmeasured or unknown chemicals in the sediment, including those for which there are no sediment quality guidelines and a decision based on sediment chemistry and sediment quality guidelines alone fail to consider. They measure the interactive effects of complex mixtures of chemicals in sediment. Toxicity tests also take into account the bioavailability of contaminants in sediment.

Toxicity tests have several limitations that influence their application in sediment quality assessments. Many of the tests that are currently available involve the short-term exposure (i.e. 10 day) of test organisms to sediment and, hence, might not be sensitive enough to detect subtle, chronic effects on sensitive species resulting from long-term exposure to chemicals. Field-collected sediments are manipulated prior to testing, which may affect their integrity and toxicity (Burton 1991, Crane et al. 2000). The highly controlled conditions in sediment toxicity tests do not replicate conditions in the natural environment (Chapman et al. 2002). Certain sediment phases (e.g. organic extracts, elutriates) may be less relevant for evaluating the *in situ* effects of toxic substances in sediments. Certain test organisms may be more sensitive to certain classes of contaminants than other organisms, and in some cases their use in testing the toxicity is of little relevance since they will never be exposed to sediment. For example, measuring the toxicity of sediment porewater using echinoid and bivalve larvae has little ecological relevance since the larvae are unlikely to ever come into contact with sediment porewater. Therefore, it is necessary to use a suite of test organisms to cover the range of sensitivities exhibited by sediment-dwelling and sediment-associated organisms in the field. This has significant cost implications since sediment toxicity tests are time-intensive.

11.2.2 Benthic invertebrate community assessment

Benthic invertebrate communities are assemblages of invertebrates that live in or on sediment. Analysis of these communities has for a long time provided one of the mainstays of sediment quality assessments. In terms of evaluating sediment quality, benthic invertebrate community assessment focuses on establishing whether there are similarities or differences in various community metrics (e.g. species richness, total abundance) between putatively impacted and un-impacted sites, and on the relationships between these metrics and physical and chemical measures of sediment quality (e.g. grain size, organic content, chemical concentrations).

Benthic invertebrate community assessments provide a number of advantages in the context of sediment quality assessment. Because most benthic invertebrates are relatively sedentary, are in direct contact with sediment, have life-spans ranging from months to years, and show different sensitivities to contaminants, they tend to be sensitive to short-term and long-term changes in sediment conditions. They also reflect the influence of

various actual stressors in the environment, in contrast to toxicity tests which are unable to replicate these conditions in the laboratory. The information generated from the assessment of benthic invertebrate communities has ecological relevance, since they are important components of aquatic food webs; the loss or reduction in abundance and diversity of benthic invertebrate communities has a direct impact on higher trophic levels.

The limitations of benthic invertebrate community assessments that restrict their use for evaluating contaminant-related impacts are numerous. While benthic invertebrate communities respond to chemical contamination of the sediment (Canfield et al. 1994, 1996), they are also affected by a wide range of physical and chemical factors that are not directly related to sediment contaminants (e.g. low dissolved oxygen concentrations, grain size differences, and water depth). These confounding variables complicate the attribution of differences in community composition to the presence of contaminants in sediment (Breneman et al. 2000, Thompson and Lowe 2004, Holmstrup et al. 2010). Benthic invertebrate community composition exhibits significant spatial, short-term temporal and seasonal variability, further complicating interpretation of the data relative to contaminant effects. Large numbers of samples are also required to the high variance associated with distribution of benthos (USEPA 1992). There is little agreement among benthic ecologists on which metrics are the most appropriate for evaluating the status of benthic invertebrate communities. Although a plethora of benthic response indices have been developed for the purpose of identifying benthic invertebrate community disturbance, no single index has achieved favour amongst benthic ecologists. Different indices do not necessarily provide a comparable assessment of benthic invertebrate community status. Part of the reason is that benthic response indices may be area specific, and thus need to be developed on this basis.

11.2.3 Bioaccumulation assessment

Certain highly bioaccumulative substances may not have any direct toxic effect on benthic organisms that are continuously exposed to very small doses of these substances, but the organisms may nonetheless accumulate the substances in their tissues and transfer them to higher trophic levels (including humans) through dietary transfer. Such chemical substances include polychlorinated biphenyls, methylmercury, furans and toxaphene. Bioaccumulation testing usually involves the exposure of test organisms to sediments under

controlled conditions in the laboratory for a period of 28 days. The organisms used in bioaccumulation tests are generally selected on the basis of their relative tolerance to contaminants (i.e. they survive the exposure) and their body size, which ensures that there is sufficient tissue recovered at the end of the exposure for chemical analysis. Bioaccumulation is determined by comparing concentrations of chemicals in organisms exposed to test sediment samples to concentrations in test organisms that were concurrently exposed to uncontaminated (reference) sediment, or by comparing the chemical concentrations to tissue residue guidelines if these are available.

Bioaccumulation tests provide a measure of exposure rather than effect. Bioaccumulation of a chemical will not always result in an adverse effect on the organism. In the case of essential nutrients (e.g. zinc and copper), a certain amount of accumulation is required to support normal physiological function. In general, adverse effects from any contaminant will only be manifest after the concentration exceeds a specific tolerance level or toxicological threshold (PIANC 2005).

The primary disadvantage of conducting sediment bioaccumulation tests is their cost, which is influenced by the time-intensive nature of the tests. Quality assurance requirements are also more stringent for bioaccumulation tests than for conventional sediment toxicity tests. In addition, an adequate tissue mass must be obtained for chemical analysis. This could require the use of additional replicates for analyses of multiple chemicals. Another disadvantage is that field collected sediments may include indigenous organisms that need to be separated from the test organisms at the end of the test. This manipulation of the sediment influences its chemical characteristics. An appropriate reference control site must also be selected, but this is often not a simple task. Finally, the relationships between bioaccumulation and direct toxicity to aquatic organisms are poorly defined.

12. Action Lists from various jurisdictions/countries

This section provides a brief review of sediment quality guidelines included in Action Lists used for making decisions on the suitability of dredged material for unconfined, openwater disposal in some jurisdictions/countries, and how the guidelines are used for decision-making. The review is by no means exhaustive, but is included to provide the reader with perspective on the ranges of metal concentrations specified by sediment quality guidelines. In line with the revision of only the

metal sediment quality guidelines of the existing National Action List, the review focuses only on metal guidelines.

This review also does not focus on decision-making frameworks within which the sediment quality guidelines/Action Lists are embedded. The frameworks usually adopt a tiered approach to decision-making, with each tier having explicit data requirements and successive tiers usually having broader data requirements. Although sediment quality guidelines are sometimes referred to as Action Levels in guidance documents, in keeping with the philosophy of Action Levels as per the Waste Assessment Guidelines of the London Protocol these are referred to as Level I and Level II guidelines wherever possible.

12.1 Europe

12.1.1 Belgium

Belgium has adopted a two Action Level approach for reaching a decision on the suitability of dredged material for unconfined, openwater disposal (Table 1). The sediment quality guidelines of the Action List were derived from chemical concentrations measured in sediment collected at reference sites along the Belgian coastline. Sediment with chemicals at concentrations at or below the Level I guideline is not considered to pose a risk and is suitable for unconfined, openwater disposal. In the context of metals, concentrations that are compared to the sediment quality guidelines are the arithmetic average of analyses on ten sediment samples. If the concentrations of chemicals fall between the Level I and Level II guidelines, then an additional five sediment samples from the same area must be analysed. If the analyses confirm results of the previous analyses, then biological testing is required to determine whether the elevated chemical concentrations are associated with adverse biological effects. If toxic effects are observed then this may lead to the prohibition of the unconfined,

Table 1. Level I and Level II metal sediment quality guidelines ($\mu\text{g.g}^{-1}$ dry weight) used for reaching a decision on the suitability of dredged sediment for unconfined, openwater disposal in Belgian coastal waters.

Metal	Level I	Level II
Arsenic	20	100
Cadmium	2.5	7
Chromium	60	220
Copper	20	100
Mercury	0.3	1.5
Nickel	70	280
Lead	70	350
Zinc	160	500

openwater disposal of dredged material. If the concentrations of three or more contaminants exceed the Level II guideline, then the dredged material is considered unsuitable for unconfined, openwater disposal.

12.1.2 Denmark

Denmark has adopted a two Action Level approach for reaching a decision on the suitability of dredged material for unconfined, openwater disposal (Table 2). The sediment quality guidelines of the Action List are based on data from the Netherlands and Finland, with some adjustments to Danish circumstances. The Level I guideline corresponds to the background concentration or a no-effect concentration. The Level II guideline is based on international guidelines. Sediment with chemicals at concentrations at or below the Level I guideline is not considered to pose a risk and is suitable for unconfined, openwater disposal. If the concentrations of chemicals fall between the Level I and Level II guidelines, a comprehensive assessment is required. The scope of this assessment is based on the amount of sediment that requires disposal and the concentrations of chemicals in the sediment. Sediment with chemicals at concentrations above the Level II guideline is usually not permitted for unconfined, openwater disposal.

Table 2. Level I and Level II metal sediment quality guidelines ($\mu\text{g}\cdot\text{g}^{-1}$ dry weight) used for reaching a decision on the suitability of dredged sediment for unconfined, openwater disposal in Denmark.

Metal	Level I	Level II
Arsenic	20	60
Cadmium	0.4	2.5
Chromium	50	270
Copper	20	90
Mercury	0.25	1.0
Nickel	30	60
Lead	40	200
Zinc	130	500

12.1.3 Finland

Finland has adopted a two Action Level approach for reaching a decision on the suitability of dredged material for unconfined, openwater disposal (Table 3). The sediment quality guidelines of the Action List were derived using sediment background concentrations and ecotoxicological approaches. For metals, the Level I guideline is 1.5 times the background concentration with the exception of mercury, which is 2.5 times the background concentration. The Level II guideline has, in general, been set using ecotoxicological data. Sediment with chemicals at concentrations at or below the Level I

guideline is not considered to pose a risk and is suitable for unconfined, openwater disposal. If the concentrations of chemicals fall between the Level I and Level II guidelines, decisions on its suitability for unconfined, openwater disposal are made on a case-by-case basis and might require further assessment. Sediment with chemicals at concentrations at or above the Level II guideline is considered unsuitable for unconfined, openwater disposal.

Table 3. Level I and Level II metal sediment quality guidelines ($\mu\text{g}\cdot\text{g}^{-1}$ dry weight) used for reaching a decision on the suitability of dredged sediment for unconfined, openwater disposal in Finland.

Metal	Level I	Level II
Arsenic	15	60
Cadmium	0.5	2.5
Chromium	65	270
Copper	50	90
Mercury	0.1	1.0
Nickel	45	60
Lead	40	200
Zinc	170	500

12.1.4 France

France has adopted a two Action Level approach for reaching decisions on the suitability of dredged material for unconfined, openwater disposal (Table 4). The sediment quality guidelines of the Action List were defined from concentrations of chemicals measured in sediment collected from French ports. The Level I and Level II guidelines for metals were defined as 2 and 4 times the median concentration. Sediment with chemicals at concentrations at or below the Level I guideline is not considered to pose a risk and is suitable for unconfined, openwater disposal. If the concentrations of chemicals fall between the Level I and Level II guidelines, a comprehensive study might be necessary (including at least toxicity testing). The scope of the assessment is determined on a case by case basis, taking into account local circumstances and the

Table 4. Level I and Level II metal sediment quality guidelines ($\mu\text{g}\cdot\text{g}^{-1}$ dry weight) used for reaching a decision on the suitability of dredged sediment for unconfined, openwater disposal in France.

Metal	Level I	Level II
Arsenic	25	50
Cadmium	1.2	2.4
Chromium	90	180
Copper	45	90
Mercury	0.40	0.80
Nickel	37	74
Lead	100	200
Zinc	276	552

sensitivity of the environment. If the concentrations of chemicals exceed the Level II guideline, unconfined, openwater disposal might be prohibited, especially if openwater disposal does not constitute the least detrimental solution for the environment. Thus, contaminant concentrations exceeding the Level II guideline does not lead to the automatic prohibition of unconfined, openwater disposal.

12.1.5 Germany

Germany has adopted a two Action Level approach for reaching decisions on whether material dredged from federal waterways is suitable for unconfined, openwater disposal (Table 5). The Level I guideline is based on chemical concentrations measured in North Sea sediments, multiplied by a factor of 1.5 to take into account accidental uncertainties due to sampling and chemical analysis. The Level II guideline is set at 5 times the Level I guideline. All concentrations of trace metals are specific to <20 µm sediment fraction. Sediment with chemicals at concentrations at or below the Level I guideline is not considered to pose a risk and is suitable for unconfined, openwater disposal. If the concentrations of chemicals fall between the Level I and Level II guidelines, the material might be deemed suitable for disposal after further analysis which is dependent on the actual contaminant concentrations and an evaluation of the likely impact on the ecosystem. If the concentrations of chemicals exceed the Level II guideline, unconfined, openwater disposal is subject to mitigation measures to reduce environmental impacts. Thus, contaminant concentrations exceeding the Level II guideline does not lead to the automatic prohibition of unconfined, openwater disposal.

Table 5. Level I and Level II metal sediment quality guidelines ($\mu\text{g.g}^{-1}$ dry weight) used for reaching a decision on the suitability of dredged sediment for unconfined, openwater disposal in Germany.

Metal	Level I	Level II
Arsenic	30	150
Cadmium	2.5	12.5
Chromium	150	750
Copper	40	200
Mercury	1	5
Nickel	50	250
Lead	100	500
Zinc	350	1750

12.1.6 Ireland

Ireland has adopted a two Action Level approach for reaching decisions on the suitability of dredged material for unconfined, openwater disposal (Table 6). With the exception of arsenic and nickel, Level I guidelines were

derived as the 95th percentile of concentrations measured in sediment samples collected from reference sites along the Irish coastline. There was insufficient arsenic and nickel data from reference sites to derive a guideline. The Effects Range Low guideline of the Long et al. (1995) sediment quality guidelines were adopted for arsenic and nickel. The Effects Range Median of the Long et al. (1995) sediment quality guidelines was used for all Level II guidelines.

Decisions on the suitability of dredged sediment for unconfined, openwater disposal are made using a weight of evidence approach. The sediment quality guidelines of the Action Levels are used to define three classes of sediment, as:

Class 1: chemical concentrations below Level I guideline.

The sediment is considered to be uncontaminated and is suitable for unconfined, openwater disposal.

Class 2: chemical concentrations between Level I and Level II guidelines. The sediment is considered to be marginally contaminated, and further sampling and analysis is necessary to delineate the problem area before reaching a decision on whether the material is suitable for unconfined, openwater disposal.

Class 3: chemical concentrations above Level II guideline.

The sediment is considered to be heavily contaminated and very likely to cause adverse biological effects. This sediment is unsuitable for unconfined, openwater disposal and alternative disposal options must be considered.

Table 6. Level I and Level II metal sediment quality guidelines ($\mu\text{g.g}^{-1}$ dry weight) used for reaching a decision on the suitability of dredged sediment for unconfined, openwater disposal in Ireland.

Metal	Level I	Level II
Arsenic	9	70
Cadmium	0.7	4.2
Chromium	120	370
Copper	40	110
Mercury	0.2	0.7
Nickel	21	60
Lead	60	218
Zinc	160	4

12.1.7 Spain

Spain has adopted a two Action Level approach for reaching decisions on the suitability of dredged material for unconfined, openwater disposal (Table 7). The sediment quality guidelines of the Action List were defined using the Sediment Background approach and are specific to chemicals in the <63 µm fraction. Sediment with chemicals at concentrations at or below the Level I guideline is not considered to pose a risk and

is suitable for unconfined, openwater disposal. If the concentrations of chemical fall between the Level I and Level II guidelines, the sediment is considered moderately contaminated and requires further assessment before a decision can be made as to its suitability for unconfined, openwater disposal. Sediment with chemical at concentrations at or above the Level II guideline is not suitable for unconfined, openwater disposal.

Table 7. Level I and Level II metal sediment quality guidelines ($\mu\text{g.g}^{-1}$ dry weight) used for reaching a decision on the suitability of dredged sediment for unconfined, openwater disposal in Spain.

Metal	Level I	Level II
Arsenic	80	200
Cadmium	1	5
Chromium	200	1000
Copper	100	400
Mercury	0.6	3
Nickel	100	400
Lead	120	600
Zinc	500	3000

12.1.8 United Kingdom (England and Wales)

The United Kingdom has adopted a two Action Level approach for reaching decisions on the suitability of dredged material for unconfined openwater disposal (Table 8). A nominal 'background' concentration was used to derive the Level I guideline. Ecotoxicological data, based largely on data from North America, was used to formulate the Level II guideline. In general, chemicals at concentrations at or below the Level I guideline are of no concern and the sediment is considered suitable for unconfined, openwater disposal. Sediment with chemicals at concentrations between the Level I and Level II guidelines requires further testing before a decision is made on its suitability for unconfined, openwater disposal. This can involve additional sampling and/or further analysis, or the use of bioassays to more effectively assess ecological risk.

Table 8. Level I and Level II metal sediment quality guidelines ($\mu\text{g.g}^{-1}$ dry weight) used for reaching a decision on the suitability of dredged sediment for unconfined, openwater disposal in United Kingdom (England and Wales).

Metal	Level I	Level II
Arsenic	20	70
Cadmium	0.4	4
Chromium	50	370
Copper	30	300
Mercury	0.25	1.5
Nickel	30	150
Lead	50	400
Zinc	130	600

Sediment with chemicals at concentrations at or above the Level II guideline is generally not considered suitable for unconfined, openwater disposal

12.2 North America

12.2.1 United States of America

The United States of America comprises a large geographical area and it is perhaps not surprising that nationally applicable sediment quality guidelines have not been derived or formulated. Regulations in the United States of America for the disposal of dredged material do not require nor endorse use of sediment quality guidelines for decision making purposes (USACE 1998). The reason that little reliance is placed on sediment quality guidelines is that there is a concern the guidelines have such a high level of uncertainty that they may incorrectly designate non-contaminated sediments as contaminated, thereby requiring expensive and unnecessary removal and remediation actions. This said, there is considerable latitude from state to state, and as discussed below some states place significant emphasis on the use of sediment quality guidelines for decision-making.

Using the Pacific Northwest (states of Washington, Oregon and Idaho) as an example, sediment quality guidelines designed to be protective of direct effects to benthic and aquatic organisms have been developed using the Apparent Effects Threshold approach. Two guidelines were derived, based on different criteria for the acceptability of sediment toxicity testing results (Table 9). The Level I guideline corresponds to a concentration at or below which there is no reason to believe that dredged material disposal will result in unacceptable adverse effects, and the material is hence suitable for unconfined, openwater disposal. Chemicals present at concentrations between the Level I and Level II guidelines require additional biological assessment for decision-making. This includes a suite of three acute and chronic toxicity tests. When any one biological test

Table 9. Level I and Level II metal sediment quality guidelines ($\mu\text{g.g}^{-1}$ dry weight) used for reaching a decision on the suitability of dredged sediment for unconfined, openwater disposal in the Pacific Northwest of the United States.

Metal	Level I	Level II
Arsenic	57	93
Cadmium	5.1	6.7
Chromium	260	270
Copper	390	390
Mercury	0.41	0.59
Lead	450	530
Zinc	410	960

exhibits a response that exceeds the toxicity-specific guidelines relative to the negative control and reference, and which is statistically significant in comparison to the reference, the sediment is considered unsuitable for unconfined, openwater disposal. When any two biological tests (amphipod, juvenile infaunal growth or sediment larval) exhibit responses that are less than the toxicity-specific reference-comparison guidelines for a single-hit failure, but are significant compared to the reference sediment, the sediment is judged to be unsuitable for unconfined, openwater disposal.

If one or more chemicals are present at concentrations above the Level II guideline, the dredging proponent is still permitted to perform biological testing but there is a high probability that the sediment will fail testing. Although some sediments with more than one Level II guideline exceedance have passed biological testing, the majority have failed. By comparing sediment chemical data to the Level II guideline, a dredging proponent is able to judge how to proceed with the project, that is, whether to invest more into testing for unconfined, openwater disposal or to rechannel that effort into other disposal options and testing for those options.

In addition to comparisons to Level I and Level II guidelines and subsequent determinations outlined above, a bioaccumulation guideline has also been derived and is used to determine when bioaccumulation testing is required. If a chemical of concern exceeds the Bioaccumulation Trigger concentration, additional information gained via bioaccumulation testing is required to determine whether the sediment is suitable for unconfined, openwater disposal. This testing is usually performed by exposing bivalves and polychaetes to sediment samples under controlled conditions in the laboratory for periods between 28 - 45 days. Tissue residues from bioaccumulation testing are compared to the Target Tissue Levels, which are the allowable concentrations of contaminants in tissue derived either from human-health risk assessments or from US Food and Drug Administration action levels.

12.1.2 Canada

The Canadian Council of Ministers of the Environment derived sediment quality guidelines for the protection of aquatic life using the Effects Level approach (CCME 2002). These guidelines, namely the Threshold Effect Level and the Probable Effect Level, have been adopted for the assessment of sediment quality in Quebec (Environment Canada and MDDEP 2007). However, the government of Quebec felt that these two guidelines alone were insufficient to address all of the different sediment management contexts, and three additional guidelines (Table 10) were derived to cover all of the management needs (using the same database as CCME 2002). These guidelines are discussed in further detail in a subsequent section of this report, and are hence only alluded to here in the context in which they are used for decision making with regard to dredged sediment. Two of the guidelines are specifically used to manage dredged sediment, namely the Occasional Effect Level and the Frequent Effect Level. Together, these effect levels categorize sediment into 3 classes of contamination that require different management decisions. Class 1 sediments, with contaminant concentrations below the Occasional Effect Level, are suitable for unconfined, openwater disposal or can be used for other beneficial uses (e.g. shoreline nourishment), because the probability of detecting adverse biological effects is deemed low. Class 2 sediments, with contaminant concentrations between the Occasional Effect Level and the Frequent Effect Level, have a higher probability of adverse biological effects, so unconfined, openwater disposal can be considered a valid option only if toxicity tests demonstrate that the sediments will not adversely affect the receiving environment. Class 3 sediments, with concentrations above the Frequent Effect Level, are prohibited for unconfined, openwater disposal.

12.3 Australasia

12.3.1 Australia

As an interim measure, Australia has adopted sediment quality guidelines defined for North American coastal

Table 10. Level I and Level II metal sediment quality guidelines ($\mu\text{g.g}^{-1}$ dry weight) used for reaching a decision on the suitability of dredged sediment for unconfined, openwater disposal in the province of Quebec, Canada.

Metal	REL	TEL	OEL	PEL	FEL
Arsenic	4.3	7.2	19	42	150
Cadmium	0.32	0.67	2.1	4.2	7.2
Chromium	30	52	96	160	290
Copper	11	19	42	110	230
Mercury	0.051	0.13	0.29	0.7	1.4
Lead	18	30	54	110	180
Zinc	70	120	180	270	430

waters by Long et al. (1995), with minor modifications to reflect local conditions (e.g. the guideline for arsenic was increased), since there are insufficient local data to derive empirical effects-based sediment quality guidelines (Table 11). The guidelines have been shown to perform per their narrative intent for sediment in at least the Sydney Harbour and vicinity of New South Wales (McCready et al. 2006). In the context of dredging, Australia uses only the lower of the guidelines defined by Long et al. (1995), namely the Effects Range Low (referred to as Screening Level in Australia). Assessment of the suitability for disposal begins with the calculation of the upper 95% confidence limit of the mean concentration measured for a chemical in an area of interest. If the concentrations of all chemicals are below the Screening Level, the dredged material is suitable for unconfined, openwater disposal. If the 95% upper confidence limit for one or more chemicals exceeds the Screening Level, then the chemicals are identified as contaminants of potential concern. Comparison to the ambient baseline concentration for sediments of comparable grain size is then required, due to the fact that the suitability of the Screening Level to Australian conditions has not been conclusively demonstrated. If the mean concentration of the chemicals in question is at or below the 80th percentile of their ambient baseline concentrations in the vicinity of the disposal site, the sediment is considered to be suitable for unconfined, openwater disposal even though the relevant Screening Level(s) were exceeded. If concentrations of chemicals in the material are found to be above ambient baseline concentrations, the chemical is again identified as a contaminant of potential concern and further testing is required. The first step is to determine whether contaminants in the material are mobile, through an elutriate test. The concentrations of chemicals measured in the elutriate are compared to ANZECC marine water quality guidelines (1994). If concentrations in the elutriate exceed relevant water quality guidelines, then chemical bioavailability testing is required. If the

chemicals are found not to be bioavailable, the material is suitable for unconfined, openwater disposal. If the chemicals are found to be bioavailable, then acute and chronic toxicity testing of sediment is required. If this fails, then bioaccumulation testing of the sediment is required. A decision on whether the material can be disposed in marine waters, and the type of disposal (i.e. unconfined versus confined) is made from the findings of this assessment.

12.3.2 New Zealand

New Zealand has adopted a two Action Level approach for reaching decisions on the suitability of dredged material for unconfined, openwater disposal (Table 12). As an interim measure, sediment quality guidelines defined for North American coastal waters by Long et al. (1995) have been adopted, since there are insufficient local data to derive empirical effects-based sediment quality guidelines. In fact, the decision-making process in the context of the suitability of dredged material for openwater disposal is based largely on the Australian guidelines that were in force at the time (1999). The Australian guidelines have since been reviewed and accounts for some of the differences in the assessment approach between these 'neighbouring' countries. Applying the Action List results in three possible categories of waste:

1. Dredged material that contains chemicals at geometric mean concentrations below the Level I guideline (ER-L) is considered not to pose a risk and is permitted for unconfined openwater disposal without further testing provided that no other adverse ecological effects are associated with the disposal (e.g. physical smothering of sensitive habitats).
2. Dredged material that contains chemicals at geometric mean concentrations that fall between the Level I and Level II (ER-M) guidelines requires further testing to determine its suitability for unconfined

Table 11. Level I and Level II metal sediment quality guidelines ($\mu\text{g}\cdot\text{g}^{-1}$ dry weight) used for reaching a decision on the suitability of dredged sediment for unconfined, openwater disposal in Australia.

Metal	Level I	Level II
Arsenic	20	70
Cadmium	1.5	10
Chromium	80	370
Copper	65	270
Mercury	50	220
Nickel	0.15	1
Lead	21	52
Zinc	200	410

Table 12. Level I and Level II metal sediment quality guidelines ($\mu\text{g}\cdot\text{g}^{-1}$ dry weight) used for reaching a decision on the suitability of dredged sediment for unconfined, openwater disposal in New Zealand.

Metal	Level I	Level II
Arsenic		
Cadmium		
Chromium		
Copper		
Mercury		
Nickel		
Lead		
Zinc		

openwater disposal. The first step is an elutriate test, to determine whether the concentrations of contaminants in the material are mobile. For assessment purposes, the concentrations of chemicals measured in the elutriate are compared to ANZECC marine water quality guidelines (1994). If chemical concentrations in the elutriate exceed the relevant water quality guideline, the toxicity of the material to water and benthic organisms is required, or an appropriate mixing zone must be agreed with the permitting authority. In some instances, the permitting authority may require testing of the toxicity of waste components to the benthic organisms even if these relevant water quality criteria are met, in recognition of the fact that water quality criteria currently in common use in New Zealand were derived overseas, and may not be fully protective of New Zealand's marine environment. In particular, if elutriate testing shows contaminant concentrations in water close to relevant water quality criteria, additional toxicity testing might be appropriate. If it is found to be non-toxic at the acute level, the material is then considered suitable for unconfined, openwater disposal dumping. If the material is found to be acutely toxic, the applicant may opt to (a) treat the waste to make it less contaminated, or (b) undertake a further level (level 4) of investigation, or (c) consider either disposing the material into a confined/contained site in the marine environment, or not to dispose of the material in the marine environment at all.

3. If the geometric mean concentration of one or more chemicals in the dredged material are above Level II guideline (ER-M), or the previous tier of testing found the material to be acutely toxic or otherwise failed to meet any of the stipulated criteria, the material is considered as being probably unsuitable for unconfined, openwater disposal. In this case, the dredging proponent may elect to carry out further acute toxicity testing, as well as an assessment of chronic toxicity and bioaccumulative effects and prepare a comprehensive assessment of environmental effects. A decision on whether the material can be disposed in marine waters, and the type of disposal (i.e. unconfined versus confined) is made from the findings of this assessment.

12.3 Asia

12.3.1 Hong Kong

The Hong Kong administrative region of the Peoples Republic of China has adopted a two Action Level approach for reaching decisions on the suitability of dredged material for unconfined, openwater disposal

(ETWB 2002; Table 13)). The sediment quality guidelines are based largely on those defined by Long et al (1995), because there are insufficient local data to derive empirical effects-based guidelines (Chapman 1999, ETWB 2002). Chemicals at concentrations below the Level I guideline are not expected to have adverse biological effects, while chemicals at concentrations above the Level II guideline are expected frequently be associated with adverse biological effects. Sediment is classified into three categories and decisions are made as follows:

- L: Sediment with all chemicals at concentrations below the Level I is suitable for unconfined, openwater disposal.
- M: Sediment with one or more chemicals at concentrations exceeding the Level I guideline but with none exceeding the Level II guideline must be dredged and transported with care, and must be effectively isolated from the environment upon disposal unless appropriate biological tests demonstrate that the material will not adversely affect the marine environment.
- H: Sediment with one or more chemicals at concentrations exceeding the Level II guideline requires biological testing to determine the appropriate disposal option. The material must be dredged and transported with great care, and must be effectively isolated from the environment upon disposal. Highly contaminated material (i.e. 10 x Lower Chemical Exceedance Level) requires treatment (e.g. incineration) to stabilise the material before disposal.

13. Existing South African National Action List

The National Action List that is under consideration for revision is provided in an undated, (apparently) draft document issued in about 1995 by the then Department of Environmental Affairs and Tourism: Sub-directorate

Table 13. Level I and Level II metal sediment quality guidelines ($\mu\text{g.g}^{-1}$ dry weight) used for reaching a decision on the suitability of dredged sediment for unconfined, openwater disposal in Hong Kong.

Metal	Level I	Level II
Arsenic	12	42
Cadmium	1.5	4
Chromium	80	160
Copper	65	110
Mercury	0.5	1
Nickel	40	110
Lead	75	270
Zinc	200	420

Sea Fisheries Research Institute (see Appendix 3). Although the National Action List has *per se* remained unchanged since the issuing of the document, changes in the way that decisions are made on the suitability of dredged material for unconfined, openwater disposal have been implemented by the DEA (compare Appendices 3 and 4). Because of these changes, occasional reference is made hereafter to the modified National Action List to distinguish it from the 'original' National Action List.

The sediment quality guidelines of the existing National Action List comprise guidelines for so-called Annex I (cadmium and mercury) and Annex II metals (arsenic, chromium, copper, lead, nickel and zinc). Designation of metals as Annex I or Annex II is per the terminology of the London Convention and the London Protocol. Although not of direct concern in this report, the National Action List document also provides guidelines for cyanide, fluoride, 'oils', organohalogens, organosilicon compounds, and pesticides, although a single guideline is provided for most of these chemicals (see Appendix 3). These chemicals are, however, not included in the modified National Action List (see Table 14). In the original National Action List document, the Level I guideline is referred to as the Action Level for

Annex I metals and the Special Care Level for Annex II metals. In the modified National Action List, however, the Level I guideline for all metals is referred to as the Action Level (Table 14). The terms Action Level and Special Care Level are, however, used forthwith, as per the original document. The Level II guideline for both Annex I and Annex II metals is referred to as the Prohibition Level. There are also summative guidelines for Annex I and Annex II metals, these too are (confusingly) referred to as the Action Level/Special Care Level and Prohibition Level (Table 14).

The sediment quality guidelines were formulated using a 'middle of the road' approach (Lynn Jackson, personal communication), based on the sediment quality guidelines from several jurisdictions/countries. Sediment quality guidelines for Hong Kong, Ireland, Iceland, Germany, Norway and the provinces of Quebec and Ontario in Canada are provided in the document (note that the sediment quality guidelines used in most of these jurisdictions/countries have since changed). No clarity is provided on how or which of the sediment quality guidelines from these jurisdictions/countries was used to formulate the sediment quality guidelines of the National Action List. There are a number of issues pertaining to sediment quality guidelines from the

Table 14. The modified National Action List used to make decisions on the suitability of dredged material for unconfined, openwater disposal in South African coastal waters. Note that the Level I guideline for cadmium and mercury in the original National Action List document is referred to as the Action Level and for the remaining metals as the Special Care Level. ppm = parts per million, or $\mu\text{g.g}^{-1}$.

Metal	Action level (ppm)	Prohibition level (ppm)
Cadmium	1.5-10.0	>10.0
Mercury	0.5-5.0	>5.0
or for a combined level of these two	1.0-5.0	>5.0
Arsenic	30-150	>150 (1000)
Chromium	50-500	>500
Copper	50-500	>500 (1000)
Lead	100-500	>500 (500)
Nickel	50-500	>500
Zinc	150-750	>750 (1000)
or a combined level of these substances:	50-500	>500 (1000)

Explanatory notes on application:

- Once the levels of contamination in the sediments have been determined through chemical analysis, these are compared with the Action Levels contained in the above List (presented as $\mu\text{g.g}^{-1}$ dry weight sediment).
- A decision on whether or not to require biological testing, or to prohibit disposal of the sediment at sea, is determined as follows:
 - If none of the metals measured exceed the Action Levels, then no biological testing is required, and the material can be dumped.
 - If the Action Levels for both Annex I metals (Cd and Hg) are exceeded, or the combined level of Cd and Hg is $>5 \mu\text{g.g}^{-1}$, then biological testing is required.
 - If the Action Level for either of the Annex I metals, and two or more of the Annex II metals are exceeded, then biological testing is required.
 - If the Action Levels of three or more Annex II metals are exceeded, and the total of Annex II metals is $>500 \mu\text{g.g}^{-1}$, then biological testing is required.
 - If the combined level of Annex II metals is $>1\,000 \mu\text{g.g}^{-1}$, then biological testing is required.

jurisdictions/countries listed in the document that have important implications should they have been used to formulate the sediment quality guidelines of the National Action List.

Firstly, the sediment quality guidelines from several of the jurisdictions/countries are not directly compatible. Sediment quality guidelines for the provinces of Quebec and Ontario in Canada, for example, were derived using the Screening Level approach, while (some of) those for Germany and Norway were derived using the Sediment Background approach. Secondly, and perhaps most importantly, the German sediment quality guidelines are specific to the <20 µm fraction of the sediment, while the guidelines for all other jurisdictions/countries are specific to <2000 µm fraction. The specification that guidelines are for metal concentrations in the <20 µm fraction has significant implications since the mass-specific concentration of metals in this fraction is far higher than in sediment comprised of the <2000 µm fraction (unless of course, but highly improbably, the sediment is entirely comprised of <20 µm material). In other words, the German guidelines should in theory be considerably lower if they were based on the <2000 µm fraction. Whether this was considered during formulation of the sediment quality guidelines of the existing National Action List is uncertain, but it would appear that the German sediment quality guidelines were the basis for at least two metal guidelines in the list.

The definition of two sediment quality guidelines in the existing National Action List invokes three concentration ranges of management interest, namely concentrations that fall below the Action Level/Special Care Level, concentrations that fall between the latter and the Prohibition Level, and concentrations that exceed the Prohibition Level. The National Action List document provides a basic but incomplete description of how the sediment quality guidelines should be used to assess sediment quality. Sediment is considered to be uncontaminated ('trace or insignificant contamination') if chemicals are at concentrations below the Action Level/Special Care Level. Sediment is considered moderately contaminated if concentrations fall between the Action Level/Special Care Level and Prohibition Level, and highly contaminated if concentrations exceed the Prohibition Level. The document acknowledges that while chemical analyses may reveal the presence of contaminants in sediment, this does not necessarily imply that the contaminants are exerting adverse biological effects. If sediment is classified moderately contaminated, then the document states that biological testing, such as toxicity or bioavailability testing, is

required to assist decision-making. No provision for further testing is made for sediment that contains chemicals at concentrations exceeding the Prohibition Level. The presumption is, therefore, that dredged material with chemicals at concentrations exceeding the Prohibition Level is immediately considered unsuitable for unconfined, openwater disposal, that is, that the Prohibition Level is used on a pass/fail basis. The modified version of the National Action List goes into more detail on how the sediment quality guidelines are used for decision-making, and confirms the pass/fail nature of the Prohibition Level (see explanatory note on application in Table 14).

14. Limitations of the existing National Action List

As mentioned previously, experience in assessing sediment quality in South African coastal waters has shown that the existing National Action List is limited from several perspectives, and it is these limitations that motivated for its revision. The principle limitations of the existing National Action List are discussed below. The limitations will need to be avoided or addressed in the revised National Action List.

14.1 Limitation 1: Some guidelines are inappropriate for South African coastal waters

The most significant limitation of the existing National Action List is that guidelines for some metals are lower than baseline concentrations in sediment from some or all regions of the South African coastline. Strict application of the sediment quality guidelines leads to the anomalous situation of identifying sediment with metal concentrations that fall within the expected concentration range for uncontaminated sediment as posing a potential risk to sediment-dwelling organisms. This obviously cannot be the case since the major portion of the metal concentration must be bound in the crystal lattice of sediment grains and hence is not in a bioavailable form. If the metal is not in a bioavailable form, it cannot exert toxicity. This limitation was identified following the definition of baseline concentrations for metals in sediment from some regions of the South African coastline (Newman and Watling 2007 and CSIR unpublished data). Baseline concentrations provide an important tool for screening sediment quality guidelines that might be considered for the revised National Action List. The following subsections thus provide background information on how baseline concentrations for metals are defined, the variability of metal baseline concentrations in sediment

from South African coastal waters, and the use of baseline concentrations for reaching a decision on whether sediment is metal contaminated. Examples of inappropriate sediment quality guidelines of the existing National Action list are then discussed.

14.1.1 Basis, definition and practical application of baseline concentrations for metals in sediment

Determining whether sediment is contaminated by some chemicals is easy since these only have an anthropogenic origin (e.g. PCBs). These synthetically produced chemicals should simply not be present in the natural environment, and their presence thus provides incontrovertible evidence of contamination. Determining whether sediment is metal contaminated, in contrast, is far more complicated and, in fact, presents a major challenge when it comes to assessing the ecological risk posed by metals in sediment. This is because metals are a ubiquitous, naturally occurring component of sediment. The presence of metals in sediment does not, therefore, imply that contamination has occurred. Metal concentrations in uncontaminated sediment can also vary by orders of magnitude over relatively small spatial scales, depending on the sediments mineralogy, granulometry and organic content amongst other factors (Loring and Rantala 1992, Thomas and Bendell-Young 1999, Kersten and Smedes 2002). High metal concentrations in sediment do not, therefore, automatically imply that contamination has occurred but may simply reflect the natural mineralogical composition of the parent material and granulometry and organic matter content of the sediment. In fact, sediment with a lower metal concentration compared to another sediment sample may be metal contaminated and the latter sample not. As a still further complication, despite input and transport dissimilarities naturally occurring and anthropogenically introduced metals tend to accumulate in the same areas (Hanson et al. 1993).

To meaningfully interpret metal concentrations in sediment, the mineralogical and granulometric factors that influence natural variation in metal concentrations in the sediment must first be compensated for before naturally occurring concentrations can be differentiated from concentrations that have been enhanced through anthropogenic contributions (Kersten and Smedes 2002). This differentiation is commonly accomplished through the procedure of normalisation, which mathematically normalises metal concentrations in sediment to a co-occurring conservative element (the normaliser, sometimes referred to as reference element) that provides a tracer of crustal decomposition (Kersten and

Smedes 2002). Normalisation permits the definition of metal baseline concentration models, which can be used to determine whether test sediment samples are metal contaminated.

Two forms of normalisation can be used, namely primary (hereafter referred to as granulometric) and secondary (hereafter referred to as geochemical) normalisation. Granulometric normalisation involves the isolation of a defined sediment grain size class by sieving, with the aim of reducing the diluent effects of non-metal bearing minerals (principally silica) in coarse-grained sediment. Since grain size as a factor influencing metal concentration variability has theoretically been compensated for, metal concentrations in samples of uncontaminated sediment derived from the same parent material should be similar. Granulometric normalisation is most applicable to the dominant metal bearing clay fraction of sediment ($<2\ \mu\text{m}$; Kersten and Smedes 2002). Metal concentrations in the clay fraction can, however, vary considerably between regions due to the contribution of non-metal bearing components in this fraction (e.g. colloidal matter), and this often obfuscates direct comparison of metal concentrations between sieved sediment samples (Kersten and Smedes 2002). Isolation of the clay fraction is also difficult and time consuming, and hence costly (Birch and Snowdon 2004), and often involves the collection and sieving of large volumes of sediment to obtain sufficient material for laboratory analysis (especially in the high-energy marine environment where the fraction of fine-grained material in the sediment is low). Furthermore, unless extreme care is taken there may be significant leaching of contaminants adsorbed onto sediment during the wet sieving procedure. As a result, 'coarser' fractions of sediment are more commonly isolated and analysed (e.g. $<62.5\ \mu\text{m}$, $<20\ \mu\text{m}$; Sutherland 2000, Birch and Snowdon 2004, Suh and Birch 2005). These fractions may, however, still contain significant concentrations of non-metal bearing quartz (Birch and Taylor 1999, Kersten and Smedes 2002), and/or may have a different mineralogy, with the result that observed variability in metal concentrations often needs to be further compensated through geochemical normalisation (Förstner et al. 1982, Matthai and Birch 2001, Kersten and Smedes 2002).

The basis for geochemical normalisation is that while absolute metal concentrations vary between crustal material from one region to another, the relative proportions of metals within crustal material from a particular region tend to be fairly constant (e.g. Turekian and Wedepohl 1961, Taylor and McLennan 1981, Martin and Whitfield 1983, Wedepohl 1995, Kersten and

Smedes 2002). Since there is relatively little fractionation between metals and aluminosilicates during the weathering of parent material (Schropp and Windom 1988), metal concentrations in sediment tend consequently to reflect the relative proportions of metals in the material from which they are derived. The simplest geochemical normalisation approach is to express the concentration of a metal relative to the co-occurring concentration of a selected normaliser (usually a metal) in a sediment sample and to then divide this by the same ratio in anthropogenically unimpacted geological material. This permits the definition of an enrichment factor, $EF = (M/N)_{\text{sample}} / (M/N)_{\text{baseline}}$, where $(M/N)_{\text{sample}}$ is the metal:normaliser ratio in the sediment sample and $(M/N)_{\text{baseline}}$ is the metal:normaliser ratio in the anthropogenically un-impacted geological material. Although ratios for parent rock or regional sediment are most desirable as the denominator, these data are frequently not available and many workers substitute this with a so-called 'Clarke value' (average concentration of the metal in the earth's crust). Enrichment factors of or below unity are taken as indicating that the metal has a natural source and those greater than unity that the metal is enriched relative to the baseline or global average that was used. Although widely used as a sediment metal enrichment assessment tool, the calculated enrichment factor is dependent on the 'baseline' data used and, consequently, has little value if inappropriate data are used (see Covelli and Fontolan 1997, Rubio et al. 2000, Reimann and de Caritat 2005). In a South African context there is, for example, a gradient in the baseline concentration of chromium in coastal sediment of equivalent granulometry along the south-east and east coasts (see below), and evaluation of data by application of the same Clarke value for chromium fails to detect this anomaly. Rather, this approach provides a very different and erroneous appreciation of chromium enrichment/contamination of sediment. Clarke values, and by implication calculated enrichment factors, do not incorporate the natural variability in baseline concentrations of metals around the average value. As demonstrated below, the baseline concentration of a metal in sediment of a particular granulometry does not comprise a single concentration, but rather a range of concentrations. Unless the upper concentration limit of this range is used as the denominator, metal concentrations may be deemed as enriched (i.e. $EF > 1$) relative to the parent material yet they may fall within the natural concentration range for the area. Some workers have compensated for this by assuming that enrichment occurs at an enrichment factor somewhat above unity, but this is an arbitrary selection and there is often significant variation between

the enrichment factor selected as the limit of natural variability (e.g. 10 by Halstead et al. 2000, 2 by Sutherland 2000).

The relative constancy of the proportions of metals in sediment from a particular region and the usually strong inverse correlation between metal concentrations and sediment granulometry permits the use of an alternate geochemical normalisation approach, wherein relationships between metal concentrations and the co-occurring concentration of a normaliser that provides a conservative tracer of crustal decomposition are modelled through some form of regression analysis (usually simple linear regression but occasionally multiple linear regression). Simple linear regression models defined in this manner are referred to as baseline concentration models, or simply baseline models. By quantifying the variability in metal concentrations around the regression line (e.g. through the definition of prediction limits), the range in variability of baseline concentrations with an associated level of confidence can be defined. These data can then be compared to similarly normalised metal concentrations measured in test sediment samples (Figure 2). Sediment samples with metal concentrations that fall within the baseline model prediction limits are considered to fall within the expected baseline concentration range, while metal concentrations that plot above the model upper prediction limit are considered to be enriched relative to the baseline concentration and may have been enhanced through anthropogenic contributions (see further discussion below).

The use of a metal as a proxy for the natural metal-bearing phases of sediment (i.e. aluminosilicates) requires that the metal meet several assumptions, namely that it: 1. is highly refractory, 2. is structurally combined to one or more of the major metal-bearing phases, 3. co-varies in proportion to the naturally occurring concentrations of the metals of interest, 4. is insensitive to inputs from anthropogenic sources, and 5. is stable and not subject to environmental influences such as reduction/oxidation, adsorption/desorption and other diagenetic processes that may alter sediment concentrations (Luoma 1990). Several metals have been used as normalisers, including aluminium (Schropp et al. 1990, Daskalakis and O'Connor 1995, Cooke and Drury 1998, Weisberg et al. 2000, Roach 2005, Newman and Watling 2007), iron (Daskalakis and O'Connor 1995, Schiff and Weisberg 1999, Tanner et al. 2000, Cobelo-García and Prego 2003, Newman and Watling 2007), lithium (Loring 1990, 1991, Aloupi and Angelidis 2001,

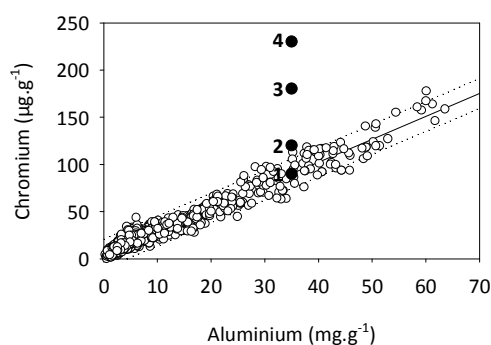


Figure 2. Demonstration of the use of baseline concentration models for interpreting metal concentrations in sediment. The baseline concentration model is for chromium in sediment from KwaZulu-Natal coastal waters. The open symbols (○) represent the chromium concentrations used to define the model, while the numbered black symbols (●) represent four hypothetical scenarios: 1. The concentration falls within the model upper and lower 99% prediction limits (stippled lines) and is therefore interpreted as falling within the expected concentration range; 2, 3 and 4. The concentrations exceed the model upper 99% prediction limit and thus reflect various levels of enrichment that can subjectively be defined from low (2) through to high (4). Situations 3 and 4 would be interpreted as reflecting enrichment through anthropogenic contributions (i.e. contamination) with a high level of confidence, but with a far lower level of confidence in situation 2.

Veinott et al. 2001), rubidium (Grant and Middleton 1990), and caesium and cobalt (Matthai and Birch 2001, Matthai et al. 2002, Roussiez et al. 2005). Aluminium and iron are, however, most frequently used as the normaliser. Of these, aluminium is generally considered to be the best normaliser since it is a major constituent of fine-grained aluminosilicates, with which the bulk of trace metals are associated. Aluminium concentrations consequently usually exhibit a strong inverse correlation to grain size and a positive correlation to co-occurring metal concentrations. Aluminium is also stable and not affected by early diagenetic processes and strong redox effects commonly observed in sediments (Kersten and Smedes 2000), and is highly refractory. Although iron is not as tightly incorporated into the crystal lattice of aluminosilicates as is aluminium, iron oxide coatings, which serve as a host for metals, are usually associated with sediments in definite quantities related to the sediment surface area. The concentration of iron consequently usually also exhibits a strong positive correlation to co-occurring concentrations of metals in sediment (Rule 1986, Daskalakis and O'Connor 1995, Kersten and Smedes 2002). A potential limitation for the use of iron, however, is that it may be highly mobile in anoxic sediments, leading to its enrichment at the sediment surface through the deposition of iron oxides (Finney and Huh 1989) or in deeper sediments as a result of co-precipitation with sulphides (Gobeil et al. 1997),

and may lead to an underestimation of enrichment of other metals when used as the normaliser.

Due to potential differences in the mineralogy of parent material and different weathering processes leading to the formation of sediment between riverine/estuarine catchments, the most appropriate procedure for differentiating between baseline and anthropogenically introduced metal concentrations in sediment for any particular coastal area (e.g. estuary) is to define baseline concentration models and baseline concentrations specific to that system. The definition of river/estuary specific baseline concentration models is, however, expensive due to the requirement for a relatively large sample size and the need for coring to retrieve anthropogenically un-impacted sediment in systems where surface sediment is suspected of being substantially impacted by anthropogenic inputs. River/estuary specific models may also only have applicability to the system for which they were defined (Schropp et al. 1990), further increasing costs through the need to develop models for each system. The development of regional models that are applicable to a variety of sediments over large areas is one way of overcoming these problems and has received considerable attention internationally, particularly in North America (e.g. Schropp and Windom 1988, Windom et al. 1989, Hanson et al. 1993, Daskalakis and O'Connor 1995, Summers et al. 1996, Schiff and Weisberg 1999).

Baseline concentration models for metals comprise a linear regression flanked by upper and lower prediction limits (either 95% or 99%; Figure 2). The prediction limits define the range within which 95% or 99% of metal concentrations should fall if they are at non-contaminant concentrations. Thus, metal concentrations that fall within the baseline model upper and lower prediction limits are considered to fall within the baseline concentration range for sediment from the area of concern. Concentrations that plot above the upper prediction limit represent enrichment (Figure 2). Metal concentrations that plot above the model upper prediction limit do not necessarily imply that the enrichment has an anthropogenic source, but rather that these concentrations are atypical of the data set used to generate the model. Several possible reasons in addition to anthropogenic inputs may lead to a metal concentration exceeding model prediction limits (i.e. enrichment). These include analytical errors, poor model assumptions, the probability that metal concentrations in some samples will naturally exceed the prediction limit (in a normal population, at 99% prediction limit 1 in every 100 concentrations could conceivably exceed the

limit) and natural enrichment not captured by the baseline data set (Schropp et al. 1990, Rae and Allen 1993). Therefore, interpretation of metal enrichment, and ultimately whether this reflects contamination, requires consideration of ancillary factors, including possible (bio)geochemical processes leading to natural enrichment, the absolute difference between measured metal concentrations and model upper prediction limits, the number of metals at a sampling site exceeding model prediction limits, and the location of enriched sediment relative to known or potential anthropogenic metal sources. The greater the difference between a measured metal concentration and model upper prediction limit, the higher the number of metals enriched in sediment from a particular site, and the closer a metal enriched site is to a known or probable anthropogenic source of metals, the greater the likelihood that enriched metal concentrations are enhanced through anthropogenic contributions and thus reflect contamination.

14.1.2 Metal baseline concentrations in sediment from South African coastal waters

Geochemically normalised baseline concentration models and baseline concentrations for a suite of trace metals in sediment from Eastern Cape and KwaZulu-Natal coastal waters have been defined by Newman and Watling (2007) and the CSIR (unpublished data). Similar, but less comprehensive data have also been generated for coastal waters in the St Helena Bay and Saldanha Bay areas of the Western Cape (CSIR unpublished data). The trace metal concentration data have been used to define baseline concentration models and baseline concentrations following the procedures outlined by Newman and Watling (2007). The reader should note that certain trace metals, most notably arsenic, cadmium and mercury, often do not show a strong linear and positive relationship to the normalisers. In these cases, an alternate procedure, based on cumulative frequency plots of concentrations, was used to define baseline concentrations. The baseline concentration models and baseline concentrations are presented in Figure 3.

As is evident from Figure 3, it is evident that the baseline concentrations of most trace metals in granulometrically equivalent sediment are higher in KwaZulu-Natal compared to Eastern Cape coastal waters. There are also differences in the baseline concentrations for some metals (e.g. cobalt, manganese and lead) between sediment from Eastern Cape coastal waters and the St Helena Bay and Saldanha Bay areas of the Western Cape. The notable exceptions of higher metal concentrations in sediment from KwaZulu-Natal coastal waters are arsenic

and especially cadmium, which are slightly and substantially higher respectively in sediment from the St Helena Bay and Saldanha Bay areas of the Western Cape compared to the Eastern Cape and KwaZulu-Natal. It must be reiterated that there is relatively little metal concentration data for sediment from the Western Cape coastal waters, with the result that the baseline concentrations for the St Helena Bay and Saldanha Bay areas might not provide a realistic reflection of baseline concentrations for the entire region. The very high cadmium concentrations measured in sediment from St Helena Bay and Saldanha Bay are particularly interesting, but there is no evidence that these concentrations were enhanced through anthropogenic contributions. This evidence comes from high concentrations in sediment from both bays, and high concentrations in sediment far from anthropogenic sources of metals in each bay. There is furthermore no apparent significant anthropogenic source of cadmium to each bay. Although it was long considered that cadmium has no biological function, cadmium does appear to be actively taken up by marine phytoplankton in certain circumstances, such as when zinc is limiting (Lee and Morel 1995, Lee et al. 1995, Luoma et al. 1998). This behaviour has the effect of concentrating low dissolved concentrations of the metal into small particles. These particles may then either be recycled within the bloom, or exported to the seabed when the phytoplankton senesce. Sediment from Namibian coastal waters has amongst the highest cadmium concentrations in the world, which has been attributed to the high productivity of these waters. Whether the same explanation applies to the St Helena and Saldanha Bay areas of the Western Cape is uncertain, but possible.

14.1.3 Inappropriate sediment quality guidelines of the existing National Action List

As stated previously, the most significant limitation of the existing National Action List is that the guidelines for some metals are inappropriate for application in South African coastal waters, because they specify metal concentrations that are lower than baseline concentrations in some or all regions of the coastline. The principle guidelines in this context are the Action Level for cadmium and the Special Care Level for chromium and nickel. This limitation is clearly illustrated when the guidelines are compared to baseline concentration models and baseline concentrations for KwaZulu-Natal and Eastern Cape coastal waters, and for the St Helena Bay and Saldanha Bay areas of the Western Cape (Figure 4). The Special Care Level for chromium intersects the baseline concentration models for each of these regions, while the Special Care Level for

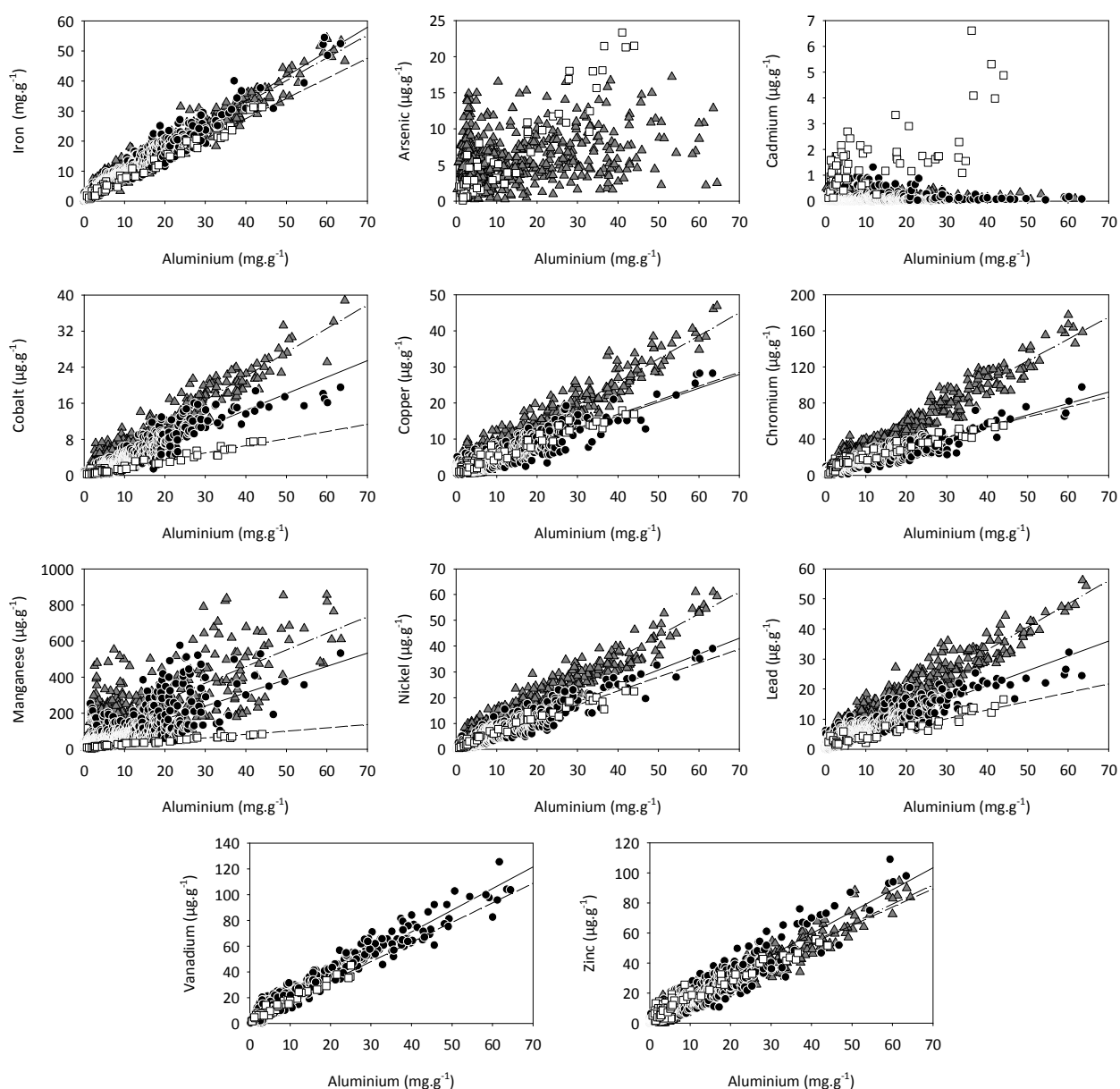


Figure 3. Baseline concentration models and baseline concentration distributions for various major and minor (trace) metals in sediment from coastal waters in different regions of South Africa. ▲ - KwaZulu-Natal, ● - Eastern Cape, □ - St Helena Bay and Saldanha Bay areas of the Western Cape.

nickel intersects the baseline concentration model for KwaZulu-Natal, albeit admittedly only in sediment comprised predominantly of fine-grained material (as interpreted through high aluminium concentrations). The implication is that even though chromium and nickel concentrations measured in sediment from these regions may fall within the baseline concentration range, these concentrations may nevertheless exceed the Special Care Level and thus be interpreted as presenting a potential risk to sediment-dwelling organisms. Strict application of the existing National Action List would identify these sediments as being 'moderately contaminated', and hence requiring further biological assessment to inform decision-making. Clearly, the conclusion and need for further biological assessment in

this situation is unwarranted and would incur unnecessary delays and costs.

The Action Level for cadmium ($1.5 \mu\text{g.g}^{-1}$) is considerably higher than the baseline concentration for sediment from KwaZulu-Natal and Eastern Cape coastal waters, but is considerably lower than the highest concentration ($6.61 \mu\text{g.g}^{-1}$) measured in sediment from the St Helena Bay and Saldanha Bay areas of the Western Cape (Figure 4). The latter concentration is not much lower than the Prohibition Level for cadmium ($10 \mu\text{g.g}^{-1}$; Figure 4). As mentioned previously, whether the metal concentration data for St Helena Bay and Saldanha Bay areas are appropriate to coastal sediment from the broader Western Cape region is uncertain, but nevertheless

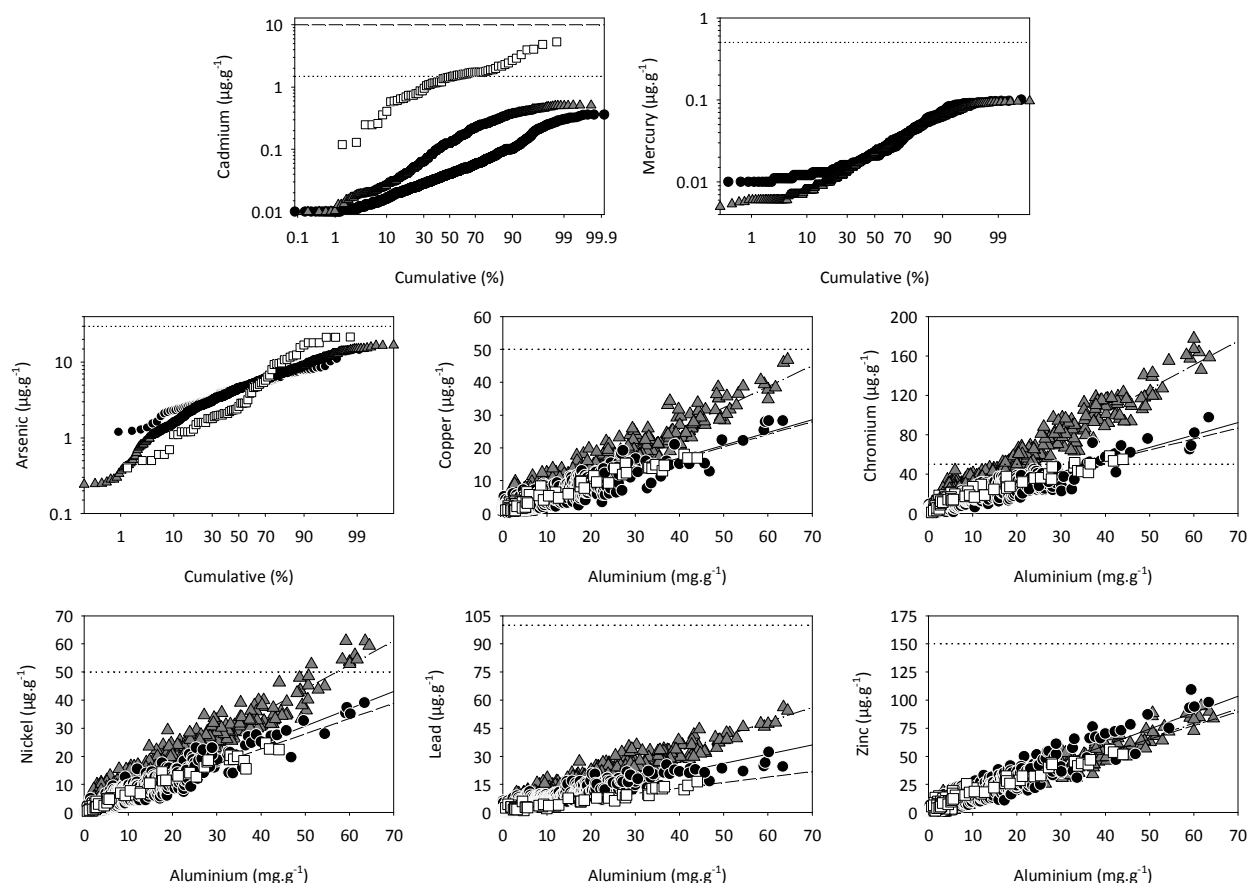


Figure 4. Baseline concentration models and baseline concentration distributions for various major and minor (trace) metals in sediment from coastal waters in different regions of South Africa compared to sediment quality guidelines of the existing National Action List. The stippled horizontal lines represent the Action Level/Special Care Level and the dashed line the Prohibition Level. ▲ - KwaZulu-Natal, ● - Eastern Cape, □ - St Helena Bay and Saldanha Bay areas of the Western Cape.

indicates that at least in these areas the Action Level for cadmium is of limited use for assessing sediment quality.

The Special Care Level for arsenic ($30 \mu\text{g.g}^{-1}$) is only slightly higher than the baseline concentration for the St Helena Bay and Saldanha Bay areas of the Western Cape ($23.36 \mu\text{g.g}^{-1}$, Figure 4). Again, whether this is a significant limitation to the application of this guideline is uncertain due to the paucity of metal concentration data for the broader Western Cape region. The Special Care Level for copper is also only marginally higher than baseline concentrations in sediment from some areas of KwaZulu-Natal, albeit again only in sediment comprised predominantly of fine-grained material.

14.2 Limitation 2: Some guidelines are over-protective and some possibly under-protective

The narrative intent of the sediment quality guidelines and Action Levels of the existing National Action List is uncertain. However, it is assumed that the principle intent is to protect sediment-dwelling organisms. It is further presumed that the Action Level and Special Care

Level have intent comparable to Level I guidelines of other sediment quality guidelines, namely to identify chemical concentrations in sediment below which adverse effects to sediment-dwelling organisms are anticipated to be infrequently observed. Considering the previous discussion on the inappropriateness of some sediment quality guidelines of the existing National Action List for application in South African coastal waters, it is apparent that although the Level I guidelines for some metals probably meet this narrative intent this is due to the guidelines being over-protective (i.e. too low). It is important to note that while concentrations specified by the Action Level and Special Care Level are within the range of Level I guidelines derived for application in many jurisdictions/countries, this similarity does not imply that the guidelines are appropriate. Rather, it implies that the sediment quality guidelines derived for application in many jurisdictions/countries are inappropriate for application in South African coastal waters.

The Prohibition Level for some metals is high and might be under-protective. Based on comparison of the Prohibition Level to Level II guidelines for other

jurisdictions/countries, there is reason to suspect that metal concentrations falling between the Action Level/Special Care Level and the Prohibition Level might frequently be associated with adverse effects to sediment-dwelling organisms. This said, if the intent of the Prohibition Level is to identify chemical concentrations in sediment that are likely to almost always be associated with adverse effects to sediment-dwelling organisms, which seems reasonable considering the guideline terminology and which is also consistent with the London Convention and London Protocol intent for the upper Action Level, then the presumption of under-protectiveness might not be relevant.

14.3 Limitation 3: Additive effects

The existing National Action List provides summative guidelines for Annex I and Annex II metals that, if exceeded, necessitate further biological assessment before a decision is made on the suitability of the material for unconfined, openwater disposal. For Annex I metals, the Action Level and Prohibition Level are $1 \mu\text{g.g}^{-1}$ and $5 \mu\text{g.g}^{-1}$ respectively, while for Annex II metals the Special Care Level and Prohibition Level are $50 \mu\text{g.g}^{-1}$ and $500 \mu\text{g.g}^{-1}$ respectively. Although additive effects of certain contaminants has been identified through laboratory experimentation (e.g. Plesha et al. 1988, Swartz et al. 1988, Kraak et al. 1994, Norwood et al. 2003), the toxicity of metal mixtures cannot reliably be predicted based on the toxicity of individual metals. Antagonistic interactions have also been reported (e.g. selenium has an antagonistic effect on mercury toxicity; Oakden et al. 1984, Nuutinen and Kukkonen 1998, Chen et al. 2001).

Although contaminant interactions do require consideration, the illogicality of the existing National Action List approach, notwithstanding the fact that baseline concentrations for some metals in sediment exceed $50 \mu\text{g.g}^{-1}$ (see previous discussion), is that the Special Care Level for all but one of the Annex II metals is set at or exceeds $50 \mu\text{g.g}^{-1}$. In other words, if the concentration of only a single Annex II metal marginally exceeds the Special Care Level, then it by implication also exceeds the summative guideline.

14.4 Limitation 4: Confusing terminology

Terminology used in the existing National Action List is often confusing, as illustrated by the following examples.

The Action Levels of an Action List can comprise one or more assessment criteria. These can include, amongst others, the volume of dredged material, concentrations of chemicals in sediment, and defined biological effects.

However, the existing National Action List uses the term Action Level to describe the Level I sediment quality guideline for cadmium and mercury. The modified National Action List in fact uses this term for the Level I guidelines for all metals. The confusion created in this context is self-evident, unless the intent was for the assessment criteria of the Action Levels to comprise only sediment quality guidelines. This is not the case, however, since the original and modified National Action Lists make specific and repeated reference to biological assessment as a decision-making assessment criterion.

The Level I guideline for Annex I metals is referred to as the Action Level and for Annex II metals as the Special Care Level in the original National Action List document. Adopting separate terms for the Level I guideline for Annex I and Annex II metals is confusing and creates the impression that Annex I metals are more worthy of management 'action' in the case of guideline exceedance compared to Annex II metals, which evidently only require Special Care. While this may indeed be a valid assumption considering that cadmium and mercury are important contaminants of sediment and mercury can biomagnify, the author suggests that a single term be used to describe the Level I guideline for Annex I and Annex II metals.

The Level II guideline for Annex I and Annex II metals is referred to as the Prohibition Level. This immediately invokes the assumption that, when exceeded, dredging and/or unconfined, openwater disposal of dredged material is prohibited. This is consistent with the London Convention and London Protocol intent for the upper Action Level. However, the existing National Action List document states that 'If either of the Prohibition Levels for the Annex I metals is exceeded, or if the Prohibition Level of two or more of the Annex II metals is exceeded, dumping will not be allowed'. In other words, the prohibition of unconfined, openwater disposal of dredged material does not necessarily depend on exceedance of a Prohibition Level but on a number of different metals exceeding a Prohibition Level. The term prohibition is sufficiently prescriptive that a single exceedance of a Prohibition Level should provide grounds for prohibiting the unconfined, openwater disposal of dredged sediment.

The modified National Action List provides criteria for determining whether unconfined, openwater disposal of dredged material should be prohibited (Table 14). Decision criterion (e) states that 'If the combined level of Annex II metals is $>1000 \mu\text{g.g}^{-1}$, then biological testing is required'. Decision criterion (f) states that 'If either of the Prohibition Levels for the Annex I metals is

exceeded, or if the Prohibition Level of two or more of the Annex II metals is exceeded, dumping will not be allowed'. A situation could arise, however, where the concentration of arsenic and the concentration of either chromium, copper, lead and nickel in a sediment sample marginally exceed the Prohibition Level but the combined concentration is still below $1000 \mu\text{g.g}^{-1}$. In this situation, unconfined, openwater disposal of the dredged material should immediately be prohibited because concentrations of two Annex II metals exceed the Prohibition Level. However, in another situation the combined concentration of Annex II metals may exceed $>1000 \mu\text{g.g}^{-1}$ and presumably pose similar risks as the abovementioned situation, but now there is no prohibition until biological testing has confirmed whether or not there is risk associated with unconfined, openwater disposal of the material. Clearly, these anomalous situations need to be avoided to the greatest extent possible in the revised National Action List.

The existing National Action List document repeatedly states that sediment is considered uncontaminated ('trace or insignificant contamination') if metal concentrations fall below the Action Level or Special Care Level, moderately contaminated if metal concentrations fall between these and the Prohibition Level, and highly contaminated if concentrations exceed the Prohibition Level. Apart from the fact that these conclusions for some metals are immediately invalidated by the guidelines being lower than baseline concentrations for some metals, sediment quality guidelines/Action Levels are not intended to define when contamination has occurred unless these are based on geochemical considerations and are calibrated to sediment grain size variability. This is not the case for the sediment quality guidelines of the existing National Action List. This is notwithstanding the fact that when a Level I guideline is exceeded contamination should already have occurred, the degree of contamination determined by the guidelines conservativeness.

Lastly, the term 'heavy metal' should not be used since it has no scientific basis. Although this term is often used in the context of so-called toxic metals and is understood by the general public as such, there is in fact nothing 'heavy' about many of the metals that are frequently associated with adverse biological effects. Use of the term as a synonym for toxic or 'bad' metals is unjustified given that toxicity is a function of the chemical properties of the element/compound and the biological properties of the organism at risk. In fact, all metals/elements are toxic in certain forms and in sufficiently high doses, and not only the 'heavy metals'. A more apt term is trace metals.

14.5 Limitation 5: Biological testing

The existing National Action List document makes explicit and repeated reference to biological assessment as a criterion for decision-making. The document does not, however, provide guidance on the type of biological assessment required under different circumstances (e.g. acute versus chronic toxicity testing, toxicity testing versus bioaccumulation testing), nor how the results of biological assessment should be used for decision-making. This limitation must be addressed in the revised National Action List, either through the explicit inclusion of biological assessment tools and guidance on their use for decision-making, or through the removal of biological assessment as a line of evidence until such time as biological assessment tools are sufficiently advanced in South Africa for their inclusion.

14.6 Limitation 6: Too much emphasis on metals

Although the existing National Action List provides guidelines for various potential contaminants in sediment, the overwhelming emphasis is on metals. In fact, decisions on the suitability of dredged material for unconfined, openwater disposal in South Africa are based almost exclusively on metal concentrations.

Other chemicals could conceivably be adversely affecting sediment-dwelling organisms when the concentrations of all metals are too low to cause such effects. Organic contaminants (e.g. polycyclic aromatic hydrocarbons, polychlorinated biphenyls) are of particular concern in this regard, since many of these chemicals have a high bioaccumulation and biomagnification potential (e.g. Sorenson et al. 2007) and are known or strongly suspected endocrine disruptors and carcinogens. In contrast, very few metals can be bioaccumulated to any significant degree and even fewer can be biomagnified.

The emphasis on metals may thus lead to a situation where sediment that is significantly contaminated with organic contaminants is considered suitable for unconfined, openwater disposal because all metal concentrations are below the Level I guideline.

15. Implications of the London Convention and London Protocol for revision of the National Action List

As mentioned elsewhere in this report, the Waste Assessment Guidelines of the London Protocol state 'Each Contracting Party shall develop a national Action List to provide a mechanism for screening candidate wastes and their constituents on the basis of their

potential effects on human health and the marine environment', and that 'An Action List shall specify an upper level and may also specify a lower level'. The Waste Assessment Guidelines are clear on the intent of the upper and lower Action Levels, as follows: 'The Upper Level should be set so as to avoid acute or chronic effects on human health or on sensitive marine organisms representative of the marine ecosystem. The upper Action Level is intended to provide a definitive decision, namely prohibition of the waste under consideration for marine disposal unless it is further managed. Application of an Action List should result in three possible categories of waste:

1. wastes which contain specified substances, or which cause biological responses, exceeding the relevant upper level shall not be dumped, unless made acceptable for dumping through the use of management techniques or processes;
2. wastes which contain specified substances, or which cause biological responses, below the relevant lower levels should be considered to be of little environmental concern in relation to dumping; and
3. wastes which contain specified substances, or which cause biological responses, below the upper level but above the lower level require more detailed assessment before their suitability for dumping can be determined'.

Given that the intent of upper and lower Action Levels is different, it is not essential that the same approach or the same narrative intent be used to formulate both Action Levels. The requirement that the upper Action Level should be 'set so as to avoid acute or chronic effects on human health or on sensitive marine organisms representative of the marine ecosystem' is prescriptive enough that the Action List should have a biologically protective function, for both acute and/or chronic effects. Furthermore, the stipulation that 'acute or chronic effects' should be avoided has implications for the revised National Action List inasmuch that these effects have different implications for the definition of guidelines (i.e. level of protection required) and the type of biological testing that may be required.

The Waste Assessment Guidelines state further that 'In selecting substances for consideration in an Action List, priority shall be given to toxic, persistent and bio-accumulative substances from anthropogenic sources (e.g., cadmium, mercury, organohalogens, petroleum hydrocarbons and, whenever relevant, arsenic, lead, copper, zinc, beryllium, chromium, nickel and vanadium, organosilicon compounds, cyanides, fluorides and pesticides or their by-products other than

organohalogens)'. Although the list is not prescriptive, inclusion of persistent and especially bioaccumulative substances has implications for the National Action List in that, as discussed elsewhere in this report, it places an overwhelming emphasis on metals for decision-making. Although the DEA called for a review and updating of only the metals included in the Action List, the process followed in this context will have implications for the review of other potential chemical contaminants included in the action List.

16. Implications of the Integrated Coastal Management Act for revision of the National Action List

Section 73 of the ICM Act states that:

- (3) The Minister must progressively and subject to available resources, develop a national action list to provide a mechanism for screening waste and other material on the basis of their potential effect on human health and the marine environment.
- (4) The national action list must—
 - (c) be developed in accordance with the Waste Assessment Guidelines set out in Schedule 2; and
 - (d) contain the prescribed information.

As mentioned previously, Schedule 2 of the ICM Act is a verbatim extract of Annex 2 of the London Protocol. Addressing the requirements of the Waste Assessment Guidelines will thus address the requirements of the London Protocol and the ICM Act. As intimated in the condition that 'The Minister... progressively and subject to available resources...." provides the option for a progressive formulation/revision of the National Action List, as dictated by available resources.

17. Factors to consider and options for revising the sediment quality guidelines of the existing National Action List

This section discusses various factors that need to be considered and provides options for the revision of the sediment quality guideline component of the existing National Action List. Although the predominant focus is on metals, similar factors and options apply to other chemicals that might be considered for inclusion in the revised National Action List. Stakeholders should note that when formulating a response to several of Key Issues highlighted in the text of this section, it will be necessary to first consider information in all subsections and in some cases also in subsequent sections of the report.

17.1 Should sediment quality guidelines be defined for different management objectives?

As far as the author could determine, sediment management objectives have not been identified for any aquatic ecosystem in South Africa. The importance of sediment management objectives is that these define the level of protection required and identify the indicators needed to measure attainment. There is growing recognition that while the same level of environmental quality for all aquatic ecosystems is desirable, many aquatic ecosystems sustain economically important activities that impair water and sediment quality and associated organisms. For example, ports are essential to the trade of goods between South Africa and the rest of the world, but there are inevitably higher concentrations of contaminants in water and sediment from ports compared to similarly sheltered habitats due to the nature of port activities. These impacts will occur irrespective of management interventions to reduce contaminant input. A simple example is the release of contaminants from antifouling coatings on the hulls of ocean going vessels. The coatings reduce marine organisms (e.g. mussels, barnacles) from settling on (fouling) vessel hulls. This is important as fouling increases drag and consequently has cost implications, since more fuel is required to travel from one point to another. Antifouling coatings achieve their effect by the slow release (leaching) of active biocides incorporated into the coating. Copper, for example, is an important biocide in coatings. Release of the biocide proceeds unabated, including in port environments. These ‘unavoidable’ or difficult (read costly) to manage sources of contaminants to some environments has led some jurisdictions/countries to define environmental quality objectives and targets for aquatic ecosystems according to their designated uses. Environmental quality objectives and targets for a commercially important port are thus set differently to those for an estuary within a rural setting, or in a designated conservation area. It should be noted, however, that defining different levels of protection for systems does not imply tacit approval for unregulated environmental degradation.

A revised National Action List that comprises two Action Levels will inherently provide two levels of protection. The critical issue, however, is whether either or both of the guidelines should provide an equivalent level of protection as that afforded in, for example, an estuary situated distant from or relatively un-impacted by anthropogenic disturbances (including contaminant inputs). If so, then the same sediment quality guidelines

can be designated for all coastal aquatic ecosystems within particular regions of South Africa. This would have the benefit of simplifying assessment. If the same level of protection is not considered attainable, however, then the implication is that the sediment quality guidelines of the revised National Action List will differ from those used, for example, for estuaries in rural settings.

Key Issue:

1. Should the level of protection afforded to sediment-dwelling organisms differ between coastal ecosystems designated for different uses (e.g. ports and dredged spoil disposal grounds versus estuaries in rural settings with a predominantly recreational and subsistence use)?

17.2 Narrative intent of sediment quality guidelines

Following on from the above discussion, the first factor that needs to be resolved in the formulation of sediment quality guidelines for the revised National Action List is the narrative intent of the guidelines. The author has assumed that the sediment quality guidelines of the revised National Action List will, as is the situation for the existing National Action List, comprise two guidelines, that is, Level I and Level II guidelines. The narrative intent is, in essence, a statement on what the guidelines are intended to protect and the level of protection afforded. For example:

Level I and Level II sediment quality guidelines are designated for the protection of sediment-dwelling organisms, such that:

- The Level I guideline is intended to identify contaminant concentrations at or below which adverse effects to sediment-dwelling organisms are unlikely to be observed.
- The Level II guideline is intended to identify contaminant concentrations above which adverse effects to sediment-dwelling organisms are likely to be frequently observed.

The terms ‘unlikely’ and ‘frequently’ in this example are defined through the sediment quality guideline derivation approach, but ultimately are defined by the level of protection that stakeholders consider necessary or acceptable. For example, the Effects Range sediment quality guidelines derived by Long et al. (1995) consider the 10th percentile of adverse effects to be consistent with the narrative intent of ‘unlikely to be observed’ and the 50th percentile as consistent with ‘likely to be frequently observed’. Another jurisdiction/country

/worker might, however, decide that the 15th and 75th percentiles are consistent with 'unlikely to be observed' and 'likely to be frequently observed' respectively.

The Waste Assessment Guidelines of the London Protocol provide guidance in the context of the narrative intent of the Action Levels of an Action List in that it states 'An Action List shall specify an upper level and may also specify a lower level. The Upper Level should be set so as to avoid acute or chronic effects on human health or on sensitive marine organisms representative of the marine ecosystem'. In other words, the narrative intent of a Level II sediment quality guideline in the context of dredged material disposal is for the protection of sediment-dwelling organisms against acute or chronic toxic effects. However, it is uncertain whether acute or chronic effects should be avoided, or both. This is important since chronic effects can occur at concentrations somewhat below those at which acute effects occur. Of course, the sediment quality guidelines could be derived to be protective of both of acute and chronic effects. The Waste Assessment Guidelines do not provide guidance on the narrative intent for the Level I guideline.

There thus appears to be leeway in considering acute or chronic effects, or both. However, the proportion of sediment-dwelling organisms that should be protected is not prescribed. For example, a Level II guideline might be set conservatively, such that the probability for acute and/or chronic effects to even sensitive sediment-dwelling organisms is low, or it could be set leniently such that there is a high probability that acute and/or chronic effects would have manifested in sensitive receptors at a lower concentration than specified by the guideline. The decision in this context will revolve around the need to balance the incidence of false positives and false negatives, and as discussed later also socio-economic factors.

As stated elsewhere in this report, there are virtually no matching sediment chemistry and biological effect data for sediment from South African coastal waters. Consequently, it will be impossible to determine whether the sediment quality guidelines of the revised National Action List provide a level of protection that is consistent with their narrative intent. The metal baseline concentrations discussed previously provide some guidance in defining appropriate Level I guidelines, by identifying guidelines that are clearly not suitable (discussed below). But they are of no use in determining when adverse effects to sediment-dwelling organisms are likely to manifest at higher concentrations.

Nevertheless, a decision on whether the guidelines should err on the side of caution (i.e. protectiveness) will need to be made. The nearer that metal (and other chemical) concentrations specified by the Level I and Level II guidelines of the revised National Action List are to their baseline concentrations in sediment the higher the level of protection afforded to sediment-dwelling organisms. If the sediment quality guidelines and other decision criteria of the revised National Action List are so protective that further assessment is frequently required, or in the extreme situation unconfined, openwater disposal of dredged sediment is frequently prohibited, then this will have obvious and significant socio-economic implications. In fact, whether South Africa is even in a position to prohibit the unconfined, openwater disposal of dredged sediment is debatable since we do not presently have the technical expertise and facilities and the financial resources to consider alternate disposal (e.g. confined disposal facilities) or sediment remedial options, at least not for large volumes of sediment. These realities are juxtaposed with the responsibility to protect South Africa's natural resources and the government's obligations to international conventions that not only seek to reduce our impact on coastal ecosystems but to do so through the precautionary principle. In the absence of data against which to make a decision on the level of protectiveness afforded by sediment quality guidelines, the precautionary principle implies that sediment quality guidelines should err on the side of conservativeness, that is, be over- rather than under-protective. As stated by Apitz et al. (2004), contaminated sediment management is complex and multivariate, involving a careful balance of science, politics, and economics. As is true for most complex issues, a single correct way to address a problem does not exist, but depends on the ecological, political, and economic goals of all interested parties. Any decision on the formulation of the sediment quality guidelines and other decision-making criteria of the revised National Action List will need to consider these competing demands and realities.

Key issue:

2. Should the sediment quality guidelines of the revised National Action List comprise more than two guidelines, to take into account different management objectives?

17.3 Derive sediment quality guidelines using empirical data

Ideally, the sediment quality guidelines of the revised National Action List should be derived from matching

sediment chemistry, toxicity and benthic invertebrate community data. However, as mentioned above there are virtually no such data for coastal waters in South Africa. Therefore, derivation of empirically-based sediment quality guidelines for inclusion in the revised National Action List is not an option at this time.

The lack of this data means that the generation of matching sediment chemistry, toxicity and benthic invertebrate community data assumes a high level of priority. Strategies for the generation of such data thus need to be developed and implemented.

17.4 Derive sediment quality guidelines using the Sediment Background approach

The Sediment Background approach is the most intuitive and straightforward approach for deriving sediment quality guidelines for the revised National Action List considering that there is a relatively good understanding of the baseline concentrations of toxicologically relevant metals in sediment from several regions of the South African coastline. For those regions of the coastline where data are not available or are insufficient for the definition of baseline concentrations with a high level of confidence, the data could be generated within a reasonable period and at a reasonable cost. The Sediment Background approach has been used to derive sediment quality guidelines for Action Lists in several countries, especially Level I guidelines but in some cases also Level II guidelines (see section 12 above).

The Sediment Background approach does, however, have a number of important limitations. Firstly, metal baseline concentrations are often site-specific, because of the often significant differences in the provenance and mineralogy of sediment between regions. It is for this reason that baseline concentrations for metals should ideally be defined at the system specific level (e.g. for individual estuary). Section 14.1.2 provided evidence for significant differences in the baseline concentrations of several metals between regions of the South African coastline, which presumably reflect differences in the geology of the sediment parent material. While baseline concentrations of metals in sediment have been defined for Eastern Cape and KwaZulu-Natal coastal waters, it is not certain whether these concentrations apply to all coastal areas within the regions. This is because an often significant proportion of the metal concentrations used to define the baseline models were extracted from sediment collected from relatively few locations. Recent evidence has shown, for example, that the KwaZulu-Natal baseline concentration

models for some metals in do not apply to sediment from estuaries in the Durban area. Data for estuaries in this area was intentionally not used for baseline model definition, because of the high probability for metal contamination of the sediment.

Baseline concentrations have not been defined for some regions of the South African coastline, but these generally fall outside of the areas where dredging is frequently performed. Although metal baseline concentrations for sediment from the St Helena Bay and Saldanha Bay areas of the Western Cape are presented in this report, there is insufficient data from anthropogenically un-impacted sites in this region for the definition of scientifically defensible baseline concentrations at this time. It is impossible, therefore, to determine whether the (evidently naturally) high cadmium concentrations measured in sediment from some areas of the Western Cape apply to all areas of this region.

A second limitation is that while baseline metal concentrations have been defined for certain regions of the coastline, the metal concentration data were not always generated using a consistent analytical procedure. Thus, some studies used a total digestion procedure to extract metal concentrations from sediment, while others used near-total digestion. These methods extract different proportions of metals from sediment and probably account for some of the differences observed between studies. For example, while there is generally good agreement between baseline concentrations for metals defined by Newman and Watling (2007) for Eastern Cape coastal waters, there are differences to baseline concentrations for some metals recently defined by the CSIR for the Algoa Bay area of this coastline (unpublished data). The reason for these differences is uncertain, but a difference in method extraction efficiency has been identified as a possible cause. A further limitation is that a consistent suite of metals have not been consistently targeted, making it difficult to compare baseline concentrations for some metals between regions.

A further limitation of the Sediment Background approach is that it is impossible to define a single baseline concentration for most metals. As discussed elsewhere in this report, the concentrations of most metals in sediment are inversely correlated to sediment grain size. Using the baseline concentration for chromium in sediment from KwaZulu-Natal coastal waters as an example (Figure 5), the baseline concentration at co-occurring aluminium concentrations (which are a proxy for grain size) at 10, 20, 30 mg.g⁻¹, and

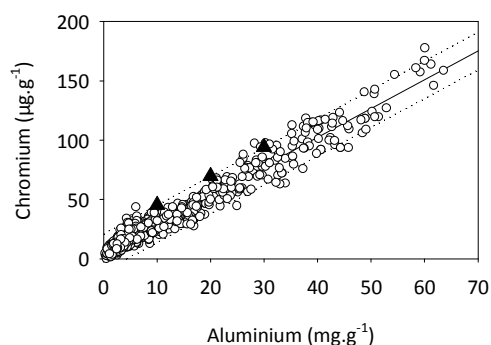


Figure 5. An example of why a single baseline concentration cannot be defined for many trace metals in sediment. The solid, triangular symbols (▲) represent concentrations of chromium at the upper 99% prediction limit of the baseline concentration model for KwaZulu-Natal coastal waters at three corresponding aluminium concentrations, namely 10, 20 and 30 mg.g^{-1} . The change in chromium concentrations with sediment grain size (aluminium as proxy) means that a single baseline concentration for chromium cannot be defined.

so forth is different. In this case, the respective chromium concentrations at the baseline model upper 99% prediction limit are 45.53, 69.80 and 94.07 $\mu\text{g.g}^{-1}$. In essence, therefore, baseline concentrations need to be defined for specific 'categories' of sediment, as defined by its grain size (in this case the mud fraction would be the most reliable proxy). The principle problems with defining baselines for 'categories' of sediment is that it will result in a complex, cumbersome assessment process, since each 'category' would need to have an Action Level/s defined. A further problem is that the relationship between the grain size and metal content of sediment from many regions of the South African coastline has not been adequately defined and this will in turn make it difficult to define Action Levels. The highest aluminium concentration measured in sediment from an area can be used to define sediment quality guidelines using the Sediment Background approach, but since the metal concentrations used in the baseline models were not collected in a sufficiently random basis there will be some inconsistency between regions.

The most significant and intractable limitation of the Sediment Background approach, however, is that the derived sediment quality guidelines do not have an (at least not intentional) toxicological basis. The reason is that chemicals do not all begin to exert toxicity at the same proportional level of exceedance of a baseline concentration, either within or between regions. We can, therefore, only guess at what level of exceedance adverse effects to sediment-dwelling organisms might begin to manifest, and when they would become serious and so forth. This can be demonstrated by comparing the ratio between Level I and Level II guidelines (Table 15).

Using copper and zinc as an example, the Level II guideline for copper in the Long et al. (1995) sediment quality guidelines is about 8 times higher than the Level I guideline, but the Level II guideline for zinc is about 2.7 times higher than the Level I guideline. This example demonstrates that adverse biological effects do not begin to manifest at the same level of exceedance of a baseline concentration for different metals. The differences in toxicity between chemicals arise because their bioavailability is dependant on numerous factors and processes in sediment, many of which are not properly understood, and which vary between chemicals and sites. Furthermore, some metals are essential elements for the normal physiological functioning of organisms. Defining sediment quality guidelines for metals on the basis of a defined proportional exceedance of a baseline concentration is therefore not defensible from a toxicological perspective.

The derivation of sediment quality guidelines using the Sediment Background approach thus involves consensus on the level of contamination considered 'acceptable' before management intervention is required, and/or on socio-economic or political decisions. Of course, the uncertainty of this approach from a toxicological perspective can be reduced by defining classes of exceedance of the baseline concentration such that an Action List comprises more than two Action Levels (e.g. Class 1 = 1 - 30% exceedance, Class 2 = 31 - 60% exceedance, and so on). Each class could then have associated administrative rules for decision-making.

Despite the limitations of the Sediment Background approach for deriving sediment quality guidelines, the approach has merit and has been used for this purpose by several jurisdictions/countries. Although the author recommends that this approach be considered only in the event that none of the options identified below is deemed suitable by stakeholders for the formulation of

Table 15. Proportional difference between the Level I (ERL – Effects Range Low) and Level II (ERM – Effects Range Median) guidelines of the sediment quality guidelines derived by Long et al. (1995).

Metal	ERL	ERM	Proportional difference
Arsenic	8.2	70	8.54
Cadmium	1.2	9.6	8.00
Chromium	81	370	4.57
Copper	34	270	7.94
Mercury	0.15	0.71	4.73
Nickel	20.9	51.6	2.47
Lead	46.7	218	4.67
Zinc	150	410	2.73

sediment quality guidelines for the revised National Action List, a decision on the suitability of this approach must be reached through consensus. One option is to use the Sediment Background approach for the definition of a warning level guideline, which could define concentrations of metals lower than the Level I guideline and have the intent of identifying (warning of) incipient contamination. If so, then a decision needs to be made on the proportional exceedance of baseline concentrations that will be used to define the warning level, and whether any administrative rules (e.g. further assessment) should be attached to the guideline. The concept of a warning level and an examples is provided in a subsequent section of this report.

Irrespective of whether the Sediment Background approach is used to formulate sediment quality guidelines for the revised National Action List, baseline metal concentrations provide a powerful, scientifically defensible tool for identifying whether sediment is metal contaminated. As such baseline metal concentrations provide a line of evidence that can and should be incorporated into the revised National Action List and Dredging Assessment Framework.

Key Issue:

3. Should the Sediment Background approach be considered the preferred approach for deriving sediment quality guidelines for the revised National Action List? If so, what proportional exceedance of the baseline concentrations should be used to define the Level I and Level II guidelines?

Alternately, should the Sediment Background approach be used to define a warning level? If so, what proportional exceedance of the baseline concentrations should be used to define the warning level? If a warning level is defined, should administrative rules be attached to the warning level, or should it be used merely as a basis for warning of incipient contamination?

17.5 Define sediment quality guidelines using the Equilibrium Partitioning approach

As discussed previously, the Equilibrium Partitioning approach has been used to derive sediment quality guidelines in some jurisdictions/countries. The primary advantage of this approach is that, unlike other approaches for deriving sediment quality guidelines, it accounts for bioavailability. The most significant limitation of the approach is that it is suitable for

deriving guidelines for relatively few chemicals that are of interest in sediment quality assessments. For example, this approach is suitable for deriving sediment quality guidelines for five of the 9 - 11 trace metals that are typically of interest in sediment quality assessments. If this approach is adopted, then sediment quality guidelines for some metals will need to be derived using an alternate approach.

Although the author recommends that this approach be considered only in the event that none of the options identified above and below is deemed suitable for formulating sediment quality guidelines for the revised National Action List, the decision on the suitability of this approach must be reached through consensus. The Simultaneously Extracted Metal/Acid Volatile Sulphide approach, which is based on equilibrium partitioning, has merit as a further assessment tool and is discussed in a subsequent section of this report.

Key Issue:

4. Should the Equilibrium Partitioning approach be considered the preferred approach for deriving sediment quality guidelines for the revised National Action List?

17.6 Adopt sediment quality guidelines from another jurisdiction/country

In the absence of (sufficient) matching sediment chemistry, toxicity and benthic invertebrate community data for the derivation of empirically-based sediment quality guidelines, several jurisdictions/countries (e.g. Australia, Hong Kong, New Zealand) have adopted (with minor modifications to take into account local conditions) sediment quality guidelines derived for application in another jurisdiction/country. Other jurisdictions/countries are also considering this approach (e.g. France, see Babut et al. 2003). Although the adoption of sediment quality guidelines from another jurisdiction/country is intuitively appealing, since it eliminates the effort and costs of generating empirical data, several issues require careful consideration.

The narrative intent of sediment quality guidelines considered for adoption must be consistent with the narrative intent identified by the adoptee jurisdiction/country. Of course, the adoptee jurisdiction/country can simply adopt sediment quality guidelines that are consistent with a pre-defined narrative intent and sediment management needs, or frame the narrative intent as per the adopted guidelines. Many jurisdictions/countries use the Level II guideline of Action Lists as the basis for identifying dredged material

that is unsuitable for unconfined, openwater disposal. There is little point then in adopting a Level II sediment quality guideline that has the narrative intent of identifying contaminant concentrations in sediment above which adverse effects to sediment-dwelling organisms are likely to be frequently observed when the guideline is intended for definitive use. Depending on the definition of 'frequently observed' in the sediment quality guideline derivation process (e.g. above 50th percentile of adverse effects), there is a strong possibility that unconfined, openwater disposal of sediment could be prohibited because contaminant concentrations exceed the guideline but the sediment may in fact not have been toxic. As discussed previously, the narrative intent of the sediment quality guidelines of the revised National Action List must still be defined. It is not possible therefore to identify candidate sediment quality guidelines that are consistent with the narrative intent, since the narrative intent is unknown. Consequently, a subsequent section of this report identifies candidate sediment quality guidelines for the revised National Action List regardless of their narrative intent.

An inherent assumption when adopting sediment quality guidelines from another jurisdiction/country is that sediment in both areas has a similar mineralogy. In other words, baseline concentrations of metals are similar in both areas. This is usually not the situation, but is often not considered. As discussed elsewhere in this report, the mineralogy of sediment parent material along the South African coastline is variable, with the result that baseline concentrations of some metals differ between regions. If there are differences in the mineralogy of sediment parent material between the jurisdictions/countries, then the adopted sediment quality guidelines will not afford the level of protection to sediment-dwelling organisms as per their derivation intent. Due to the importance of baseline concentrations when considering sediment quality guidelines for adoption, a subsequent section of this report screens candidate sediment quality guidelines for inclusion in the revised National Action List against baseline concentrations for metals in sediment from South African coastal waters.

A further, inherent assumption (and linked to the assumption discussed above) is that sediment-dwelling organisms in the adoptee jurisdiction/country exhibit a similar sensitivity and tolerance to contaminants compared to the jurisdiction/country for which the sediment quality guidelines were derived. This is not necessarily the case. Fauna and flora have evolved in the presence of metals and exhibit geographic distributions that reflect variable requirements for and tolerance to

certain metals. There is evidence that water and sediment quality guidelines cannot simply be extrapolated from one (climatic) region to another and be expected to provide the same level of protection (e.g. Chapman and Mann 1999, Chapman et al. 1999, Chapman and Riddle 2005, Chapman et al. 2006, Kwok et al. 2007). Kwok et al. (2007), for example, reported differences in the sensitivities of tropical and temperate freshwater animals to a number of chemicals. Tropical species were found to be more sensitive to ammonia, phenol, and chlorpyrifos, whereas temperate species were generally more sensitive to metals. The significance is that all empirically-based sediment quality guidelines that can be considered as candidates for the revised National Action List have been derived for application in North American and European coastal waters, which are by and large temperate (warm or cool temperate) and in some cases border on polar in nature. While these sediment quality guidelines may be appropriate for cooler waters of the western, southern and possibly even south-eastern coasts of South Africa, they may not be appropriate to sub-tropical waters along the east coast.

Jurisdictions/countries that have adopted sediment quality guidelines have done so for the reason that there is insufficient data for the derivation of empirically-based sediment quality guidelines. As mentioned previously, this leads to the problem that because there are insufficient data to derive empirically-based sediment quality guidelines there are also insufficient data for the adoptee jurisdiction/country to evaluate whether the sediment quality guidelines are performing per their narrative intent. Several studies have shown that the performance (predictive ability) of sediment quality guidelines is variable when they are applied in areas for which they were not derived (O'Connor et al. 1998, Long et al. 1998, MacDonald et al. 2000, Field et al. 2002, Vidal and Bay 2005). This said, other studies have shown that sediment quality guidelines adopted from a jurisdiction/country appear to broadly perform according to their narrative intent in the adoptee jurisdiction/country, even though they may be separated by many thousands of kilometres (e.g. McCready et al. 2006).

Although there are numerous limitations to adopting sediment quality guidelines from another jurisdiction/country for the revised National Action List, this is nevertheless a feasible approach, not least because guidelines for metals and other chemicals can be adopted at the same time. This option is explored in more detail in a subsequent section of this report.

Key Issue:

- Should the adoption of sediment quality guidelines from another jurisdiction/country be considered the preferred approach for formulating sediment quality guidelines for the revised National Action List?

17.7 Remove or replace only the inappropriate guidelines of the existing National Action List

As discussed previously, certain Level I guidelines for metals in the existing National Action List are inappropriate for application to some or all regions of the South African coastline, because the guideline concentrations are lower than baseline concentrations. One option is to replace entirely the Level I guidelines of the existing National Action List, or to replace/modify only the inappropriate Level I guidelines. The inappropriate guidelines could, for example, be derived using the Sediment Background approach, or could be replaced by a Level I guideline from another jurisdiction/country. Similarly approaches could be used to replace all Level I guidelines. None of the Level II guidelines are inappropriate for application in South African coastal waters, but this does not imply that they provide an appropriate level of protection and/or are consistent with a narrative intent that might be identified for the guidelines/Action Levels of the revised National Action List.

This option is no more or less defensible than the options identified above and below. The main implication is that the Level I guidelines for each metal in the revised National Action List will not have the same derivation approach if only inappropriate Level I guidelines are replaced.

Key Issue:

- Should only inappropriate Level I guidelines of the existing National Action List be replaced, or all Level I guidelines? If so, what approach should be used to derive/formulate the guidelines?

17.8 Formulate sediment quality guidelines using a hybrid approach

The sediment quality guidelines of the revised National Action List could be formulated using a hybrid approach, wherein the Level I guidelines are derived using the Sediment Background approach and the Level II guidelines are adopted from another jurisdiction/country. This approach has been followed in some jurisdictions/countries (e.g. Finland).

Key Issue:

- Should the sediment quality guidelines of the revised National Action List be formulated using a hybrid approach?

18. Identification of candidate sediment quality guidelines for the revised National Action List

As discussed above, perhaps the most suitable approach for revising the sediment quality guidelines of the existing National Action List is to adopt sediment quality guidelines from another jurisdiction/ country. There are a large number of sediment quality guidelines in the published and grey literature that could conceivably be adopted. In fact, the sheer number of sediment quality guidelines, the wide range (several orders of magnitude; see Chapman and Mann 1999) of concentrations specified for the same metal between comparable guidelines (e.g. Level I guideline), and the lack of consensus on the most appropriate approach for deriving sediment quality guidelines makes this decision a daunting task.

To narrow the list of sediment quality guidelines for consideration, peer-reviewed scientific publications and government reports dealing with sediment quality guidelines that were published on or after 1990 were identified. Sediment quality guidelines in the publications and reports were collated and screened to include only those that are frequently used to assess sediment quality by regulatory authorities and research scientists. Twenty-seven sediment quality guidelines that were derived using the Effects Range, Effects Level, Apparent Effects Threshold, Screening Level, and Sediment Background approaches were identified through this process. These included sediment quality guidelines from Action Lists that are specifically used by various jurisdictions/countries for assessing the suitability of dredged material for unconfined openwater, disposal. Sediment quality guidelines derived using the Equilibrium Partitioning approach were not considered, due to the relatively limited number of chemicals for which guidelines can be derived through this approach.

Several criteria were used to further screen the sediment quality guidelines. Firstly, only sediment quality guidelines that are appropriate to bulk sediment (i.e. sediment with a grain size $\leq 2000 \mu\text{m}$) were considered. Secondly, sediment quality guidelines that define metal concentrations lower than baseline concentrations in sediment from South African coastal

waters were usually not considered. Screening was performed by comparing sediment quality guidelines to baseline concentrations for metals in sediment from Eastern Cape and KwaZulu-Natal coastal waters. Although a previous section of this report provided baseline concentrations for metals in sediment from the St Helena and Saldahna Bay areas of the Western Cape, the data set for this area is too limited to confidently define baseline concentrations. A larger data set, based on sediment samples collected from more locations, is required to define baseline concentrations for Western Cape coastal waters. However, baseline concentrations for most metals of interest for the revised National Action List are broadly comparable between Western Cape and Eastern Cape coastal waters (see Figure 4), and it was thus assumed that the same sediment quality guidelines could be used for both regions. There are exceptions, as discussed further below.

The baseline concentrations used to screen the sediment quality guidelines are presented in Table 16. As discussed elsewhere in this report, since arsenic, cadmium and mercury concentrations in sediment from many areas of the South African coastline are poorly correlated to the mud fraction, and hence also to co-occurring aluminium concentrations, baseline concentrations for these metals were defined using cumulative distribution plots (see Figure 4).

The underlying assumption for baseline concentration determination in the context of biological effects is that the baseline concentrations do not pose a risk. However, because concentrations of most metals in sediment increase as the grain size decreases there is, as discussed previously, no such thing as a single baseline concentration. Rather, the baseline concentration is specific to sediment of a particular grain size composition, but which is typically most dependent on the mud fraction of the sediment. A situation could, therefore, arise where the same metal concentration is at a contaminant level and exerting adverse biological

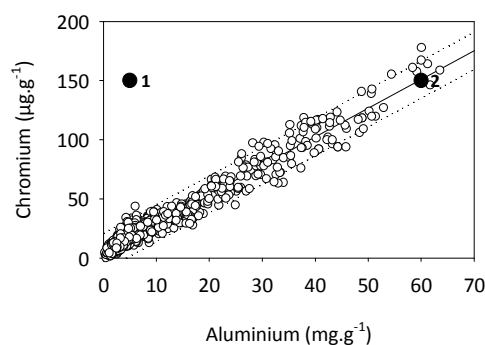


Figure 6. An example of two hypothetical scenarios for the same chromium concentration in sediment. In scenario 1, the chromium concentration exceeds the baseline model upper prediction limit and the sediment sample is interpreted as being chromium enriched. Exceedance of the baseline model upper prediction limit is in fact so high that the sediment sample would be interpreted as being severely contaminated with chromium. In scenario 2, the chromium concentration falls within the baseline model prediction limits and is thus interpreted as falling within the expected concentration range for uncontaminated sediment.

effects in sediment of a particular grain size but falls within the baseline concentration for sediment comprised of finer grained material (see Figure 6). Therefore, for metals other than arsenic, cadmium and mercury the baseline concentration was defined as the concentration predicted at the upper 99% prediction limit of the baseline concentration model at a corresponding aluminium concentration of 60 mg.g⁻¹. Aluminium concentrations in excess of 60 mg.g⁻¹ were measured in only a few sediment samples used to define the baseline models (see Figure 3) and was thus considered a convenient limit. In those situations where the maximum aluminium concentration measured was lower than 60 mg.g⁻¹, the baseline concentration was defined by extrapolation of the baseline model to an aluminium concentration of 60 mg.g⁻¹.

The guidelines of some sediment quality guidelines specify metal concentrations that only slightly exceed baseline concentrations in Eastern Cape and KwaZulu-

Table 16. Baseline concentrations of metals in sediment from Eastern Cape and KwaZulu-Natal coastal waters that were used to screen candidate sediment quality guidelines for the revised National Action List.

Metal	Eastern Cape		KwaZulu-Natal	
	Baseline concentration	Baseline concentration x 1.25	Baseline concentration	Baseline concentration x 1.25
Arsenic	15.65	19.56	17.21	21.51
Cadmium	0.360	0.450	0.518	0.648
Mercury	0.097	0.121	0.100	0.125
Copper	27.70	34.63	44.11	55.14
Chromium	89.63	112.04	166.88	208.60
Nickel	41.04	51.30	58.62	73.28
Lead	35.79	44.74	54.78	68.48
Zinc	98.95	123.69	90.30	112.88

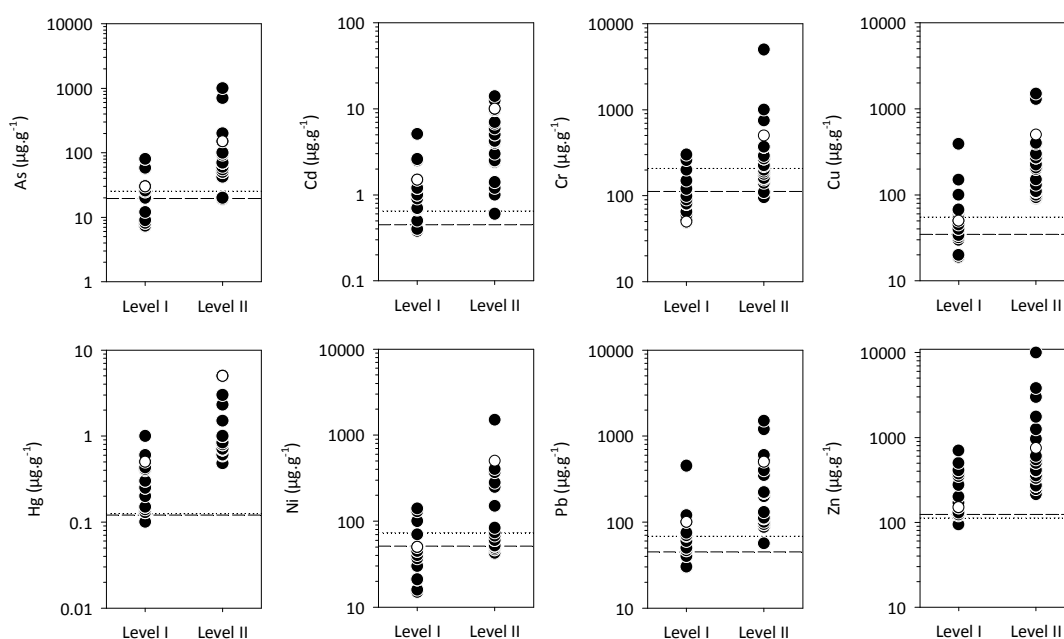


Figure 7. Sediment quality guidelines considered for the revised National Action List (●) compared to baseline concentrations for Eastern Cape coastal waters (dashed horizontal lines) and KwaZulu-Natal (horizontal stippled lines) coastal waters and sediment quality guidelines of the existing National Action List (○).

Natal coastal waters. The level of exceedance was, however, so low that the author considered it unreasonable to expect adverse biological effects to manifest at these concentrations. To account for this situation, and due to the fact that there are uncertainties associated with some of the baseline concentrations, each baseline concentration was multiplied by a factor of 1.25 (Table 16). There is no scientific basis for this decision, but it was made on a best professional judgement basis.

Sediment quality guidelines from other jurisdictions/countries that were initially considered are compared to the baseline concentrations for metals in sediment from Eastern Cape and KwaZulu-Natal coastal waters and to sediment quality guidelines of the existing National Action List in Figure 7. Numerous Level I and in certain cases Level II guidelines were identified as inappropriate based on the fact that the guidelines specify metal concentrations lower than baseline concentrations in South African coastal waters. In cases where two or more Level I guidelines from a sediment quality guideline were identified as inappropriate, the entire sediment quality guideline was not considered further with two exceptions (see below), irrespective of whether the Level II guidelines were potentially appropriate. This approach was taken since the Level II guidelines of numerous sediment quality guidelines were lower than Level I guidelines of other sediment quality guidelines.

At the end of the screening process, four candidate sediment quality guidelines for the revised National Action List were identified. Three of the sediment quality guidelines are appropriate for application in KwaZulu-Natal coastal waters, while all four are appropriate for Eastern Cape (and by implication Western Cape) coastal waters. This said, each of the candidate sediment quality guidelines will need to be modified, either because a Level I guideline is inappropriate to a particular area or because there are no guidelines for chromium and/or nickel.

Numerous of the sediment quality guidelines were in fact discounted from consideration because their Level I guidelines for chromium and nickel are inappropriate for application in South African coastal waters. Whether it was justified to remove them on this basis only is however uncertain. Chromium and nickel appear to be particularly problematic from a sediment quality guideline derivation perspective. Long et al. (1995) placed little confidence in the nickel guidelines they derived, while several sediment quality guidelines do not have guidelines for this metal or for chromium. As stated above, many sediment quality guidelines were screened on the basis that chromium and nickel guidelines are lower than baseline concentrations. The low concentrations of these metals specified by the Level I guidelines of these sediment quality guidelines might have arisen due to the fact that empirically-based sediment quality guidelines are derived for each

chemical individually, but using data that comprises a mixture of chemicals in sediment samples. Thus, if an adverse biological effect was evident then this was attributed to all chemicals in the sediment sample, irrespective of whether the chemical was responsible for the adverse effect. Thus, if a chemical was present at low concentrations in sediment from impacted sites, then it would appear in a large percentage of the samples in which toxicity is observed. This will result in the guideline specifying low concentrations for the chemical and may in fact lead to a situation where the guideline is lower than baseline concentrations. This situation is exacerbated by the fact that there is clear evidence for the natural enrichment of chromium in sediment from the east coast of South Africa. Chromium concentrations measured in sediment from these waters are amongst the highest naturally occurring concentrations in the world.

19. Candidate sediment quality guidelines for the revised National Action List

The first of the candidate sediment quality guidelines was derived for various sediment management needs in the province of Quebec, Canada (Environment Canada and MDDEP 2007; see Table 10). Two of the five guidelines, the Threshold Effect Level and the Probable Effect Level, were derived by the Canadian Council of Ministers of the Environment, using the Effects Level approach (CCME 2002). These guidelines define three concentration ranges with management implications. Chemicals in sediment at concentrations at or below the Threshold Effect Level are anticipated to rarely cause adverse biological effects to sediment-dwelling organisms. Chemicals in sediment at concentrations between the Threshold Effect Level and Probable Effect Level are anticipated to occasionally be associated with adverse biological effects to sediment-dwelling organisms, while concentrations exceeding the Probable Effect Level are anticipated to be frequently associated with adverse effects to sediment-dwelling organisms. The definition of these ranges is based on the premise that the probability of toxic effects resulting from exposure to a given chemical increases with the concentration of that substance in sediments. For most substances, the incidence of adverse effects is 10% or less for the range of concentrations below the Threshold Effect Level. For the range of concentrations above the Probable Effect Level (biological effects are frequently observed), the incidence of adverse effects varies considerably among chemicals and is sometimes lower than 50% (CCME 2002). The low incidence of adverse

effects observed for a number of chemicals in this range indicates that the degree of correspondence between the concentrations obtained for the Probable Effect Level and the narrative intent of the Probable Effect Level (concentration above which adverse effects are frequently observed) is occasionally somewhat weaker than is the case for the Threshold Effect Level.

The government of Quebec decided to adopt the Probable Effect Level and the Threshold Effect Level for sediment quality assessment in the province, but was of the opinion that the guidelines alone are insufficient to address all sediment management needs. Three additional guidelines were thus derived to cover other management needs. The management needs include the prevention of sediment contamination from industrial discharges, the management of dredged sediment, and the remediation of contaminated aquatic sites. To monitor the status of vulnerable sites and to provide advance warning of incipient contamination, through industrial discharges, the Rare Effect Level was derived. The concentrations of chemicals defined by the Rare Effect Level are somewhat lower than the Threshold Effect Level, but both of these guidelines are intended for use as signposts for preventing contamination of sediment (see Table 10).

The unconfined, openwater disposal of dredged sediment in Canada is prohibited except where the material presents no threat to aquatic organisms. The government of Quebec considered that a contamination threshold beyond which toxicity testing is mandatory was necessary, since experience has shown that most toxicity tests are not very sensitive to low levels of contamination. A value higher than the Threshold Effect Level but lower than the Probable Effect Level was considered necessary, the Occasional Effect Level. The Occasional Effect Level represents the concentration of a chemical in sediment above which adverse effects are anticipated to many sediment-dwelling organisms. To further facilitate the management of dredged sediment, a sufficiently high threshold of contamination above which unconfined, openwater disposal is prohibited without the need for additional analyses was considered necessary. Since the Probable Effect Level is not a high enough threshold for this type of decision, the Frequent Effect Level was derived. This guideline represents the concentration of a chemical in sediment above which adverse effects are anticipated for the majority of sediment-dwelling organisms. The Occasional Effect Level and the Frequent Effect Level are the primary guidelines governing the management of dredged sediment, but the Probable Effect Level and the

Frequent Effect Level can also be used to provide guidance for decisions on whether contaminated sediment needs to be remediated.

For the purposes of dredged material management, when the concentrations of all chemicals in sediment are lower than or equal to the Occasional Effect Level, the probability for observing adverse effects to sediment-dwelling organisms is anticipated to be relatively low. The dredged sediment is therefore of a suitable quality for unconfined, openwater water disposal or for other beneficial uses, provided that disposal does not contribute to the deterioration of the receiving environment (e.g. through a change in sediment grain size at the disposal site). When the concentration of a chemical is higher than the Occasional Effect Level but lower than or equal to the Frequent Effect Level, the probability for observing adverse biological effects to sediment-dwelling organisms is anticipated to be relatively high and to increase with increasing chemical concentrations. Unconfined, openwater disposal of dredged sediment can be a valid management option in this case, but only when toxicity tests demonstrate that the sediment will not adversely affect the receiving environment (e.g. through direct toxic effects and indirect bioaccumulation effects). Proper characterisation of the dredged material disposal site is required in this instance prior to authorisation of openwater disposal. Chemical concentrations in the dredged sediment must be lower than or equal to concentrations measured in sediment from the dredged material disposal site. When the concentration of a chemical exceeds the Frequent Effect Level, the probability for observing adverse effects to sediment-dwelling organisms is anticipated to be very high and the sediment is considered unsuitable for unconfined, openwater disposal. These sediments must instead be treated or safely contained (e.g. in confined sediment facilities).

Comparison of sediment quality guidelines used in Quebec to the baseline concentrations for metals in sediment from South African coastal waters demonstrated that one or more of the Rare Effect Level, Threshold Effect Level and Occasional Effect Level are lower than baseline concentrations. These guidelines were consequently eliminated from further consideration. The Probable Effect Level and Frequent Effect Level are potentially suitable for inclusion in the revised National Action List, as Level I and Level II guidelines respectively (Table 17). The Level I guideline for chromium is, however, inappropriate for application in sediment from both Cape and KwaZulu-Natal coastal waters since it is lower than the baseline concentration in the former area and only slightly lower than the baseline concentration in the latter area (Figure 8). The Quebec sediment quality guidelines also do not provide guidelines for nickel (Table 17). Consequently, guidelines for nickel will need to be adopted from another jurisdiction/country or formulated using the Sediment Background approach should the Quebec sediment quality guidelines be considered for inclusion in the revised National Action List. The narrative intent of the Level I guideline will not, however, be per the original intent of the Probable Effect Level since it will be necessary to consider the narrative intent of this guideline as the concentration of a chemical in sediment below which adverse effects to sediment-dwelling organisms are anticipated to be infrequent. This is consistent with the narrative intent of the Level I guideline of most sediment quality guidelines that adopt a two guideline (Action Level) approach. There is thus no scientific basis for such a change in narrative intent apart from the ease of application in the revised National Action List. Whether the waiving of the original narrative intent of the sediment quality guidelines is of any relevance is open to debate, since irrespective of the sediment quality guidelines adopted it will be impossible to determine whether their original narrative intent is

Table 17. Candidate metal sediment quality guidelines ($\mu\text{g}\cdot\text{g}^{-1}$ dry weight) for the revised National Action List. PEL - Probable Effect Level, FEL - Frequent Effect Level (Environment Canada and MDDEP 2007); SGC_{SCS} - Sediment Quality Criteria for Sensitive Contaminated Sites, SGC_{TCS} - Sediment Quality Criteria for Typical Contaminated Sites (MacDonald et al. 2003); SL1 - Screening Level 1, SL2 - Screening Level 2 (USACE 2006); SL - Screening Level, BT - Bioaccumulation Trigger, ML - Maximum Level (USACE 2008).

Metal	Quebec, Canada		British Columbia, Canada		Pacific Northwest, USA		Pacific Northwest, USA		
	PEL	FEL	SGC _{SCS}	SGC _{TCS}	SL1	SL2	SL	BT	ML
Arsenic	42	150	26	50	57	93	58	507.1	700
Cadmium	4.2	7.2	2.6	5	5.1	6.7	5.1	11.3	14
Chromium	160	290	99	190	260	270	-	267	-
Copper	110	230	67	130	390	390	390	1027	1300
Mercury	0.7	1.4	0.43	0.84	0.41	0.59	0.41	1.5	2.3
Nickel	-	-	-	-	-	-	140	370	370
Lead	110	180	69	130	450	530	450	975	1200
Zinc	270	430	170	330	410	960	410	2783	3800

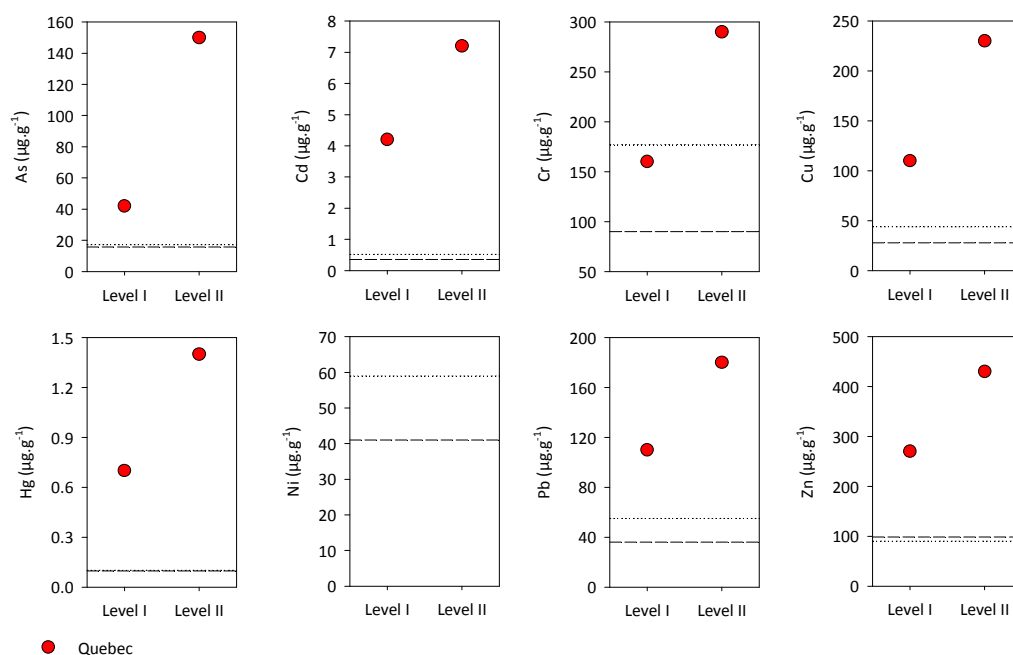


Figure 8. Comparison of sediment quality guidelines for Quebec (Environment Canada and MDDEP 2007) to baseline concentrations for metals in sediment from Eastern Cape (dashed lines) and KwaZulu-Natal coastal waters (stippled lines).

consistent with South African conditions due to the lack of empirical data against which to evaluate their performance.

The second of the candidate sediment quality guidelines were derived for different management needs in the province of British Columbia, Canada (Table 17; MacDonald et al. 2003). Two guidelines were derived, using the Effects Level approach. The guidelines are intended to identify concentrations of chemicals in sediment below which risks to sediment-dwelling organisms are considered 'tolerable', but they either have application to so-called Sensitive Contaminated Sites or Typical Contaminated Sites. Sensitive Contaminated Sites are sites that contain species or habitats of special importance, and one of the guidelines (Sediment Quality Criteria for Sensitive Contaminated Sites, hereafter referred to as the Level I guideline) was derived to provide a relatively high level of protection (about a 20% probability for adverse effects) to sediment-dwelling organisms from these sites. The Sediment Quality Criteria for Typical Contaminated Sites (hereafter referred to as the Level II guideline) offers a lower level of protection (about a 50% probability for adverse effects) to sediment-dwelling organisms, and has application at sites where human related disturbances are prevalent (e.g. in ports). The definition of the Level II guideline is intended to take into account that the potential for maintaining an unaltered benthic invertebrate community is reduced in these areas. Assessment of the guidelines in meeting their narrative

intent using empirical data showed that for most chemicals the guidelines had a moderate to high level of reliability. Both of the guidelines have substantial administrative rules that govern their application (see below). The guidelines are not intended for assessing whether sediment is suitable for unconfined, openwater disposal, but they could be considered for this purpose if they are adopted for the revised National Action List.

Determining whether or not Sensitive Contaminated Sites or Typical Contaminated Sites are contaminated involves a stepwise approach. First, the 90th percentile of concentrations (i.e. 9 out of 10 samples) for each chemical of concern is calculated and compared to the relevant sediment quality guideline. In other words, more than one sediment sample is required to characterise sediment quality at a site. Next, the upper limits of background concentrations are calculated and compared to the 90th percentile concentration of each chemical of concern. If the 90th percentile concentration of one or more chemicals of concern exceeds the relevant sediment quality guideline and the upper limit of the background concentration, then the site is considered to contain contaminated sediments. The presence of chemicals of concern at concentrations that exceed the relevant sediment quality guideline by a factor of two or more is also considered to be indicative of the presence of contaminated sediments. In addition to comparing the concentrations of each chemical of concern to the relevant sediment quality guideline, sites can be designated as contaminated through the

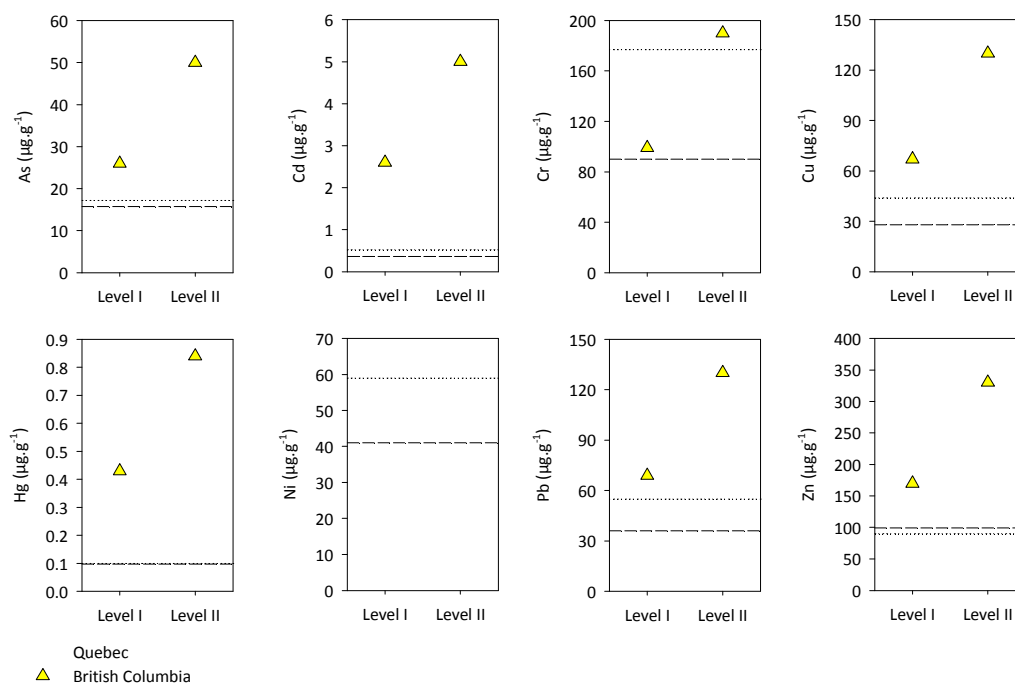


Figure 9. Comparison of sediment quality guidelines for British Columbia (MacDonald et al. 2003) to baseline concentrations for metals in sediment from Eastern Cape (dashed lines) and KwaZulu-Natal coastal waters (stippled lines)..

application of a mean sediment quality guideline quotient. The mean sediment quality guideline quotient is calculated for each sediment sample for which sediment chemistry data are available. The 90th percentile of the mean sediment quality guideline quotients is then calculated. If the 90th percentile of the mean sediment quality guideline quotients exceeds one, then the site is considered to contain contaminated sediments.

Comparison of the sediment quality guidelines used in British Columbia to baseline concentrations for metals in sediment from South African coastal waters demonstrates that the Level I guideline for chromium is inappropriate for application in KwaZulu-Natal coastal waters (Figure 9) while the Level I and Level II guidelines for cadmium are inappropriate for application in the St Helena Bay and Saldanha Bay areas of the Western Cape. The Level I guidelines for arsenic, chromium and lead are, however, only slightly higher than baseline concentrations for these metals in one or all regions (Figure 9). Guidelines for these metals will therefore need to be modified if they are adopted for the revised National Action List.

The British Columbia sediment quality guidelines do not provide guidelines for nickel (Table 17). Consequently, sediment quality guidelines for nickel will need to be adopted from another jurisdiction/country or formulated using the Sediment Background approach if

the sediment quality guidelines are adopted for the revised National Action List.

The two remaining candidate sediment quality guidelines are used to assess sediment quality in the Pacific Northwest of the USA (states of Washington, Oregon and Idaho), with a particularly strong focus on assessing sediment identified for dredging. The guidelines are provided in two reports, one prepared by the United States Army Corps of Engineers (USACE 2008) and the other by the United States Army Corps of Engineers in collaboration with several regulatory authorities. Since the latter report does not provide a citing format and the United States Army Corps of Engineers was the lead organisation, the report is henceforth referred to as USACE (2006).

USACE (2006) describes a framework for assessing and characterising the quality of freshwater and marine sediments in the Pacific Northwest, for the purpose of determining management options for dredged material and contaminated sites. The sediment quality guidelines provided in the framework (Table 17) are identical to the Washington State sediment management standards (see WSDEC 1995). The sediment quality guidelines can be used to evaluate risks posed by *in situ* sediments to sediment-dwelling organisms, or if dredging is proposed then the risks posed by the newly exposed sediment to sediment-dwelling organisms and the suitability of the dredged sediment for unconfined, openwater disposal.

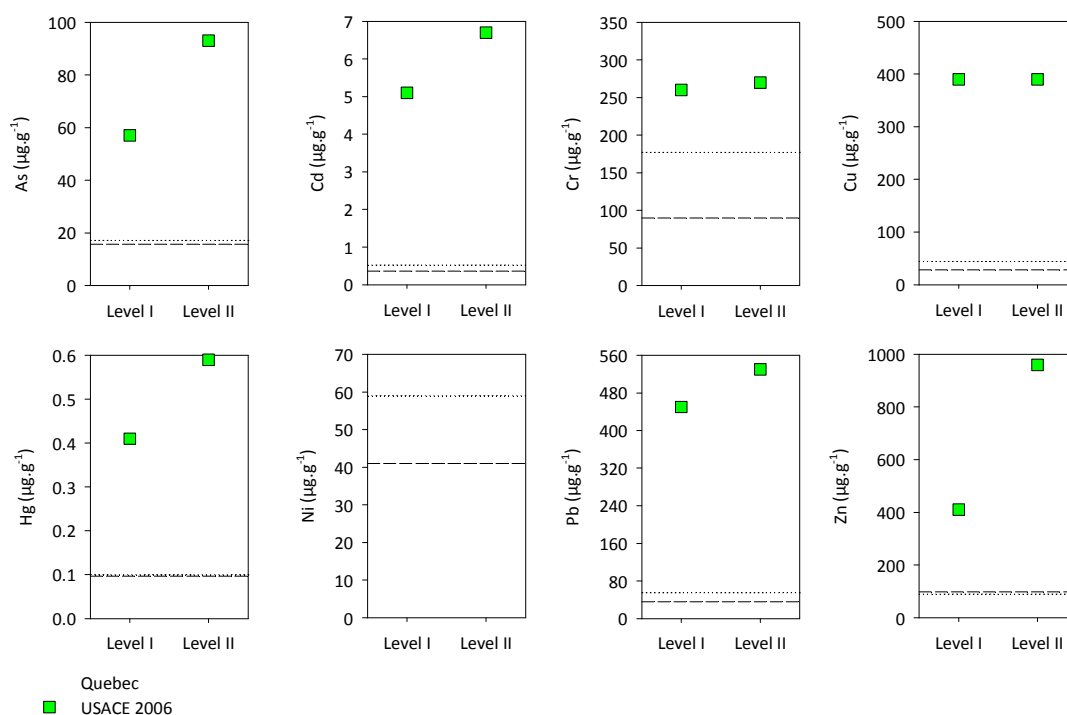


Figure 10. Comparison of sediment quality guidelines for the Pacific Northwest of the United States of America (USACE 2006) to baseline concentrations for metals in sediment from Eastern Cape (dashed lines) and KwaZulu-Natal coastal waters (stippled lines).

The sediment quality guidelines were derived through the Apparent Effects Threshold approach, using data from sites in the Pacific Northwest. The reader will recall that the Apparent Effects Threshold approach attempts to define the concentrations of contaminants in sediment above which statistically significant adverse effects for a specific endpoint (e.g. mortality) are always expected. Two guidelines were derived. The Screening Level 1 (hereafter referred to as the Level I guideline) corresponds to a chemical concentration at or below which adverse effects to sediment-dwelling organisms are expected to be low, while the Screening Level 2 (hereafter referred to as the Level II guideline) defines chemical concentrations above which minor adverse effects may be observed in the more sensitive groups of sediment-dwelling organisms. The Level I guidelines represent the Apparent Effects Threshold derived from data for alterations in benthic invertebrate community composition, while most of the Level II guidelines were derived using data from amphipod toxicity tests.

For dredging projects, the Level I guideline is intended to identify chemical concentrations at or below which there is no reason to believe that the disposal of dredged sediment will result in unacceptable adverse effects to sediment-dwelling organisms. Decision-making is not based entirely on this guideline, however, since consideration is also given to factors such as the presence of chemicals in sediment that have a high

bioaccumulation potential. Sediments with one or more chemicals at concentrations exceeding the Level I guideline and/or bioaccumulation criteria (these have not yet been defined) require toxicity testing and/or bioaccumulation testing respectively before a decision is made on the suitability of the material for unconfined, openwater disposal. If biological testing is implemented, then results from the biological testing take precedence and override the sediment chemistry results. Sediments with chemicals at concentrations that exceed the Level I guideline or bioaccumulation criteria and fail follow-on biological testing generally need to be managed in an alternate, more protective manner (e.g. in a confined disposal facility or landfill site). The Level II guideline is not used for decision-making in the context of dredging projects, but rather as a clean-up level (it is also used as a benchmark against which to prioritise different sites for clean-up).

Comparison of the USACE (2006) sediment quality guidelines to the baseline concentrations for metals in sediment from South African coastal waters (Figure 10) demonstrates that each of the guidelines is suitable for application in all local waters with exception of cadmium in the St Helena Bay and Saldanha Bay areas of the Western Cape. The USACE (2006) sediment quality guidelines do not provide guidelines for nickel (Table 17). Consequently, sediment quality guidelines for nickel will need to be adopted from another jurisdiction/country or

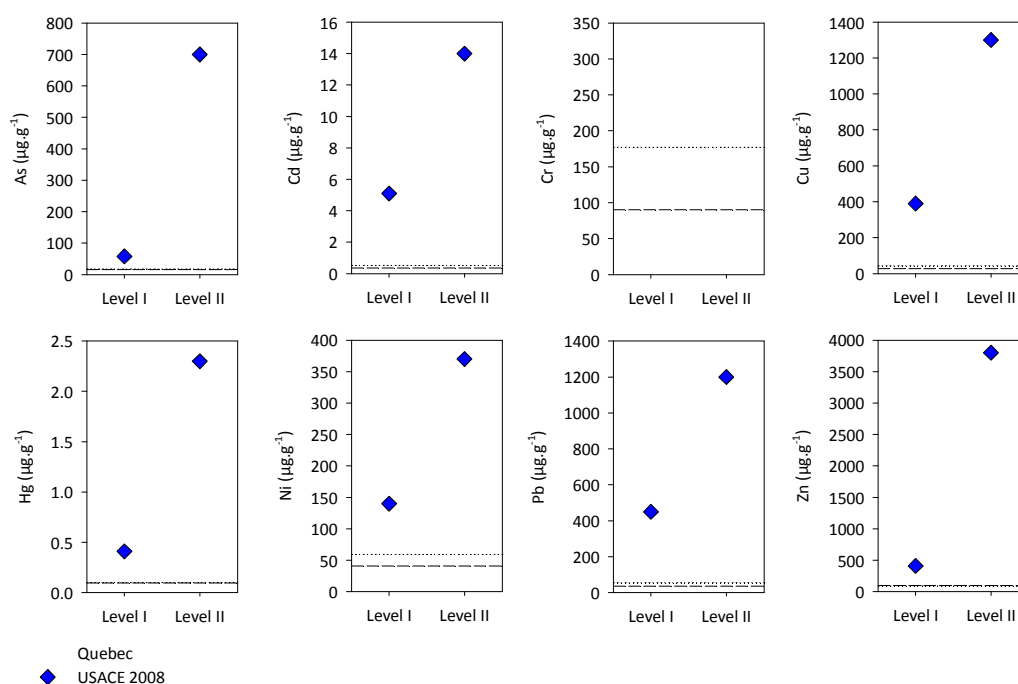


Figure 11. Comparison of sediment quality guidelines for the Pacific Northwest of the United States of America (USACE 2008) to baseline concentrations for metals in sediment from Eastern Cape (dashed lines) and KwaZulu-Natal coastal waters (stippled lines).

formulated using the Sediment Background approach should the USACE (2006) sediment quality guidelines be adopted for the National Action List.

USACE (2008) describes a Dredged Material Evaluation and Disposal Procedures user manual for the State of Washington, USA. The sediment quality guidelines comprise three guidelines (Table 17). As was the situation for USACE (2006), the sediment quality guidelines were derived using the Apparent Effects Threshold approach. The Screening Level (hereafter referred to as the Level I guideline) is identical to the Level I guideline of USACE (2006) with the exception that there is 1 µg.g⁻¹ difference for arsenic (Table 17). Whether this reflects a typographical error is uncertain. The second guideline is the Maximum Level (hereafter referred to as the Level II guideline). USACE (2008) also provides a Bioaccumulation Trigger, which represents the concentration of chemicals in sediment at or above which there is a strong possibility that the chemicals may be bioaccumulated. The Bioaccumulation Trigger defines chemical concentrations that are higher than the Level I guideline but lower than the Level II guideline. There are no Level I and Level II guidelines for chromium, but here in contrast to USACE (2006) guidelines for nickel are provided. A Bioaccumulation Trigger is, however, provided for chromium. The Level II guideline for all chemicals is usually substantially higher than the Level II guideline of USACE (2006, see Table 17 and Figure 8).

Consistent with USACE (2006), the Level I guideline of USACE (2008) corresponds to a chemical concentration at or below which adverse effects to sediment-dwelling organisms are expected to be low. The Level II guideline represents the highest Apparent Effects Threshold at which all four biological indicators used for guideline derivation showed significant effects, and thus corresponds to a chemical concentration at or above which there is a very high probability for adverse effects to sediment-dwelling organisms. For dredging projects, the Level I guideline is intended to identify chemical concentrations at or below which there is no reason to believe that the disposal of dredged material would result in unacceptable adverse effects to sediment-dwelling organisms. Sediment with chemicals at concentrations that exceed the Level I guideline requires biological testing before a decision is made on the suitability of the material for unconfined, openwater disposal. The Level II guideline is not used a decision-making, but is provided for the benefit of dredging proponents. Although some sediment in the state of Washington with one or more chemicals at concentrations exceeding the Level II guideline have passed biological testing, the majority of sediments have failed. The purpose of this guideline is to allow the dredging proponent to decide whether it is worthwhile proceeding with expensive and time consuming biological testing to determine whether the material is suitable for unconfined, openwater disposal, or whether the effort should be channelled into other disposal

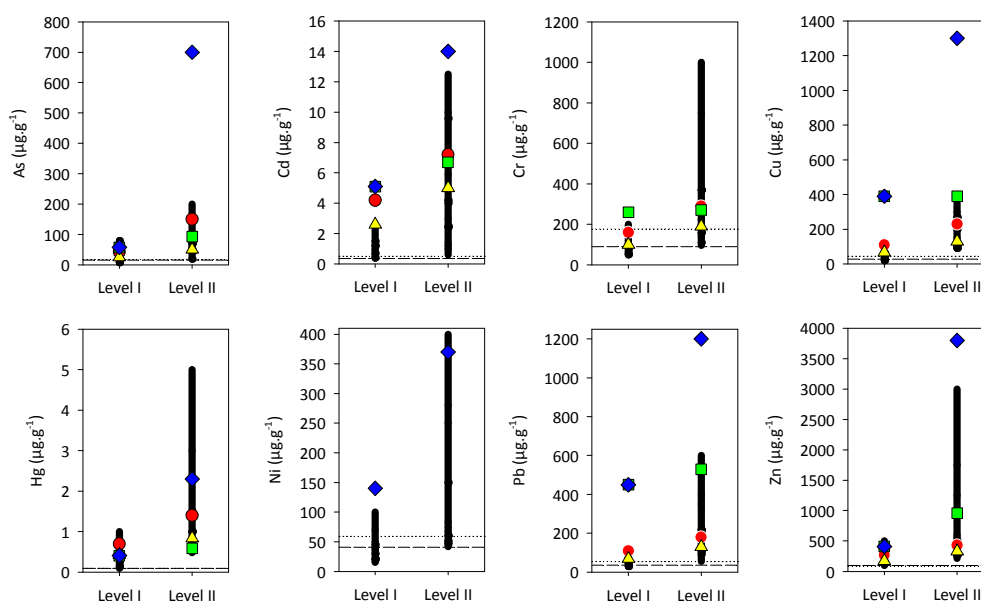


Figure 12. Comparison of candidate sediment quality guidelines for the revised National Action List (● - Quebec (Environment Canada and MDDEP 2007), ▲ - British Columbia (MacDonald et al. 2003), ■ - USACE (2006), ◆ - USACE (2008)) to the range of concentrations (solid vertical line) specified by sediment quality guidelines used in various jurisdictions/countries for dredged material assessment. The horizontal dashed and stippled lines represent baseline concentrations for metals in sediment from Eastern Cape and KwaZulu-Natal coastal waters respectively.

options (e.g. confined disposal) and testing for those options. Sediment with one or chemicals at concentrations exceeding the Bioaccumulation Trigger guideline requires bioaccumulation testing before a decision is made on the suitability of the sediment for unconfined, openwater disposal.

Comparison of the USACE (2008) sediment quality guidelines to the baseline concentrations for metals in sediment from South African coastal waters demonstrates that each of the guidelines is suitable for application in all local waters (Figure 11) with exception of the Level I guideline for cadmium in the St Helena Bay and Saldanha Bay areas of the Western Cape. As stated above, the USACE (2008) sediment quality guidelines do not provide guidelines for chromium. Consequently, sediment quality guidelines for chromium will need to be adopted from another jurisdiction/country or formulated using the Sediment Background approach should the USACE (2008) sediment quality guidelines be considered for inclusion in the revised National Action List. Although discussed further below, the Level II guideline (Maximum Level) defines very high concentrations of chemicals in sediment.

The most noticeable feature from Table 17 is that the Level II and Bioaccumulation Trigger guidelines of the USACE (2008) sediment quality guidelines are typically substantially higher than the Level II guidelines of other candidate sediment quality guidelines. This reflects the fact that the guideline represents the highest Apparent

Effects Threshold at which all four biological indicators used for guideline derivation showed significant effects. The Level I guidelines for copper and lead from the USACE (2006) and USACE (2008) sediment quality guidelines are also somewhat higher than the corresponding guideline of other candidate sediment quality guidelines.

The second noticeable feature is that Level I guidelines for chromium in Quebec and British Columbia sediment quality guidelines are inappropriate for application in KwaZulu-Natal coastal waters, since the guidelines are lower than the baseline concentration. As discussed previously, despite this limitation these sediment quality guidelines were not trimmed from the candidate list since guidelines for all other metals do not suffer this limitation. If either of the Quebec or British Columbia sediment guidelines is adopted for the revised National Action List, then the Level I guideline for chromium will need to be adopted from another jurisdiction/country or formulated using the Sediment Background approach. Either of the latter approaches will similarly need to be followed for the Level I guideline for chromium in USACE (2008) and nickel in Ontario (Environment Canada and MDDEP 2007), British Columbia (MacDonald et al. 2003) and USACE (2006) sediment quality guidelines, since guidelines are not provided for these metals.

The third noticeable feature is that concentrations of cadmium in sediment from the St Helena Bay and Saldanha Bay areas of the Western Cape usually exceed

the Level I and Level II guideline of all candidate sediment quality guidelines. As discussed elsewhere in this report, the elevated cadmium concentrations measured in sediment from these areas appears to be a natural phenomenon. However, the spatial extent of naturally elevated cadmium concentrations in Western Cape coastal waters is unknown and can only be established through a dedicated research programme aimed at defining baseline concentrations for metals in sediment from these waters. Thus, irrespective of the candidate sediment quality guidelines adopted for the National Action List, cadmium will need to be excluded from decision-making in at least the latter areas of the Western Cape.

To place the candidate sediment quality guidelines into perspective, Figure 12 compares them to sediment quality guidelines used in numerous jurisdictions/countries for decision-making with regard to dredged material, to baseline concentrations for South African coastal waters, and to sediment quality guidelines of the existing National Action List. This is by no means an exhaustive comparison, but provides an appreciation of the range of metal concentrations defined by Level I and Level II guidelines that are used for decision-making in the context of dredged sediment management. The most noticeable feature again is that the Level II guideline of the USACE (2008) sediment quality guidelines is substantially higher than the Level II guideline of other sediment quality guidelines. The Level I guidelines for chromium, copper and lead in the USACE (2006) and USACE (2008) sediment quality guidelines are also somewhat higher compared to the other guidelines.

In terms of their protectiveness to sediment-dwelling organisms, the candidate sediment quality guidelines rank in approximate order British Columbia > Environment Canada (2006) > USACE (2006) > USACE (2008). This ranking is based only on absolute concentrations specified by the guidelines and does not take into account the derivation procedure and associated narrative intent.

20. Sediment quality guideline example

As stated above, comparison of the candidate sediment quality guidelines to baseline metal concentrations in sediment from South African coastal waters demonstrates that, barring certain minor modifications and/or additions, each of the candidate sediment quality can be considered for inclusion in the revised National Action List. A decision will obviously need to be made on which sediment quality guidelines, or combination thereof, is most suitable and appropriate for the revised

National Action List. As mentioned elsewhere in this report, this decision does not rest with the author or Branch: Oceans and Coasts of the DEA, but rather should be reached through consensus after considering the various factors discussed previously (e.g. narrative intent). In this context, it will be extremely valuable if stakeholders that have implemented dredging projects in South African coastal waters (e.g. Transnet National Ports Authority) provide sediment chemistry data against which the guidelines can be compared, to determine (1) whether they lean toward being over- or under-protective, (2) their implication for the frequency of further assessment that would have been required for decision-making, and (3) their implication for the consideration and need of alternate dredged sediment disposal strategies.

The author has, however, deemed it prudent to define a sediment quality guideline example, as a starting point for discussion. Three guidelines are provided in Table 19, namely Warning Level, Level I and Level II guidelines. Most of the Level I and Level II guidelines were taken directly from USACE (2006). Since the USACE (2006) Level I and Level II guidelines for copper are identical, the Level I guideline was substituted with the Level II guideline for Quebec (Environment Canada and MDDEP 2007), to provide a gradation in concentrations. The USACE (2006) Level I guideline for lead is, in the opinion of the author, high and was consequently substituted with the more protective Level II guideline from the Long et al. (1995) sediment quality guidelines. The USACE (2006) Level II guidelines for cadmium and chromium are not much higher than the Level I guidelines and were also substituted by the Level II guidelines from the Long et al. (1995) sediment quality guidelines.

Table 18. Example of possible sediment quality guidelines for the revised National Action List. Two Warning Level concentrations are included for chromium and nickel, to account for substantial differences in the baseline concentrations for these metals between Eastern Cape and KwaZulu-Natal coastal waters.

Metal	Warning Level	Level I	Level II
Arsenic	42 ^a	57 ^b	93 ^b
Cadmium	1.2 ^c	5.1 ^b	9.6 ^c
Chromium	135 ^d /250 ^e	260 ^b	370 ^c
Copper	110 ^a	230 ^a	390 ^b
Mercury	0.43 ^b	0.84 ^b	1.5 ^c
Nickel	62 ^d /88 ^e	140 ^b	370 ^b
Lead	110 ^a	218 ^c	530 ^b
Zinc	270 ^a	410 ^b	960 ^b

a - Environment Canada and MDDEP (2007), b - USACE (2006), c - Long et al. (1995), d - for Eastern Cape, e - for KwaZulu-Natal

The motivation for defining a Warning Level is that several of the Level I and Level II guidelines from USACE (2006) are considerably higher than Level I and Level II guidelines of sediment quality guidelines that are frequently used to assess sediment quality in North America. For example, the Level I and Level II guidelines for zinc in the Long et al (1995) sediment quality guidelines specify concentrations of 150 and 410 $\mu\text{g.g}^{-1}$ respectively, while the USACE (2006) Level I and Level II guidelines specify concentrations of 410 and 960 $\mu\text{g.g}^{-1}$ respectively. These differences reflect, in part, the different approaches used to derive the sediment quality guidelines, that is, the Apparent Effects Threshold approach for USACE (2006) and the Effects Range approach for Long et al. (1995). Since it is uncertain whether the USACE (2006) Level I guideline is sufficiently protective of sediment-dwelling organisms in South African coastal waters, it might be prudent to define a guideline that provides a higher level of protection.

The Warning Level for most metals corresponds to the Probable Effects Level of the Environment Canada (2006) sediment quality guidelines (Table 19). As discussed previously, the Quebec sediment quality guidelines do not provide guidelines for nickel (Table 19) while the Level I guideline for chromium is inappropriate for application in KwaZulu-Natal coastal waters (Figure 13). The chromium and nickel Level I guidelines of the British Columbia and Long et al. (1995) sediment quality guidelines are either inappropriate for application in South African coastal waters or no guideline is provided, so neither of the guidelines could be considered as a substitute. The Warning Level for chromium and nickel was, therefore, derived using the Sediment Background approach, as 1.5 times the baseline concentration at a co-occurring aluminium concentration of 60 mg.g^{-1} . The resultant guideline for chromium is problematic as it can probably only be applied in Cape coastal waters, since 1.5 times the chromium baseline concentration for KwaZulu-Natal coastal waters (250.32 $\mu\text{g.g}^{-1}$) is only marginally below the Level I guideline (260 $\mu\text{g.g}^{-1}$) derived using this approach. There is, therefore, a strong probability that the guideline could be exceeded even in uncontaminated sediment. Of course, the Warning Level for chromium and nickel could be defined as a higher ratio of the baseline, at say 1.65. Indeed, the Warning Level for all metals could be defined as a ratio of the baseline concentration, or even as a ratio of the Level I guideline (e.g. 50%). Defining the Warning Level as a proportional exceedance of the baseline concentrations may, however, require separate guidelines for Cape and KwaZulu-Natal coastal waters because of differences in metal baseline concentrations. In fact, separate

guidelines may need to be defined for any other area of the South African coastline for which baseline concentrations may be defined and which differ from baseline concentrations for Cape and KwaZulu-Natal coastal waters.

Alternately, a suitable guideline could be identified for KwaZulu-Natal coastal waters and then simply be applied to all South African coastal waters. This is a less cumbersome approach, but does have the consequence that the guidelines will be less protective of sediment-dwelling organisms in Cape waters, where baseline concentrations for most metals are (often substantially) lower than in KwaZulu-Natal coastal waters (see further discussion below). Whether this is of any significance is a moot point since it in any case applies to the Warning Level for other metals that were not derived using the Sediment Background approach and for which differences exist between regions of the coastline.

All three guidelines could be incorporated into the revised National Action List, with decision-making restricted to the Level I and Level II guidelines. That is, the Warning Level could be used to warn of incipient contamination. The Level I guideline could then reflect the concentrations above which further assessment is required, and the Level II guideline as a prohibition of unconfined, openwater disposal of dredged sediment.

As discussed on several occasions in this report, none of the guidelines will be scientifically defensible from a performance perspective since their performance in local waters can only be established through comparison to matching sediment chemistry and biological effects data, and these data are absent. However, sediment assessment tools are required for decision-making, and as and when the state of science in South Africa becomes more advanced in this context the guidelines can and should be revised.

21. Other factors for consideration

21.1 Should sediment quality guidelines be formulated separately for different regions of South Africa?

As mentioned and discussed on several occasions elsewhere in this report, the baseline concentrations for several metals differ (in some cases substantially) between sediment from Eastern and Western Cape and KwaZulu-Natal coastal waters. The question that then arises, and which was alluded to above, is whether separate sediment quality guidelines should be defined for different regions of the country or whether a 'one

size fits all approach' should be adopted. Although this question is most relevant to sediment quality guidelines derived using the Sediment Background approach, the considerations discussed below extend to any sediment quality guideline that might be adopted from another jurisdiction/country.

The need to possibly define separate sediment quality guidelines for different regions of the South African coastline is due to the fact that sediment-dwelling organisms have evolved in the presence of the natural occurring (baseline) metal concentrations. Theoretically, therefore, if metal concentrations in sediment differ between regions then sediment-dwelling organisms from these regions should display differences in tolerance to the same metal concentration. There have, unfortunately, been no detailed studies to determine whether the tolerances of sediment-dwelling organisms from different coastal regions of South Africa display differences in tolerance to metals. If they do then a sediment quality guideline derived using the Sediment Background approach for KwaZulu-Natal coastal waters, where baseline metal concentrations are higher, might be under protective for Cape coastal waters.

As discussed previously, the findings of several studies suggest that water and sediment quality guidelines cannot simply be extrapolated from one (climatic) region to another and be expected to provide the same level of protection (e.g. Chapman and Mann 1999, Chapman et al. 1999, Chapman and Riddle 2005, Chapman et al. 2006, Kwok et al. 2007).

Deriving separate sediment quality guidelines for different regions of the South African coastline is thus the most appropriate approach from a scientific perspective, but this might result in a slightly cumbersome decision-making process. The limitation of potential under-protectiveness inherently extends to sediment quality guidelines that may be adopted for inclusion in the revised National Action List on the basis that the guidelines are identified as suitable for KwaZulu-Natal coastal waters and are then applied as a one size fits all approach to South African coastal waters.

Key issue:

8. Should separate metal sediment quality guidelines be formulated for different regions of the South African coastline to account for differences in baseline concentrations?

21.2 Absence of sediment quality guidelines for other chemicals

As mentioned in the introduction to this report, the intention is to revise only the metal sediment quality guidelines of the existing National Action List. The overwhelming emphasis of the existing National Action List on metal sediment quality guidelines for decision-making was identified previously as a significant limitation and will extend to the revised National Action List should guidelines for other chemicals not be included. The absence of sediment quality guidelines for other chemicals raises the question whether reliable decisions on sediment quality can be made using metal concentration data only.

In its existing form, the National Action List inherently uses metal concentrations in sediment as a surrogate indicator of the presence of other chemicals, that is, the assumption is that if sediment is not contaminated with metals then it is unlikely to be contaminated with other chemicals. Although metal and organic contaminants usually accumulate in sediment from the same areas because they are similarly particle reactive and often have similar sources (e.g. emissions from vehicles), this does not necessarily imply that different types of chemicals will always be at, or not at, contaminant concentrations in the same sediment (e.g. Stronkhorst and van Hattum 2003). A further significant feature is that while metals may bioaccumulate, with the exception of methylmercury they do not biomagnify. Many organic chemicals, in contrast, have a high biomagnification potential (Sorenson et al 2007). Failure to take organic chemicals with a high biomagnification potential into consideration can, therefore, have significant implications from an ecological risk perspective.

The author thus recommends that consideration be given to the inclusion of sediment quality guidelines for other chemicals in the revised National Action List. Since existing data on the concentrations of various organic chemicals in sediment from South African coastal waters are insufficient for the definition of baseline concentrations, the simplest and most logical strategy is to adopt sediment quality guidelines for these chemicals from another country/jurisdiction. In the sediment quality guideline example discussed previously, the metal sediment quality guidelines were taken predominantly from USACE (2006). The USACE (2006) sediment quality guidelines for other chemicals could thus also be adopted. This will mean that most of the sediment quality guidelines will have been derived using

the same approach and will have a similar narrative intent.

To provide the reader with perspective on guidelines for organic chemicals, the sediment quality guidelines for all chemicals in USACE (2006) are presented in Table 19. It is important to note that different sediment quality guidelines define guidelines for different suites of chemicals. For example, the Quebec sediment quality guidelines do not provide guidelines for chlorinated hydrocarbons, phthalates and miscellaneous extractables included in the USACE (2006) sediment quality guidelines, but include wider range of (organochlorine) pesticides.

The author recommends that, at a minimum, sediment quality guidelines for the metals identified in this report and a suite of 16 polycyclic aromatic hydrocarbon isomers identified as priority pollutants by the United States Environmental Protection Agency be included in the revised National Action List. Other organic chemicals that are frequently important contaminants of sediment and which can be measured by a number of laboratories in South Africa include polychlorinated biphenyls and various organochlorine and organophosphate pesticides. The Stockholm Convention on Persistent Organic Pollutants can also be used to guide what additional chemicals could be included in the revised National Action List. Another frequent, toxic contaminant of sediment from ports is tributyltin and consideration should be given to its inclusion, either as tributyltin or tin, considering that most dredging in South Africa takes place in ports.

The costs of measuring organic chemicals in the laboratory are considerably higher compared to metals. An important implication then of including sediment quality guidelines for a wider suite of chemicals in the revised National Action List is that the costs for sediment chemistry assessment will increase if all chemicals are monitored. However, options for paring down the suite of organic chemicals monitored can be explored. For example, a minimum suite of chemicals could be prescribed for measurement in all sediment samples, and a wider suite in a proportion of the samples (e.g. 20%) that are judiciously positioned in the study area (e.g. in areas where historically high contaminant concentrations have been measured, or in areas where the sediment is comprised of a high mud fraction). Alternately, measurement of a wider suite of chemicals could be required only for sediment samples that contain chemicals from the minimum suite at concentrations that exceed a Level I guideline. Ideally, as wide a suite of chemicals possible should be measured in

sediment from urban coastal environments on a 3 - 4 year basis, to determine whether a more comprehensive suite of chemicals should be monitored routinely. The responsibility for implementing and funding this type of assessment should be responsibility of the state (either nationally or provincially), but major dredging proponents have a role to play (e.g. Transnet National Ports Authority). Lastly, consideration will need to be given in future to so-called 'emerging' chemicals, including personal care products. These chemicals are increasingly becoming a source of concern to environmental scientists and regulatory authorities in most regions of the world, not least because many are known or strongly suspected endocrine disruptors. However, there are presently no sediment quality guidelines for these chemicals and in many cases their toxicological significance is poorly understood.

Key issue:

9. Should the sediment quality guidelines of the revised National Action List include guidelines for chemicals other than the suite of metals identified in this report and, if so, which chemicals should be included?
10. Should the option of defining a minimum suite of chemicals that are measured in all sediment samples, and a wider suite that are measured in exceptional cases, be explored? If so, which chemicals should comprise the minimum and wider suites? Note that stakeholders may wish to specify minimum chemical lists for ports, urban areas and rural/remote areas separately.

22. Decision-making for the revised National Action List

As discussed elsewhere in this report, an Action List is a mechanism that allows managers and regulatory authorities to reach an informed decision on the suitability of waste material proposed for disposal in the marine environment. To facilitate decision-making, the criteria against which the decision is made need to be defined. This section provides a discussion on issues and options in this context.

22.1 Narrative intent of the Level I and Level II guidelines

The author presumes that the revised National Action List will comprise two Action Levels for deciding whether dredged sediment is (a) suitable for unconfined, openwater disposal without further testing, (b) must undergo further testing before a decision can be made, or (c) is unacceptable for unconfined, openwater disposal and requires special management.

Table 20. The USACE (2006) sediment quality guidelines. $\text{mg.kg}^{-1}\text{-OC}$ = screening levels are normalized to the fraction of organic carbon in sediment, - = no numeric guideline defined for chemical.

Chemical	Screening Level 1 (dry weight)	Screening Level 2 (dry weight)	Screening Level 1 ($\text{mg.kg}^{-1}\text{-OC}$)	Screening Level 2 ($\text{mg.kg}^{-1}\text{-OC}$)
Metals (mg.kg^{-1})				
Antimony	150	150	-	-
Arsenic	57	93	-	-
Cadmium	5.1	6.7	-	-
Chromium	260	270	-	-
Copper	390	390	-	-
Lead	450	530	-	-
Mercury	0.41	0.59	-	-
Silver	6.1	6.1	-	-
Zinc	410	960	-	-
Polycyclic Aromatic Hydrocarbons ($\mu\text{g.kg}^{-1}$)				
Total Low Molecular Weight PAH's	5 200	5 200	370	780
Naphthalene	2 100	2 100	99	170
Acenaphthylene	560	1 300	66	66
Acenaphthene	500	500	16	57
Fluorene	540	540	23	79
Phenanthrene	1 500	1 500	100	480
Anthracene	960	960	220	1 200
2-Methylnaphthalene	670	670	38	64
Total High Molecular Weight PAH's	12 000	17 000	960	5 300
Fluoranthene	1 700	2 500	160	1 200
Pyrene	2 600	3 300	1 000	1 400
Benz(a)anthracene	1 300	1 600	110	270
Chrysene	1 400	2 800	110	460
Benzo(a)fluoranthene(b+k)	3 200	3 600	230	450
Benzo(a)pyrene	1 600	1 600	99	210
Indeno(1,2,3-c,d)pyrene	600	690	34	88
Dibenz(a,h)anthracene	230	230	12	33
Benzo(g,h,i)perylene	670	720	31	78
Chlorinated Hydrocarbons ($\mu\text{g.kg}^{-1}$)				
1,4-Dichlorobenzene	110	110	3.1	9
1,2-Dichlorobenzene	35	50	2.3	2.3
1,2,4-Trichlorobenzene	31	51	0.81	1.8
Hexachlorobenzene	22	70	0.38	2.3
Phthalates ($\mu\text{g.kg}^{-1}$)				
Dimethyl phthalate	71	160	53	53
Diethyl phthalate	200	200	61	110
Di-n-butyl phthalate	1 400	1 400	220	1
Butylbenzyl phthalate	63	900	4.9	64
Bis(2-ethylhexyl) phthalate	1 300	1 900	47	78
Di-n-octyl phthalate	6 200	6 200	58	4 500
Phenols ($\mu\text{g.kg}^{-1}$)				
Phenol	420	1 200	-	-
2-Methylphenol	63	63	-	-
4-Methylphenol	670	-	-	-
2, 4-Dimethylphenol	29	29	-	-
Miscellaneous Extractables ($\mu\text{g.kg}^{-1}$)				
Benzyl alcohol	57	73	-	-
Benzoic acid	650	650	-	-
Dibenzofuran	540	540	15	58
Hexachlorobutadiene	11	120	3.9	6.2
N-Nitrosodiphenylamine	28	40	11	11
Pesticides ($\mu\text{g.kg}^{-1}$)				
p,p'-DDE	16	-	-	-
p,p'-DDD	9	-	-	-
p,p'-DDT	34	-	-	-
Poly-Chlorinated Biphenyls ($\mu\text{g.kg}^{-1}$)				
Total PCB's	130	1 000	12	65
Tributyltin				
TBT pore water ($\mu\text{g.l}^{-1}$)	0.15	-	-	-

The author further presumes that chemical concentrations falling between the Level I and Level II guidelines will be taken as an indication that there is uncertainty whether the chemicals are associated with adverse effects to sediment-dwelling organisms and thus that further assessment will be required for decision-making. If so, then the criteria (or rules) that will trigger the need for further assessment based on sediment chemistry/ sediment quality guidelines need to be defined (options for further assessment are discussed in a separate section below). Lastly, the author presumes that the intent is to use the Level II guideline on an automatic fail (prohibition) basis if exceeded (i.e. for decision (c) above).

Before discussing decision-making criteria and options that require clarity and consensus, it is necessary to reiterate that a modicum of common sense and/or best professional judgment is often appropriate for deciding whether further assessment is required when contaminant concentrations exceed a Level I guideline. For example, the marginal exceedance of a Level I guideline by a single metal concentration, or even the marginal exceedance of a few guidelines by different metals, might not necessarily need to trigger further assessment if there are extenuating circumstances. These circumstances include, for example, if the area over which metal concentrations are elevated in sediment is small (i.e. a small volume relative to total dredging volume). This said, the main reason for formulating an Action List is to provide a mechanism for reaching an informed decision in a consistent and transparent manner. Best professional judgment decision-making requires experience and is open to misapplication in the case of an inexperienced decision-maker, and in the extreme case to abuse. Judgmentally-based decisions should not, therefore, be made lightly, and as many lines of evidence should be used to motivate why further assessment is not required in 'borderline' cases. If there is any doubt, then decisions should be based on the decision criteria of the revised National Action List and in concordance with the precautionary principle of the London Protocol.

Defining decision criteria for the Level II guideline will not be necessary if the intent is to use this guideline on an automatic fail (prohibition of unconfined, openwater disposal) basis if exceeded by a single chemical. The Level II guideline can, however, only be used on this basis if the Level II guideline of the sediment quality guidelines adopted for the revised National Action List are predictive that, when exceeded, there is a very high probability for adverse effects to sediment-dwelling

organisms. There is obviously little point using on this basis a Level II guideline derived to predict about 50% probability for adverse effects to sediment-dwelling organisms (e.g. the Effects Range Median of the Long et al. (1995) sediment quality guidelines). This will lead to sediment being deemed unsuitable for unconfined, openwater disposal, when it may in fact not pose a significant risk to sediment-dwelling organisms. As stated elsewhere in this report, most workers caution against the definitive (i.e. pass/fail) use of sediment quality guidelines. This said, many jurisdictions/countries do use a Level II guideline on this basis, but some (e.g. USACE 2008) do not use it definitively and permit dredging proponents to determine whether or not adverse effects are manifesting if the Level II guideline is exceeded. The major challenge for assigning decision criteria to the Level II guideline in a South African context if the guideline is not used in a definitive manner is what further assessment is possible. As mentioned elsewhere in this report, the state of science with regard to biological assessment of sediment quality in South Africa is so poor that there are few options for further assessment beyond those that would need to be implemented based on decisions for the Level I guideline.

Key issue:

11. Should the Level II guideline be used in a definitive manner, that is, if a single metal concentration in a sediment sample exceeds this guideline then the dredged material represented by the sample is automatically considered unsuitable for unconfined, openwater disposal?

22.2 Should some metals be 'weighted' more than others in decision-making?

Although all metals are toxic at elevated concentrations, certain metals are more frequently implicated with risks to ecological and human receptors. It is for this reason that the London Convention and London Protocol define so-called Annex I and Annex II metals. As a reminder, the Annex I metals are mercury and cadmium, while all other metals fall into Annex II. Mercury is of particular concern because of the risk it poses to ecological and human receptors through biomagnification. In the United States of America, which has the most comprehensive database in the world, mercury is by far the most important contaminant/chemical responsible for the issuing of fish consumption advisories. It is in fact the only metal of the five contaminants that were responsible for 97% of the fish consumption advisories issued in 2008 (latest year for which nationwide data are available). The remaining

chemicals were all of an organic type, including polychlorinated biphenyls and organochlorine pesticides.

There is undoubtedly strong motivation to weight mercury and other chemicals that have a high bioaccumulation and biomagnification potential more so than others for decision-making. This said, many jurisdictions/countries consider all chemicals equivalently in terms of guideline exceedance. Some jurisdictions/countries do, however, include bioaccumulation guidelines for the precise reason that bioaccumulative chemicals are a significant concern. Other jurisdictions/countries do not specifically incorporate a bioaccumulation factor into sediment quality guidelines but have developed tissue residue guidelines aimed at protecting wildlife species (e.g. Canada, CCME 2002) and humans that consume aquatic organisms. Many jurisdictions/countries also require that laboratory-based bioaccumulation testing be performed if chemicals with a high bioaccumulation and biomagnification potential are measured in sediment at a concentration that warrants concern.

If a decision is made to include a wider suite of chemicals in the sediment quality guidelines, then consideration will need to be given to chemicals that are persistent and bioaccumulative.

Key issue:

12. Should all chemicals included in the revised National Action List be weighted equivalently in terms of decision-making? If not, which chemicals should be considered more important than others?
13. Should sediment bioaccumulation guidelines be considered for the revised National Action List, or should bioaccumulation be monitored through other dedicated monitoring programmes for dredging area?

22.3 Chemical mixtures

As discussed elsewhere in this report, sediment quality guidelines are derived for individual contaminants. More often than not, however, contaminants occur as complex mixtures in sediment. The significance of their occurrence in complex mixtures is that contaminants may interact to effect toxicity in three ways, namely less than additive (antagonistic), more than additive (synergistic) and additive. If contaminants have a similar mode of toxic action, then dose (or concentration)-additive toxicity is typically hypothesised (Cassee et al. 1998, USEPA 2000). This assumption appears to be well met within classes of organic contaminants, including

polycyclic aromatic hydrocarbons. Joint-toxicity of chemicals with dissimilar toxic action is usually hypothesised to be independent and elicit response-addition toxicity (Broderius 1991, Faust et al. 2000). However, organisms may respond to contaminant mixtures in unexpected ways because individual contaminants sometimes interact, modifying the overall magnitude or nature of toxicity (Cassee et al. 1998). These non-additive toxicant interactions, expressed as synergisms or antagonisms, pose a significant challenge to hazard assessment. Research suggests that interactive effects are common, at least within and between certain chemical classes with different modes of toxic action. Synergisms between insecticides and herbicides appear to be frequent (Pape-Lindstrom and Lydy 1997). Norwood et al. (2003) concluded that synergisms and antagonisms for metals are more common than response-addition toxicity. However, the toxicity of metal mixtures cannot reliably be predicted based on the toxicity of individual metals - additive, synergistic, or antagonistic interactions may result (Franklin et al. 2002, Norwood et al. 2003). Metals and polycyclic aromatic hydrocarbons have dissimilar toxicology, but several studies suggest co-occurrence may elicit complex, interactive effects (Gust 2005, Fleeger et al 2007).

Considering these complexities and our relatively poor understanding of how chemicals interact in sediment to influence toxicity, whether it is realistic to consider an additive guideline for the revised National Action List is debatable. The author could not find a single additive sediment quality guideline in the scientific literature, which probably indicates that such an approach is unsuitable. An approach that has, however, received considerable attention in the context of chemical mixtures and which assumes additivity is the so-called mean sediment quality guideline quotient approach (Long et al. 1998). This approach has been shown to increase predictive ability (Long et al. 1998, MacDonald et al. 2000, Faurey et al. 2001).

A mean sediment quality guideline quotient is determined by calculating the arithmetic mean of the quotients for various contaminants, which are themselves calculated by dividing the concentrations of chemicals by their respective sediment quality guidelines. Typically, the Level II guideline is used for quotient calculation. The result is a single, unitless, effects-based index of the relative degree of contamination that can provide a basis for determining the likelihood that a sediment sample will be toxic to sediment-dwelling organisms (Long et al. 2006). The quotient incorporates the number of chemicals

exceeding a sediment quality guideline as well as the magnitude of the exceedance. The inherent assumption of mean sediment quality guideline quotient approach is that the contributions of each chemical to toxicity are additive. That is, the assumption is that the chemicals in the sample are not acting antagonistically or synergistically. This approach provides a number of advantages for interpreting the significance of complex mixtures of potentially toxic substances in sediments, including that it summarises complex data into a single value that can then be directly compared between sites.

As with all approaches, the mean sediment quality guideline quotient approach has limitations. The ability of the mean sediment quality guideline quotient to predict the likelihood for adverse effects is dependant on the number and type of sediment quality guidelines that are used to calculate the quotient (Fairey et al. 2001). Mean sediment quality guideline quotients can also be a relatively poor predictor of toxicity when applied to sediments containing high concentrations of a single chemical, since other chemicals at low concentrations drive the mean quotient downward (Vidal and Bay 2005). However, the single most significant hindrance to the use of the mean sediment quality guideline quotient for the revised National Action List is that the relationship between mean quotients and adverse effects to sediment-dwelling organisms would be unknown irrespective of the guidelines adopted. Thus, the predictive ability of quotient values has not been established for local waters and will not be established for some time if the existing state of science in South Africa is considered. Obviously, rating criteria attached to the sediment quality guidelines adopted, but three of the four candidate sediment quality guidelines have no guidance on their use in this context. Only the British Columbia sediment quality guidelines incorporate this approach. Of course, we could simply use the approach that a mean sediment quality guideline quotient of ≥ 1 is indicative of a high probability for adverse effects to sediment-dwelling organisms. The mean sediment quality guideline quotient could also be calculated using Level I and/or Level II guidelines. In the case of quotients calculated from Level II guidelines, the intent would then be to identify sediment that should not be considered suitable for unconfined, openwater disposal.

Key issue:

14. Should chemical mixtures be considered? If so, should this be on an additive basis or through a mean sediment quality guideline quotient approach?

22.4 Decision criteria

As stated elsewhere in this report, it is not the authors' responsibility to formulate the sediment quality guideline component of the revised National Action List or the decision-making criteria linked to the list. The author has, however, considered it prudent to provide some options in this context.

In many jurisdictions/countries, decision criteria associated with sediment quality guidelines for dredging projects are straightforward. In most cases, exceedance of a single Level I guideline triggers the need for further assessment, while a single exceedance of a Level II guideline indicates that the dredged material is unsuitable for unconfined, openwater disposal.

However, many workers consider the exceedance of a single empirically-based sediment quality guideline to be an unreliable indicator of toxicity/adverse effects (e.g. Long et al. 2000, Fairey et al. 2001, Vidal and Bay 2005). The presence of many contaminants in a sediment sample and the often high degree of correlation amongst them indicates that most empirically-based sediment quality guidelines should not be used in isolation, but rather in combination to provide an overall indication of the potential for adverse effects. It is for these reasons that some jurisdictions follow a complex decision-making process and consider the potential effects of chemical mixtures using the mean sediment quality guideline quotient approach. As an example, the decision criteria associated with the sediment quality guidelines of British Columbia (MacDonald et al. 2003) are provided below. The reader will recall that the sediment quality guidelines comprise two guidelines (these are referred to as criteria), for so-called Sensitive Contaminated Sites (Level I) or Typical Contaminated Sites (Level II).

A sensitive or typical site is a contaminated site if any of the following conditions exist:

- The 90th percentile concentration of one or more chemicals of concern equals or exceeds their respective Sediment Quality Criteria for Sensitive/Typical Contaminated Sites (i.e. 9 of 10 measurements must be below the Sediment Quality Criteria to designate a site as uncontaminated) and exceeds upper limit of background for that substance (i.e. mean + 2 standard deviations),
- The concentration of one or more analytes exceeds their respective Sediment Quality Criteria for Sensitive/Typical Contaminated Sites by a factor of two or more in any sediment sample and exceeds upper limit of background for that substance (i.e. mean + 2 standard deviations),

- The 90th percentile mean Sediment Quality Criteria for Sensitive/Typical Contaminated Sites quotients for the contaminant mixture equals or exceeds 1.0,
- The mean Sediment Quality Criteria for Sensitive/Typical Contaminated Sites quotients for the contaminant mixture in any sediment sample equals or exceeds 2.

The level of straightforwardness or complexity of decision criteria associated with the revised National Action List needs to be reached through consensus but, importantly, also requires consideration of several issues related to the sampling design used for sediment collection. The British Columbia sediment quality guideline example above illustrates that a decision is not made from data for a single sediment sample, but rather on the percentile distribution across a number of sediment samples (in this case 10 samples). This and other factors that require consideration in this context are discussed below.

23. Some factors that influence decision criteria

Although this report focuses predominantly on the use of sediment chemistry for reaching decisions on the suitability of sediment identified for dredging for unconfined, openwater disposal, other factors should be considered for decision-making. Thus, the revised National Action List should comprise more than sediment quality guidelines and associated biological assessment. Further, how decisions are made can be influenced by the manner in which sediment samples are collected and used to characterise the dredging area.

23.1 Nature of the dredged material

In many jurisdictions/countries, the assessment process begins by considering the grain size composition and total organic content of the material identified for dredging. Many jurisdictions/countries exempt sediment comprised predominantly of sand and gravel from detailed assessment (predominantly usually defined as above about 85% contribution to bulk sediment, and sand classified as that retained on a No 200 sieve (mesh diameter of 63 µm)). In these cases, the focus is on the environmental impacts of the physical processes of dredging the material and its disposal. The rationale for excluding sediment comprised predominantly of sand is that, as discussed elsewhere in this report, the relatively large surface area to volume ratio of fine-grained particles and environmental characteristics of areas where fine-grained particles settle out of the water column, contaminants are generally associated with silts

and clays. Sand is, in contrast, unable to sequester contaminants due to its relatively small surface area to volume ratio and 'uncharged' surface. Therefore, it is highly unlikely that sediment comprised predominantly of sand will accumulate contaminants to a degree that these pose an ecological risk.

Other exemption criteria include previously undisturbed geological material and small volumes of sediment. With regard to the latter criterion, small is typically taken as in the order of about 1000 cubic meters.

23.2 Nature of dredging site

Sediment dredged from minimally anthropogenically impacted areas is often also excluded from a detailed assessment. For example, the dredging of sediment from a marina in an estuary surrounded by a small village is likely to meet these criteria as there is unlikely to be a high load of contaminants and the volume of sediment that requires dredging is likely to be small. In such a situation there is little point in burdening the local authority and/or the dredging proponent with the costs of performing a detailed chemical assessment because there is a high probability that chemical concentrations will not exceed the Level I guidelines. In contrast, most ports in South Africa will require a detailed assessment on a yearly basis.

23.3 Disposal site characteristics

Other criteria include the nature of the disposal site, primarily whether it is of a dispersive or non-dispersive type. There are both disadvantages and advantages in this context. Dispersive sites are generally found in high-energy hydrodynamic environments and, because of strong currents and other forms of turbulence, are unlikely to contain and retain fine-grained sediment and are therefore unlikely to be contaminated with, or retain, particle-reactive contaminants. Obviously, if dredged material that is winnowed from the site accumulates predominantly in a specific offsite area, then there may be continual deposition of contaminated sediment at this site and this may impact on benthic fauna and other organisms. However, there is often such a large volume of sediment moving through the area by bedload transport that this is unlikely to be a problem. At non-dispersive (or depositional) sites, in contrast, the hydrodynamic environment is of a low-energy nature and fine-grained dredged material accumulates at the site. If this material is significantly contaminated, then the risk exists for organism exposure. However, these types of sites are amenable to capping and other types of contaminated sediment containment.

Key issue:

15. What factors other than sediment chemistry and biological assessment should be considered for decision-making?

23.4 Sampling design considerations

The sampling design for sediment quality assessments needs to be resolved. This report does not consider the numerous, critical aspects of sediment sampling campaigns and laboratory assessment. These should be addressed within a dredged material assessment framework, of which the revised National Action List will comprise a part. However, several factors associated with sediment sampling campaigns will have an important bearing on decision criteria associated with the Action Levels of the revised National Action List. These are discussed below.

23.4.1 Consideration of dredged material management units

For dredging projects in the United States of America, the US Army Corp of Engineers uses the concept of dredged material management units to facilitate decision-making. A dredged material management unit is the smallest volume of dredged material that is truly dredgeable (i.e. capable of being dredged independently from adjacent sediments) and for which a separate disposal decision can be made. Thus, a given volume of sediment can only be considered a dredged material management unit if it is capable of being dredged, evaluated and managed separately from all other sediment in the project area.

Although a dredged material management unit will have a minimum spatial extent, there is no upper limit to the size of a dredged material management unit. Sediment from each dredged material management unit is independently evaluated to determine whether it is suitable for unconfined, openwater disposal. A dredged material management unit can be identified on the basis of the grain size composition of the sediment. The rationale for identifying a dredged material management unit on the basis of sediment grain size is that sediments with a similar physical composition will have a similar potential for contaminant accumulation. Of course, in an environment such as a port, sediment of a similar grain size composition (e.g. mud) is likely to accumulate over a large area of the port but may not be comparably contaminated. For these reasons, other dredged material management unit definition considerations include the type and concentrations of chemicals that have previously been measured in the sediment (if a

sediment quality assessment has been performed in the area previously) and proximity to known sources of contaminants. Dredged material management units should be identified before (detailed) fieldwork is performed, since this will define the sampling design that will be implemented (e.g. location and number of sediment sampling sites). Although this might require a pre-survey that focuses on, for example, the grain size composition of sediment, these types of surveys are relatively inexpensive and can provide valuable information on how to define dredged material management units. Ultimately, such surveys may provide a cost saving.

An alternate sampling design that is frequently implemented but is conceptually similar to the dredged material management unit type of design is some form of stratified (including random-stratified) sampling. This type of sampling divides the area of interest into predefined areas (or strata), usually of a square or hexagonal shape. These are the strata, and each stratum is then sampled, or a few strata are sampled but those sampled are identified using random tables. This type of sampling design is particularly powerful for determining spatial trends. The size of strata can vary, so that the sampling density is higher in locations where the probability of sediment contamination is greatest. For example, in Hong Kong the dredging area is divided into a grid of square shaped strata, with the maximum size of 200 m x 200 m in sediment where contamination is expected or known (from previous data) to be low, and 50 m x 50 m for sediment where a higher level of contamination is expected or known to occur (ETWB 2002).

Although the subject of sampling design for dredging projects does not fit directly into the revision of the National Action List, it is discussed here since the sampling design influences decision-making, especially from a representivity perspective.

23.4.2 Number of samples, compositing of samples, and re-sampling

The number of sediment samples and the number of laboratory analyses required to characterise a dredged material management unit (or any other concept that may be used in South Africa) must be clarified. The usual approach in South Africa is to collect a single sediment sample from a site and to analyse the sample in the laboratory for a suite of chemicals. The inherent assumption in this approach is that the magnitude of contamination is uniform in the vicinity of the site. However, the 'vicinity of the site' is hardly ever explicitly

defined, but rather depends on the nearest neighbour site. The distance between some sediment sampling sites in South African ports may range from less than 100 meters to almost 1000 meters. Although the assumption of uniformity between sites might well apply in some situations, when sites are situated far apart this assumption almost certainly does not hold true. It is in these disparate situations where the definition of dredged material management units and the number of sediment samples to characterise such a unit require consideration.

The measurement of chemicals (or any other parameters) in a single sediment sample collected from a dredged material management unit assumes that this sample is representative of the entire unit. However, there are two possible scenarios that may arise. Firstly, the single sediment sample might be collected from a part of the dredged material management unit where there is significant but highly localised contamination. In contrast, the sediment sample might be collected from a part of the dredged material management unit where the level of contamination is far lower compared to the remainder of the unit. Either of these situations has a greater probability of being realised as the surface area of the dredged material management unit increases. Nevertheless, both situations will create an inaccurate understanding of the degree of contamination of the sediment and may influence management decisions. In the first scenario, the failure to properly characterise the dredged material management unit will penalise the dredging proponent, since the regulatory authority may require an alternate and costly approach to dredging and disposal of the sediment. In the second scenario, the failure to properly characterise the dredged material management unit may pose unacceptable risks during the dredging process and at the dredged sediment disposal site.

A logical strategy to overcome these potential scenarios is to define dredged material management units based on the grain size composition of the sediment and the volume of sediment that requires dredging. Historical monitoring data and available information on potential sources of contaminants to the dredged material management unit can be used as a basis for deciding whether it should be divided into smaller dredged material management units. There obviously needs to be a minimum size for a dredged material management unit, based on dredgeability. To fully characterise the dredged material management unit, composite rather than single sediment samples could be collected. This is, for example, the approach adopted by the US Corps of

Engineers in the USA. Composite sample are generated by collecting a similar volume of sediment from a number of sites uniformly distributed across a dredged material management unit, combining the sediment, homogenising it, and then removing an aliquot for laboratory analysis. The obvious benefits of this approach are that the composite sample provides a better estimate of the average concentration of contaminants in sediment over the dredged material management unit, and that fewer sediment samples need to be analysed in the laboratory and hence the costs for analysis are lower. The number of separate samples required for a composite sample needs to be determined on a case-by-case basis, but power analysis can be used for this purpose. Alternately, dredged material volume based numbers of sediment samples identified by other jurisdictions/countries can be used (e.g. Chevrier and Topping 1998). The most important disadvantages of this approach are that because of the averaging effect of the composite sample, small but highly contaminated areas of sediment might not be detected, and the time required to collect additional sediment samples for the composite sample has cost implications because fieldwork teams will need to remain in the field for longer periods.

There is obviously an option to collect and analyse single sediment samples, and to then resample in the vicinity of sites where high contaminant concentrations were measured in order to characterise the extent of the contaminated sediment. The obvious disadvantage of this approach is the increased costs associated with the redeployment of the fieldwork team and the analysis of additional sediment samples.

Key issue:

16. Should the concept of dredged material management units be adopted? If so, what factors will be used to determine the spatial extent of a dredged material management unit?
17. Should a site, or dredged material management unit if this concept is adopted, be represented by a single or a composite sediment sample?

23.4.3 Depth of sediment sampled

In situ sediment quality assessment programmes commonly only sample the surface 5 - 10 cm of the sediment. This is generally regarded as the most biologically active zone, and hence represents the zone of immediate interest. For dredging programmes, however, assessment typically requires that deeper layers of sediment be sampled and analysed, for the

reason that dredging operations are not restricted to the surface few centimetres of the sediment and the deeper layers may be contaminated. This said, many regulatory authorities only require the collection and analysis of surface sediment in areas that are frequently (e.g. maintenance) dredged.

The main implication of sampling deeper layers of sediment are that the costs will increase, since the collection of this sediment requires some form of coring.

Key issue:

18. Should sediment samples only be collected from the surface, or should deeper sediments also be sampled?

24. Options for further assessment

If the concentrations of chemicals in sediment fall between Level I and Level II guidelines then most jurisdictions/countries require that the sediment be tested further before a decision is made on its suitability of for unconfined, openwater disposal. Further assessment can be approached from two, not mutually exclusive perspectives, namely chemical and biological. This section provides a brief overview of these approaches and identifies possible tests that could be implemented in South Africa. As stated elsewhere in this report, the existing state of science in terms of biological assessment of sediment quality in South Africa is poor, meaning that few of the approaches used in other jurisdictions/countries can be applied locally.

24.1 Chemical assessment

As discussed previously, the measurement of chemicals in sediment provides an understanding of whether the sediment is contaminated. Sediment chemistry does not, however, provide an understanding of whether the measured chemicals are in a bioavailable form. Taking metals as an example, a proportion of the metal concentration measured in the laboratory will have been released from the sediment crystal lattice during acid digestion. Clearly, metals bound in the crystal lattice of sediment are not in a bioavailable form and hence cannot exert toxicity.

Although biological assessment is required to determine whether the metals are in a bioavailable form, several chemical procedures have been developed to estimate whether impacts associated with the dredging process and the disposal of dredged sediment are likely to manifest. The most common approach is the sediment elutriate test (this test also has application for biological

assessment, as discussed below). The test was designed and developed by the United States Environmental Protection Agency and US Corps of Engineers to provide a technically more appropriate alternative to sediment chemistry for evaluating the potential adverse effects of dredged sediment on water quality. The test simulates the release of contaminants from sediment during hydraulic dredging-hopper disposal operations.

In short, the test entails the mixing of one volume of sediment with four volumes of water in the laboratory. This mixture is then either agitated with compressed air or in a sealed vessel on an automatic shaker for 30 minutes, and allowed to settle for one hour. The supernatant is then decanted, filtered and/or centrifuged, and concentrations of targeted chemicals in the supernatant are then measured.

The United States Environmental Protection Agency guidelines for assessing the impact of dredged material disposal on water quality require that applicable marine water quality criteria or other toxicity criteria specified not be exceeded after allowing for initial mixing. Initial mixing is defined as that: 'which occurs within four hours after dumping' (USEPA 1992). The extent of initial mixing is determined using a mathematical model if sufficient data are available, or by other means if insufficient data are available (e.g. 'The liquid and suspended particulate phases of the waste may be assumed to be evenly distributed after four hours over a column of water bounded on the surface by the release zone and extending to the ocean floor, thermocline or halocline if one exists, or to a depth of 20 m, whichever is shallower.' (a dilution factor can readily be calculated from such data)). The elutriate test obviously uses a sediment dilution of 1:4, which greatly overestimates water quality impacts because within a four hour period dilutions would normally be hundreds of times greater, especially for dispersive site. The test data must therefore be entered into the model, if available, or multiplied by an appropriate dilution factor after four hours in order to assess whether or not the water quality criteria will be exceeded after disposal.

The principle advantage of the sediment elutriate test is that it is relatively simple to perform. The principle disadvantage is that low detection limits are required for some contaminants. This means that the analyses can be expensive.

As second chemical assessment approach is the Simultaneously Extracted Metal/Acid Volatile Sulphide approach, although this is specific only to metals. The

theory relating to this procedure and how it is used to assess sediment quality was discussed elsewhere (see section 9.1.3)

Key issue:

19. Should the sediment elutriate test be considered as a further assessment tool?
20. Should the Simultaneously Extracted Metal/Acid Volatile Sulphide approach be considered as a further assessment tool?

24.2 Biological assessment

The ultimate concern in any sediment quality assessment is whether biological resources are being adversely impacted by the accumulation of contaminants in sediment. Various biological lines of evidence for reaching a decision on whether dredged sediment is suitable for unconfined, openwater disposal have been developed. These include whole sediment toxicity testing, sediment elutriate toxicity testing, bioaccumulation testing, and analysis of the composition of benthic invertebrate communities.

Whole sediment toxicity testing, sediment elutriate toxicity testing, and bioaccumulation testing are commonly used for dredging assessments (e.g. Torres et al. 2009), but at two different stages (tiers) in the decision-making framework. In many jurisdictions/countries, sediment identified for dredging and in which the concentrations of chemicals are such that further testing is required, the sediments toxicity must be tested using a suite of tests that consider both acute and chronic toxic effects. Unfortunately, and as mentioned on several occasions elsewhere in this report the state of science with regard to the biological assessment of sediment quality in South Africa is poor. Presently, only one toxicity test is fairly frequently used for assessing the toxicity of sediment porewater and elutriates, namely the sea urchin fertilisation test. Although this test can be used for this purpose, there is still a need to evaluate the influence of numerous confounding factors on test results. These include the influence of ammonia and sulphides, which are both highly toxic to organisms at elevated concentrations. A further limitation of this test is that sea urchins in a spawning condition are not always available, due to season reproduction.

No whole sediment toxicity or bioavailability tests have been developed for application in South Africa. Considering that the development of scientifically defensible toxicity tests is a lengthy process (minimum of two year but up to five years), there is little probability

that other toxicity tests will be available for testing sediment in the near future unless test organisms used in other countries are imported and used in South Africa. The legal implication of importing and culturing these organisms in South Africa is uncertain but warrants investigation. There is of course the option of sending sediment samples to international laboratories for toxicity testing, but this will have significant financial implications. For example, the costs for various sediment toxicity tests in the state of California in the USA range between about R4 800 to R11 200. These costs exclude the costs of transporting sediment to the testing laboratory.

Although analysis of the composition of benthic invertebrate communities is a widely used tool for assessing sediment quality, the state of this science in South Africa is poor. The primary reason is that almost all South African benthic ecologists do not perform research in pollution related issues, and therefore have not tailored their research to address the requirements of pollution detection through benthic invertebrate community composition. Thus, benthic response indices that can be used to identify when these communities have been adversely affected through pollution have not been developed. A significant limitation of using the composition of benthic invertebrate communities to detect adverse effects in sediment from ports is that sediment from ports is routinely disturbed through dredging and propeller wash. In other words, the benthic invertebrate communities too are frequently disturbed by these impacts, to the extent that it becomes difficult to discriminate between pollution and propeller wash and dredging induced impacts. Nevertheless, the use of benthic invertebrate community composition as a tool for assessing sediment quality should be considered. However, it is improbable whether this should be included within the National Action List.

Key issue:

21. What form of biological assessment is required in situations where chemical concentrations fall between the Level I and Level II guidelines?

25. National Action List terminology

Although a seemingly innocuous consideration, appropriate terminology for the National Action List must be defined. Some jurisdictions/countries have used the qualifier interim for Action Lists and/or sediment quality guidelines (e.g. Interim Sediment Quality Guidelines). The author is not of the opinion that the term 'interim' should be used since this implies that the National Action List or the sediment quality guidelines

are of a provisional or temporary nature and, in time, will be formalised. The reality is that as and when new data becomes available, and new skills and tools for performing sediment quality assessments in South Africa are developed, the National Action List should be continually revised. In fact, the author strongly recommends that consideration be given on an annual basis as to whether the National Action List can or should be revised. The National Action List could, in these circumstances then be permanently considered as of an interim nature.

Stakeholders will need to make a decision on the terminology used to describe the National Action List, and some suggestions include Dredged Material National Action List, or a Dredging Assessment National Action List.

Suitable terms for the sediment quality guideline component of the National Action List will also need to be defined. It is the authors option that terms such as Action Level, Special Care Level and Prohibition Level should be avoided. Sediment quality guidelines comprise one component of an Action Level, and hence should not themselves be referred to as an Action Level unless the sediment quality guidelines are the only component of the Action Level. The terms Special Care Level and Prohibition Level are loose and definitive respectively. The term prohibition is in fact so definitive that it implies that no further consideration will be given. But, as stated elsewhere in this report sediment quality guidelines are not definitive decision-making tools, but rather should be used in a weight of evidence approach. Although prohibition may be invoked followed a sediment quality assessment, this should not be made against a single measurement. In many cases, the simplest terms are the most apt. For example, the guidelines could be referred to as levels (e.g. Level 1) or as lower and upper guidelines.

Key issue:

22. Stakeholders are requested to recommend terms for the National Action List *per se* and for the Level I and II sediment quality guidelines. Should stakeholders consider the need for more than two sediment quality guidelines, they are requested to recommend terms in this context.

26. 'Treatment' considerations

If sediment is deemed so contaminated that it is unsuitable for unconfined, openwater disposal, alternative disposal or treatment options need to be considered. As mentioned previously, whether South

Africa is in a position to prohibit the unconfined, openwater disposal of dredged sediment is debatable. Presently we do not have the technical expertise and especially the financial resources to consider alternate disposal or remediation options, at least not for large volumes of sediment. It is inevitable, therefore, that we will either need to ensure that sediment does not reach a stage where it becomes so contaminated that it cannot be dredged, or alternate management strategies will need to be identified. The challenges facing our country should not be seen as a 'free ticket' to the disposal of contaminated sediment. This will render the South African government in breach of international treaties and conventions, but also endangers our natural resources and ultimately our citizens, from health as well as socio-economic perspectives.

One management strategy that may be possible is sediment blending, wherein contaminated sediment is mixed with uncontaminated sediment. Thus, concentrations of contaminants in the sediment are 'diluted' with clean(er) material. This will certainly be a challenge in small ports, however, such as Cape Town, Port Elizabeth and East London. The reason is that there might not be sufficient 'clean' sediment in these ports for this purpose. This will then either require the dredging of sediment from sand traps, or from offshore areas. Either way, this will pose problems. In most cases, sediment dredging from sand traps is used for shoreline nourishment. Dredging sediment from offshore areas may in turn require an Environmental Impact Assessment. These considerations should be sufficient warning to organisations that the implications of sediment contamination are potentially significant from a financial perspective.

Any decisions and strategies in this context are beyond the scope of this report, but are identified because they will require consideration in future. Branch Oceans and Coasts of the Department of Environmental Affairs should consult experts in the field of dredging at each port in South Africa to determine which 'treatment' options are feasible in the event that dredged material is so contaminated that its unconfined, openwater disposal is likely to pose significant and unacceptable risks in the receiving environment.

27. National dredging assessment framework

The revised National Action List will not comprise a dredging assessment framework, but will need to be incorporated within such a framework. The objective of a dredging assessment framework is to define the

procedures that will be followed to determine management options for dredged material. As such it provides a predictable and consistent procedure to the evaluation of dredging assessment. Besides including the National Action list, which is one of the tools for decision-making, such frameworks include specifications and guidance on such issues as the methods for sediment sampling in the field and for sample analysis in the laboratory, the permitting procedure and so on.

The framework should be sufficiently prescriptive that decision-making is consistent, predictable and transparent, while at the same time incorporating flexibility in decision-making at the site specific level and for the allowance of best professional judgement. The framework should be tiered, such that successive involve increasingly complex assessments and build on data generated in lower tiers. Such a framework provides structure to assessments, and prevents the common shotgun approach to assessments, where all and everything is analysed in the hope that a story will emerge. It also prevents such a cursory assessment that no matter how hard one tries, a confident decision simply cannot be made. The framework should be described in a document in such a manner that it essentially comprises a guidance manual targeted at both dredging proponents and environmental consultants that may be employed by dredging proponents to perform the assessment. The responsibility for the compilation of the dredging assessment framework rests with branch Oceans and Coasts of the Department of Environmental Affairs and Tourism, but will require input and consensus from a range of stakeholders, especially for the technical components of the framework. However, there are frameworks developed by a number of jurisdictions/countries that can be used as a template for the South African dredging assessment framework - the wheel does not need to be reinvented, but rather adapted to local conditions and available expertise.

28. References

- ALLEN HE, FU G and DENG B (1993) Analysis of acid-volatile sulfide (AVS) and simultaneously extracted metals (SEM) for the estimation of potential toxicity in aquatic sediments. *Environmental Toxicology and Chemistry* 12: 1441-1453.
- ALLOUPIS M and ANGELIDIS M (2001) Normalization to lithium for the assessment of metal contamination in coastal sediment cores from the Aegean Sea, Greece. *Marine Environmental Research* 52: 1-12.
- ANKLEY GT, DI TORO DM, HANSEN DJ and BERRY WJ (1996) Technical basis and proposal for deriving sediment quality criteria for metals. *Environmental Toxicology and Chemistry* 15: 2056-2066.
- ANZECC (Australian and New Zealand Environment and Conservation Council) (1994). *Australian Water Quality Guidelines for Fresh and Marine Waters*.
- APITZ SE, AYERS B and KIRTAY VJ (2004) *The Use of Data on Contaminant/Sediment Interactions to Streamline Sediment Assessment and Management*. Final Report, Prepared for Y0817 Navy Pollution Abatement Ashore Technology Demonstration/Validation Program. San Diego (CA): SPAWAR Systems Center. Report nr 1918.
- AVOCET CONSULTING (2003) *Development of freshwater sediment quality values for use in Washington State*. Prepared for Sediment Management Unit, Washington Department of Ecology.
- BABUT M, GARRIC J, CAMUSSO M and DEN BESTEN PJ (2003) Use of sediment quality guidelines in ecological risk assessment of dredged materials: Preliminary reflections. *Aquatic Ecosystem Health and Management* 6: 359-367.
- BAKER B and KRAVITZ M (1992) *Sediment classification method compendium*. United States Environmental Protection Agency. EPA823-R-92-006. Office of Water, Washington, DC.
- BARRICK R, BECKER S, PASTOROK R, BROWN L and BELLER H (1988) *Sediment quality values refinement: 1988 update and evaluation of Puget Sound AET*. Prepared by PTI Environmental Services for Environmental Protection Agency. Bellevue, Washington.
- BATLEY GE, STAHL RG, BABUT MP, BOTT TL, CLARK JR, FIELD LJ, HO KT, MOUNT DR, SWARTZ RC and TESSIER A (2005) Scientific underpinnings of sediment quality guidelines. In: *Use of sediment quality guidelines and related tools for the assessment of contaminated sediments*, Wenning RJ, Batley GE, Ingersoll CG, Moore DW (Eds). SETAC Press Pensacola, FL.
- BIRCH GF and SNOWDON RT (2004) The use of size-normalization techniques in interpretation of soil contaminant distributions. *Water, Air and Soil Pollution*. 157: 1-12.
- BIRCH GF and TAYLOR SE (1999) Sources of heavy metals in sediments of the Port Jackson estuary, Australia. *Science of the Total Environment* 227: 123-138.
- BOLAM SG, REES HL, SOMERFIELD P, SMITH R, CLARKE KR, WARWICK RM, ATKINS M and GARNACHO E (2006) Ecological consequences of dredged material disposal in the marine environment: a holistic assessment of activities around the England and Wales coastline. *Marine Pollution Bulletin* 52: 415-426.
- BOYD SE, LIMPENNY DS, REES HL, COOPER KM and CAMPBELL S (2003) Preliminary observations of the effects of dredging intensity on the re-colonization of dredged sediments off the south-east coast of England (Area 222). *Estuarine, Coastal and Shelf Science* 57: 209-223.
- BRADEN JB, PATUNRU AA, CHATTOPADHYAY S and MAYN N (2004) Contaminant cleanup in the Waukegan harbor area of concern: homeowner attitudes and economic benefits. *Journal of Great Lakes Research* 30: 474-491.
- BRENEMAN D, RICHARDS C and LOZANO S (2000) Environmental influences on benthic community structure in a Great Lakes embayment. *Journal of Great Lakes Research* 26: 287-304.
- BRODERIUS SJ (1991) Modeling the joint toxicity of xenobiotics to aquatic organisms: Basic concepts and approaches. *Aquatic Toxicology* 14: 107-127.
- BURTON GA, Jr (1991) Assessing freshwater sediment toxicity. *Environmental Toxicology and Chemistry* 10: 1585-1627.

- BURTON GA, Jr (2002) Sediment quality criteria in use around the world. *Limnology* 3: 65-75.
- CAETANO M, MADUREIRA M-J and VALE C (2002) Metal remobilisation during resuspension of anoxic contaminated sediment: short-term laboratory study. *Water, Air and Soil Pollution* 143: 23-40.
- CAILLE N, TIFFREAU C, LEYVAL C and MOREL JL (2003) Solubility of metals in an anoxic sediment during prolonged aeration. *Science of the Total Environment* 301: 239-250.
- CALMANO W, HONG J and FÖRSTNER U (1993) Binding and mobilisation of heavy metals in contaminated sediments affected by pH and redox potential. *Water Science and Technology* 28: 223-235.
- CARLSON AR, PHIPPS GL, MATTSOON VR, KOSIAN PAS and COTTER AM (1991) The role of acid volatile sulfide in determining cadmium bioavailability and toxicity in freshwater sediments. *Environmental Toxicology and Chemistry* 10: 1309-1319.
- CASSEE FR, GROTEN JP, VAN BLADEREN PJ and FERON VJ (1998) Toxicological evaluation and risk assessment of chemical mixtures. *Critical Reviews in Toxicology* 28: 73-101.
- CCME (CANADIAN COUNCIL OF MINISTERS OF THE ENVIRONMENT) (2002) Canadian sediment quality guidelines for the protection of aquatic life. Winnipeg, Manitoba.
- CHAPMAN PM (1989) Current approaches to developing sediment quality criteria. *Environmental Toxicology and Chemistry* 8: 589-599.
- CHAPMAN PM (2007) Determining when contamination is pollution — Weight of evidence determinations for sediments and effluents. *Environment International* 33: 492-501.
- CHAPMAN PM and MANN GS (1999) Sediment quality values (SQVs) and ecological risk assessment (ERA). *Marine Pollution Bulletin* 38: 339-344.
- CHAPMAN PM and WANG F (2000) Issues in ecological risk assessment of inorganic metals and metalloids. *Human and Ecological Risk Assessment* 6: 965-988.
- CHAPMAN PM and RIDDLE MJ (2005) Toxic effects of contaminants in polar marine environments. *Environmental Science and Technology* 38: 200-207.
- CHAPMAN PM, ALLARD PJ and VIGERS GA (1999) Development of sediment quality values for Hong Kong special administrative region: a possible model for other jurisdictions. *Marine Pollution Bulletin* 38: 161-169.
- CHAPMAN PM, WANG F, ADAMS WJ and GREEN A (1999) Appropriate applications of sediment quality values for metals and metalloids. *Environmental Science and Technology* 33: 3937-3941.
- CHAPMAN PM, MCDONALD BG and LAWRENCE GS (2002) Weight of evidence frameworks for sediment quality and other assessments. *Human and Ecological Risk Assessment* 8: 1489-1515.
- CHAPMAN PM, WANG F, JANSSEN C, GOULET RR and KAMUNDE CN (2003) Conducting ecological risk assessments of inorganic metals and metalloids - Current status. *Human and Ecological Risk Assessment* 9: 641-697.
- CHEVRIER A and TOPPING PA (1998) *National Guidelines for Monitoring Dredged and Excavated Material at Ocean Disposal Sites*. Environment Canada, Marine Environment Division.
- COBELO-GARCÍA A and PREGO R (2003) Heavy metal sedimentary record in a Galician Ria (NW Spain): background values and recent contamination. *Marine Pollution Bulletin* 46: 1253-1262.
- COLLINS MA (1995) Dredging-induced near-field resuspended sediment concentrations and source strengths. Miscellaneous Paper D-95-2, US Army Engineer Waterways Experiment Station.
- COMMONWEALTH OF AUSTRALIA (2009) National Assessment Guidelines for Dredging, Canberra.
- COOKE TD and DRURY DD (1998) Calabazas Creek Pilot Sediment Sampling Study. *Proceedings of the 1998 NWQMC National Monitoring Conference: Monitoring Critical Foundations to Protect Our Waters*. Reno, Nevada.
- COUCH JA and HARSHBARGER JC (1985) Effects of carcinogenic agents on aquatic animals: An environmental and experimental overview. *Environmental Carcinogenesis Review* 3: 63-105.
- COVELLI S and FONTOLAN G (1997) Application of a normalization procedure in determining regional geochemical baselines. *Environmental Geology* 30: 34-45.
- CRANE JL, MACDONALD DD, INGERSOLL CG, SMORONG DE, LINDSKOOG RA, SEVERN CG, BERGER TA and FIELD LJ (2000) *Development of a framework for evaluating numerical sediment quality targets and sediment contamination in the St. Louis River Area of Concern*. EPA 905-R-00-008. Great Lakes National Program Office. United States Environmental Protection Agency. Chicago, Illinois.
- CROMMENTUIJN T, SIJM D, DE BRUIJN J, VAN LEEUWEN K, VAN DE PLASSCHE E (2000a) Maximum permissible and negligible concentrations for some organic substances and pesticides. *Journal of Environmental Management* 58: 297-312.
- CROMMENTUIJN T, SIJM D, DE BRUIJN J, VAN LEEUWEN K, VAN DE PLASSCHE E (2000b) Maximum permissible and negligible concentrations for metals and metalloids in the Netherlands, taking into account background concentrations. *Journal of Environmental Management* 60: 121-143.
- CRUZ-MOTTA JJ and COLLINS J (2004) Impacts of dredged material disposal on a tropical soft-bottom benthic assemblage. *Marine Pollution Bulletin* 48: 270-280.
- CUMMINGS VJ and THRUSH SF (2004) Behavioural response of juvenile bivalves to terrestrial sediment deposits: implications for post-disturbance recolonization. *Marine Ecology Progress Series* 278: 179-191.
- DASKALAKIS K and O'CONNOR T (1995) Normalization and elemental sediment contamination in the coastal United States. *Environmental Science and Technology* 29: 470-477.
- DE GROOTE J, DUMON G, VANGHELUWE M and JANSEN C (1998) Environmental monitoring of dredging operations in the Belgian nearshore zone. *Terra et Aqua* 70: 21-5.
- DI TORO DM, MAHONY JD, HANSEN DJ, SCOTT KJ, HICKS MB, MAYS SM and Redmond MS (1990) Toxicity of cadmium in sediments: The role of acid-volatile sulfide. *Environmental Toxicology and Chemistry* 9: 1487-1502.
- DI TORO DM, MAHONEY JD, HANSEN DJ, SCOTT KJ, CARLSON AR and ANKLEY GT (1992) Acid volatile sulfide predicts the acute toxicity of cadmium and nickel in sediments. *Environmental Science and Technology* 26: 96-101.
- EC and MENVIQ (Environment Canada and Ministère de l'Environnement du Québec) (1992) *Interim criteria for quality assessment of St. Lawrence River sediment*. ISBN 0-662-19849-2. Environment Canada. Ottawa, Ontario.
- ESSINK K (1999) Ecological effects of dumping of dredged sediments: options for management. *Journal of Coastal Conservation* 5: 69-80.

- ETWB (Environment, Transport and Works Bureau) (2002) *Management of Dredged/Excavated Sediment*. Environment, Transport and Works Bureau, Hong Kong. Technical Circular (Works) No. 34/2002.
- FAIREY R, LONG ER, ROBERTS CA, ANDERSON BS, PHILLIPS BM, HUNT JW, PUCKETT HR and WILSON CJ (2001) An evaluation of methods for calculating mean sediment quality guideline quotients as indicators of contamination and acute toxicity to amphipods by chemical mixtures. *Environmental Toxicology and Chemistry* 20: 2276-2286.
- FAN W, WANG W-X, CHEN J, LI X and YEN Y-F (2002) Cu, Ni and Pb speciation in surface sediments from a contaminated bay of northern China. *Marine Pollution Bulletin* 44: 816-832.
- FAUST M, ALTENBURGER R and GRIMME LH (2000) Predictive assessment of the aquatic toxicity of multiple chemical mixtures. *Journal of Environmental Quality* 29: 1063.
- FERRARO SP, SWARTZ RC, COLE FA and SCHULTS DW (1991) Temporal changes in the benthos along a pollution gradient: Discriminating the effects of natural phenomena from sewage-industrial wastewater effects. *Estuarine and Coastal Shelf Science* 33: 383-407.
- FIELD J, NORTON S, MACDONALD D, SEVERN C and INGERSOLL C (1999) Evaluating sediment chemistry and toxicity data using logistic regression modeling. *Environmental Toxicology and Chemistry* 18: 1311-1322.
- FIELD LJ, MACDONALD DD, NORTON SB and INGERSOLL CG (2002) Predicting amphipod toxicity from sediment chemistry using logistic regression models. *Environmental Toxicology and Chemistry* 21: 1993-2005.
- FLETCHER R, WELSH P and FLETCHER T (2008) *Guidelines for Identifying, Assessing, and Managing Contaminated Sediments in Ontario*. Ontario Ministry of the Environment.
- FINNEY B and HUH CA (1989) High resolution sedimentary records of heavy-metals from the Santa Monica and San Pedro Basins, California. *Marine Pollution Bulletin* 20: 181-187.
- FLEEGER JW, GUST KA, MARLBOROUGH SJ and TITA G (2007) Mixtures of metals and polynuclear aromatic hydrocarbons elicit complex, nonadditive toxicological interactions in meiobenthic copepods. *Environmental Toxicology and Chemistry* 26: 1677-1685.
- FÖRSTNER U (1989) Contaminated sediments. In: Bhattacharji S, Friedman GM, Neugebauer HJ and Seilacher A (Eds), *Lecture notes in earth sciences*. Springer Verlag, Berlin.
- FÖRSTNER U and WITTMANN GTW (1979) *Metal Pollution in the Aquatic Environment*. Springer-Verlag, New York.
- FÖRSTNER U, CALMANO W and SCHOER J (1982) Heavy metals in bottom sediments and suspended material from the Elbe, Weser and Ems estuaries and from the German Bight (southeastern North Sea). *Thalassia Jugoslavica* 18: 97-122.
- FRANCINGUES NR, PALERMO MR, LEE CR and PEDDICORD RK (1985) *Management strategy for disposal of dredged material: contaminant testing and controls*. US Army Corps of Engineers. Miscellaneous Paper D-85-1.
- FRANKLIN NM, STAUBER JL, LIM RP and PETOCZ P (2002) Toxicity of metal mixtures to a tropical freshwater alga (*Chlorella* sp.): the effect of interactions between copper, cadmium, and zinc on metal cell binding and uptake. *Environmental Toxicology and Chemistry* 21: 2412-2422.
- FREDETTE TJ and FRENCH GT (2004) Understanding the physical and environmental consequences of dredged material disposal: history in New England and current perspectives. *Marine Pollution Bulletin* 49: 93-102.
- GOBEIL C, MACDONALD RW and SUNDBY B (1997) Diagenetic separation of cadmium and manganese in suboxic continental margin sediments. *Geochimica et Cosmochimica Acta* 61: 4647-4654.
- GOOSSENS H and ZWOLSMAN JGG (1996) An evaluation of the behaviour of pollutants during dredging activities. *Terra et Aqua* 62: 20-27.
- GOYETTE D, BRAND D and THOMAS M (1988) *Prevalence of idiopathic liver lesions in English sole and epidermal abnormalities in flatfish from Vancouver Harbour, British Columbia, 1986*. Regional Program Report 87-09. Environment Canada, Vancouver.
- GRANT A and MIDDLETON R (1990) An assessment of metal contamination of sediments in the Humber Estuary, U.K. *Estuarine Coastal and Shelf Science* 31: 71-85.
- GUST KA and FLEEGER JW (2005) Exposure-related effects on Cd bioaccumulation explains toxicity of Cd-phenanthrene mixtures in *Hyaella azteca*. *Environmental Toxicology and Chemistry* 24: 2918-2926.
- HALSTEAD MJR, CUNNINGHAME RG and HUNTER KA (2000) Wet deposition of trace metals to a remote site in Fjordland, New Zealand. *Atmospheric Environment* 34: 665-76.
- HANSON P, EVANS D, COLBY D and ZDANOWICS V (1993) Assessment of elemental contamination in estuarine and coastal environments based on geochemical and statistical modeling of sediments. *Marine Environmental Research* 36: 237-266.
- HATJE V, PAYNE TE, HILL DM, MCORIST G, BIRCH GF and SZYMCAK R (2003) Kinetics of trace element uptake and release by particles in estuarine waters: effects of pH, salinity, and particle loading. *Environmental International* 29: 619-629.
- HAYES D and WU P-Y (2001) Simple approach to TSS source strength estimates. Western Dredging Association Proceedings, WEDA XXI, Houston, TX, June 25-27, 2001.
- HERBICH JB and BRAHME SB (1991) Literature review and technical evaluation of sediment resuspension during dredging. Contract Report HL-91-1, prepared for the Department of the Army, U.S. Army Corps of Engineers, Washington, D.C.
- HOLMSTRUP M, BINDESBØL A-M, OOSTINGH GJ, DUSCHL A, SCHEIL V, KÖHLER H-R, LOUREIRO S, SOARES AMVM, FERREIRA ALG, KLENLE C, GERHARDT A, LASKOWSKI R, KRAMARZ PE, BAYLEY M, SVENDSEN C and SPURGEON DJ (2010) Interactions between effects of environmental chemicals and natural stressors: A review. *Science of the Total Environment* 408: 3746-3762.
- HONEYMAN BD and SANTOSCHI PH (1988) Metals in aquatic systems. *Environmental Science and Technology* 22: 862-871.
- HOROWITZ AJ (1991) *A Primer on Sediment-Trace Element Chemistry*. Lewis Publishers Inc, Michigan.
- HUH CA, FINNEY BP and STULL JK (1992) Anthropogenic inputs of several heavy metals to nearshore basins off Los Angeles. *Progress in Oceanography* 30: 335-51.
- INGERSOLL CG, HAVERLAND PS, BRUNSON EL, CANFIELD TJ, DWYER FJ, HENKE CE, KEMBLE NE, MOUNT DR and FOX RG (1996) Calculation and evaluation of sediment effect concentrations for the amphipod *Hyaella azteca* and the midge *Chironomus riparius*. *Journal of Great Lakes Research* 22: 602-623.
- INGERSOLL CG, DILLON T and BIDDINGER RG (1997) Methodological uncertainty in sediment ecological risk assessment. In: *Ecological risk assessments of contaminated sediment*. SETAC Press, Pensacola, Florida.

- INGERSOLL CG, BAY SM, CRANE JL, FIELD LJ, GRIES TH, HYLAND JL, LONG ER, MACDONALD DD and O'CONNOR TP (2005) Ability of SQGs to estimate effects of sediment-associated contaminants in laboratory toxicity tests or in benthic community assessments. In: *Use of sediment quality guidelines and related tools for the assessment of contaminated sediments*, Wenning RJ, Batley GE, Ingersoll CG, Moore DW (Eds). SETAC Press Pensacola, FL.
- JACOB J (2008) The significance of polycyclic aromatic hydrocarbons as environmental carcinogens. 35 years research on PAH-a retrospective. *Polycyclic Aromatic Compounds* 28: 242-272.
- JOHNSON BH and PARCHURE TM (2000) Estimating dredging sediment resuspension sources. DOER Technical Notes Collection, TN DOER-E6, U.S. Army Engineer Research and Development Center, Vicksburg, MS.
- JONES DS, SUTER GW and HULL RN (1997) Toxicological benchmarks for screening contaminants of potential concern for effects on sediment-associated organisms: 1997 revision. US Department of Energy, Office of Environmental Management. ES/ER/TM-95/R4.
- JONES-LEE A and LEE GF (2005) Unreliability of co-occurrence-based sediment quality guidelines for contaminated sediment evaluations at Superfund/hazardous chemical sites. *Remediation* (Spring 2005): 19-33. Published online in Wiley Interscience (www.interscience.wiley.com); DOI: 10.1002/rem.20040.
- KERSTEN M and SMEDES F (2002) Normalization procedures for sediment contaminants in spatial and temporal trend monitoring. *Journal of Environmental Monitoring* 4: 109-115.
- KWOK KWH, LEUNG KMY, LUI GSG, CHU VKH, LAM PKS, MORRITT D, MALTBY L, BROCK TCM, VAN DEN BRINK PJ, WARNE MSTJ and CRANE M (2007) Comparison of tropical and temperate freshwater animal species' acute sensitivities to chemicals: Implications for deriving safe extrapolation factors. *Integrated Environmental Assessment and Management* 3: 49-67.
- USACE (US Army Corps of Engineers) (1998) *Use of Sediment Quality Guidelines (SQGs) in Dredged Material Management*. Technical Note EEDP-04-29, Vicksburg, MS.
- LEE GF and JONES-LEE A (1993) *Sediment Quality Criteria: Numeric Chemical vs. Biological Effects-Based Approaches*. Proceedings of Water Environment Federation National Conference, Anaheim, California.
- LEE GF, BANDYOPADHYAY P, BUTLER J, HOMER D, JONES R, LOPEZ J, MARIANI G, MCDONALD C, NICAR M, PIWONI M and SALEH F (1977) Investigation of water quality parameters at the offshore disposal site, Galveston, Texas. Technical Report D-77-20, US Army Corps of Engineers Dredged Material Research Program, Vicksburg, MS.
- LEE GF and JONES-LEE A (2004). Appropriate incorporation of chemical information in a best professional judgment "triad" weight of evidence evaluation of sediment quality. *Aquatic Ecosystem Health and Management* 7: 351-356.
- LEE J and MOREL FM (1995) Replacement of zinc by cadmium in marine phytoplankton. *Marine Ecology Progress Series* 127: 305-309.
- LEE J, ROBERTS S and MOREL FM (1995) Cadmium: a nutrient for the marine diatom *Thalassiosira weissflogii*. *Limnology and Oceanography* 40: 1056-1063.
- LI X, SHEN Z, WAI OWH and LI Y-S (2000) Chemical partitioning of heavy metal contaminants in sediments of the Pearl River Estuary. *Chemical Speciation and Bioavailability* 12: 17-25.
- LOHRER AM, THRUSH SF, HEWITT JE, BERKENBUSCH K, AHRENS M and CUMMINGS VJ (2004) Terrestrially derived sediment: response of marine macrobenthic communities to thin terrigenous deposits. *Marine Ecology Progress Series* 273: 121-138.
- LONG ER and MORGAN LG (1991) The potential for biological effects of sediment-sorbed contaminants tested in the National Status and Trends Program. NOAA Technical Memorandum NOS OMA 52. National Oceanic and Atmospheric Administration. Seattle, Washington.
- LONG ER and MACDONALD DD (1998) Recommended uses of empirically derived, sediment quality guidelines for marine and estuarine ecosystems. *Human and Ecological Risk Assessment* 4: 1019-1039.
- LONG ER, FIELD JE and MACDONALD DD (1998) Predicting toxicity in marine sediments with numerical sediment quality guidelines. *Environmental Toxicology and Chemistry* 17: 714-727.
- LONG ER, MACDONALD DD, SEVERN CG and HONG CB (2000) Classifying the probabilities of acute toxicity in marine sediments with empirically-derived sediment quality guidelines. *Environmental Toxicology and Chemistry* 19: 2598-2601.
- LONG ER, INGERSOLL CG and MACDONALD DD (2006) Calculation and uses of mean sediment quality guideline quotients: A critical review. *Environmental Science and Technology* 40: 1726-1736.
- LORING DH (1990) Lithium - a new approach for the granulometric normalization of trace metal data. *Marine Chemistry* 26: 155-168.
- LORING DH (1991) Normalization of heavy-metal data from estuarine and coastal sediments. *ICES Journal of Marine Science* 48: 101-115.
- LORING DH and RANTALA RTT (1992) Manual for the geochemical analyses of marine sediments and suspended particulate matter. *Earth-Science Review* 32: 235-283.
- LUOMA SN (1983) Bioavailability of trace metals to aquatic organisms - a review. *Science of the Total Environment* 28: 1-22.
- LUOMA SN, VAN GEEN A, LEE B-G and CLOERN JE (1998) Metal uptake by phytoplankton during a bloom in south San Francisco Bay. *Limnology and Oceanography* 43: 1007-1016.
- LUOMA SN (1990) Processes affecting metal concentrations in estuarine and coastal marine sediments. In: Furness RW and Rainbow PS (Eds), *Heavy Metals in the Marine Environment*. CRC Press, Boca Raton.
- MACDONALD DD, SMITH SL, WONG MP and MURDOCH P (1992) *The development of Canadian marine environmental quality guidelines*. Ecosystem Sciences and Evaluation Directorate, Environment Canada, Ottawa.
- MACDONALD D (1994) *Approach to the Assessment of Sediment Quality in Florida Coastal Waters*. Florida Department of Environmental Protection Office of Water Policy. Tallahassee, Florida.
- MACDONALD DD (1997) *Sediment injury in the Southern California Bight: Review of the toxic effects of DDTs and PCBs in sediments*. Prepared for National Oceanic and Atmospheric Administration. United States Department of Commerce. Long Beach, California.
- MACDONALD DD, CARR RS, CALDER FD, LONG ER and INGERSOLL CG (1996) Development and evaluation of sediment quality guidelines for Florida coastal waters. *Ecotoxicology* 5:253-278.

- MACDONALD DD, INGERSOLL CG and BERGER TA (2000) Development and evaluation of consensus-based sediment quality guidelines for fresh water ecosystems. *Archives Environmental Toxicology and Chemistry* 39: 20-31.
- MACDONALD DD, INGERSOLL CG, SMORONG DE and LINDSKOOG RA (2003) *Development and applications of sediment quality criteria for managing contaminated sediment in British Columbia*. Prepared for British Columbia Ministry of Water, Land and Air Protection, Environmental Management Branch, Victoria, British Columbia.
- MALINS DC, MCCAIN BB, BROWN DW, CHAN SL, MYERS MS, LANDAHL JT, PROHASKA PG, FRIEDMAN AJ, RHODES LD, BURROWS DG, GRONLUND WD and HODGKINS HO (1984) Chemical pollutants in sediments and diseases of bottom-dwelling fish in Puget Sound, Washington. *Environmental Science and Technology* 18: 705-713.
- MARITIME SAFETY AUTHORITY OF NEW ZEALAND (1999) *New Zealand Guidelines for Sea Disposal of Waste*. Advisory Circular Part 180: Dumping of Waste or Other Matter, Issue No 180-1.
- MARTIN JM and WHITFIELD M (1983) The significance of the river inputs to the ocean. In: Wong CS, Boyle E, Bruland EW, Burton JD and Goldberg ED (Eds), *Trace metals in seawater*. Plenum Press, New York.
- MAURER D, KECK R, TINSMAN J, LEATHEM W, WETHE C, HUTZINGER M, LORD C and CHURCH T (1978) Vertical migration of benthos in simulated dredged material overburdens; Volume I: Marine benthos. Technical Report No. D-78-35. US Army Engineer Waterways Experiment Station, Vicksburg, MS.
- MAURER D, KECK R, TINSMAN J, LEATHEM W, WETHE C, HUTZINGER M, LORD C and CHURCH T (1986) Vertical migration and mortality of marine benthos in dredged material: A synthesis. *Internationale Revue gesamten Hydrobiologie* 71: 49-63.
- MATTHAI C and BIRCH GF (2001) Detection of anthropogenic Cu, Pb and Zn in continental shelf sediments off Sydney - a new approach using normalization with cobalt. *Marine Pollution Bulletin* 42: 1055-1063.
- MATTHAI C, BIRCH GF and BICKFORD GP (2002) Anthropogenic trace metals in sediment and settling particulate matter on a high-energy continental shelf (Sydney, Australia). *Marine Environmental Research* 54: 99-127.
- MCCAULEY DJ, DEGRAEVE GM and LINTON TK (2000) Sediment quality guidelines and assessment: overview and research needs. *Environmental Science Policy* 3: 133-144.
- MCCREADY S, BIRCH GF, LONG ER, SPYRAKIS G and GREELY CR (2006) An evaluation of Australian sediment quality guidelines. *Archives of Environmental Contamination and Toxicology* 50: 306-315.
- MCGEE BL, SCHLEKAT CE, BOWARD DM and WADE TL (1995) Sediment contamination and biological effects in a Chesapeake Bay marina. *Ecotoxicology* 4: 39-59.
- MCPHERSON C, CHAPMAN PM, DE BRUYN AMH and COOPER L (2008) The importance of benthos in weight of evidence sediment assessments - A case study. *Science of the Total Environment* 394: 252-264.
- MWANUZI F and DE SMEDT F (1999) Heavy metal distribution model under estuarine mixing. *Hydrological Processes* 13: 789-804.
- NEFF JM, BEAN DJ, CORNABY BW, VAGA RM, GULBRANSEN TC and SCANLON JA (1986) *Sediment quality criteria methodology validation: Calculation of screening level concentrations from field data*. Prepared for Environmental Protection Agency Region V. Washington, District of Columbia.
- NEWELL RC, SEIDERER LJ and HITCHCOCK DR (1998) The impact of dredging works in coastal waters: a review of the sensitivity to disturbance and subsequent recovery of biological resources on the seabed. *Oceanography and Marine Biology Annual Review* 36: 127-178.
- NEWMAN MC and UNGER MA (2003) *Fundamentals of ecotoxicology*. Lewis Publishers. Boca Raton.
- NEWMAN BK and WATLING RJ (2007) Definition of baseline metal concentrations for assessing metal enrichment of sediment from the south-eastern Cape coastline of South Africa. *Water SA* 33: 677-694.
- NORWOOD WP, BORGMANN U; DIXON DG and WALLACE A (2003) Effects of metal mixtures on aquatic biota: A review of observations and methods. *Human and Ecological Risk Assessment* 9: 795-811.
- NYSDEC (New York State Department of Environmental Conservation) (1999) *Technical guidance for screening contaminated sediments*. Division of Fish and Wildlife, Division of Marine Resources, Albany (NY), USA.
- O'CONNOR TP (2004) The sediment quality guideline, ERL, is not a chemical concentration at the threshold of sediment toxicity. *Marine Pollution Bulletin* 49: 383-385.
- O'CONNOR TP, DASKALAKIS KD, HYLAND JL, PAUL JF and SUMMERS JK (1998) Comparisons of sediment toxicity with predictions based on chemical guidelines. *Environmental Toxicology and Chemistry* 17: 468-471.
- O'CONNOR TP and PAUL JF (2000) Misfit between sediment toxicity and chemistry. *Marine Pollution Bulletin* 40: 59-64.
- OLSEN CR, CUTSHALL NH and LARSEN IL (1982) Pollutant-particle associations and dynamics in coastal marine ecosystems: a review. *Marine Chemistry* 11: 501-533.
- PAINE MD, CHAPMAN PM, ALLARD PJ, MURDOCH MH and MINIFIE D (1996) Limited bioavailability of sediment PAH near an aluminum smelter: contamination does not equal effects. *Environmental Toxicology and Chemistry* 15: 2003-2018.
- PALERMO MR, SCHROEDER PR, ESTES TJ and FRANCINGUES NR (2008) Technical guidelines for environmental dredging of contaminated sediments. United States Army Corps of Engineers. Engineer Research and Development Center. ERDC/EL TR-08-29.
- PAPE-LINDSTROM PA and LYDY MJ (1997) Synergistic toxicity of atrazine and organophosphate insecticides contravenes the response addition mixture model. *Environmental Toxicology and Chemistry* 11: 2415-2420.
- PEDDICORD RK (1987) Overview of the influence of dredging and dredged material disposal on the fate and effects of sediment-associated chemicals. In: *Fate and effects of sediment-bound chemicals in aquatic systems*. Dickson KL (ed), New York: Pergamon.
- PENNEKAMP JGS, ESKAMP RJC, ROSENBRAND WF, MULLIE A, WESSEL GL, ARTS T and DECIBEL IK (1996) Turbidity caused by dredging; viewed in perspective. *Terra et Aqua* 64: 10-17.
- PERSAUD D, JAAGUMAGI R and HAYTON A (1993) *Guidelines for the protection and management of aquatic sediment quality in Ontario*. Water Resources Branch. Ontario Ministry of the Environment. Toronto, Ontario.
- PETERS C and GANDRASS J (2001) Bioassays as a tool for the assessment of the quality of dredged material. In: *Dredged material in the port of Rotterdam—interface between Rhine catchment area and North Sea*, Gandrass J and

- Salomons W (eds). Project report, GKSS Research Centre, Geesthacht, Germany.
- PIANC (2005) *Biological assessment guidance for dredged material*. International Navigation Association, Brussels, Belgium.
- PIETERS A, VAN PARYS M, DUMON G and SPEELERS L (2002) Chemical monitoring of maintenance dredging operations at Zeebrugge. *Terra et Aqua* 86.
- RAE JE and ALLEN JRL (1993) The significance of organic matter degradation in the interpretation of historical pollution trends in depth profiles of estuarine sediment. *Estuaries* 16: 678-682.
- REIBLE DD, FLEEGER JW, PARDUE J and TOMSON M (2002) Contaminant release during removal and resuspension. <http://www.hsrrc.org/hsrrc/html/ssw/ssw-contaminant.html>.
- REIMANN C and DE CARITAT P (2005) Distinguishing between natural and anthropogenic sources for elements in the environment: regional geochemical surveys versus enrichment factors. *Science of the Total Environment* 337: 91-107.
- ROACH AC (2005) Assessment of metals in sediments from Lake Macquarie, New South Wales, Australia, using normalisation models and sediment quality guidelines. *Marine Environmental Research* 59: 453-472.
- ROBERTS RD and FORREST BM (1999) Minimal impact from long-term dredge spoil disposal at a dispersive site in Tasman Bay, New Zealand. *New Zealand Journal of Marine and Freshwater Research* 33: 623-633.
- ROUSSIEZ V, LUDWIG W, PROBST J-L and MONACO A (2005) Background levels of heavy metals in surficial sediments of the Gulf of Lions (NW Mediterranean): An approach based on 133Cs normalization and lead isotope measurements. *Environmental Pollution* 138: 167-177.
- RUBIO B, NOMBELA MA and VILAS F (2000) Geochemistry of major and trace elements in sediments of the Ria de Vigo (NW Spain): an assessment of metal pollution. *Marine Pollution Bulletin* 40: 968-980.
- RULE JH (1986) Assessment of trace element geochemistry of Hampton Roads Harbor and Lower Chesapeake Bay area sediments. *Environmental Geology and Water Science* 8: 209-219.
- SAULNIER I and MUCCI A (2000) Trace metal remobilization following the resuspension of estuarine sediments: Saguenay Fjord, Canada. *Applied Geochemistry* 15: 191-210.
- SCHIFF KC and WEISBERG SW (1999) Iron as a reference element for determining trace metal enrichment in Southern California coastal shelf sediments. *Marine Environmental Research* 48: 161-176.
- SCHROPP S and WINDOM H (1988) A guide to the interpretation of metal concentrations in estuarine sediments. Florida Department of Environmental Protection. Tallahassee, Florida.
- SCHROPP S, LEWIS FG, WINDOM HL, RYAN JD, CALDER FD and BURNEY LC (1990) Interpretation of metal concentrations in estuarine sediments of Florida using aluminum as a reference element. *Estuaries* 13: 227-235.
- SHERIDAN P (2004) Recovery of floral and faunal communities after placement of dredged material on seagrasses in Laguna Madre, Texas. *Estuarine, Coastal and Shelf Science* 59: 441-458.
- SIMPSON SL, APTE SC and BATELY GE (1989) Effect of short-term resuspension events on trace metal speciation in polluted anoxic sediments. *Environmental Science and Technology* 23: 620-625.
- SMITH SDA and RULE MJ (2001) The effects of dredge-spoil dumping on a shallow water soft-sediment community in the Solitary Island Marine Park, NSW, Australia. *Marine Pollution Bulletin* 42: 1040-1048.
- SMITH SL, MACDONALD DD, KEENLEYSIDE KA, INGERSOLL CG and FIELD LJ (1996) A preliminary evaluation of sediment quality assessment values for freshwater ecosystems. *Journal of Great Lakes Research* 22: 624-638.
- SORENSEN MT, CONDER JM, FUCHSMAN PC, MARTELLO LB and WENNING RJ (2007) Using a sediment quality triad approach to evaluate benthic toxicity in the Lower Hackensack River, New Jersey. *Archives of Environmental Contamination and Toxicology* 53: 36-49.
- STRONKHORST J and VAN HATTUM B (2003) Contaminants of concern in Dutch marine harbor sediments. *Archives of Environmental Contamination and Toxicology* 45: 306-316.
- SUH J-Y and BIRCH GF (2005) Use of grain-size and elemental normalization in the interpretation of trace metal concentrations in soils of the reclaimed area adjoining Port Jackson, Sydney, Australia. *Water, Air, and Soil Pollution* 160: 357-371.
- SUMMERS JK, WADE TL, ENGLE VD and MALEB ZA (1996) Normalization of metal concentrations in estuarine sediments from the Gulf of Mexico. *Estuaries* 19: 581-594.
- SUTHERLAND RA (2000) Bed sediment-associated trace metals in an urban stream, Oahu, Hawaii. *Environmental Geology* 39: 611-627.
- SWARTZ RC, DITSWORTH GR, SCHULTS DW and LAMBERSON JO (1985) Sediment toxicity to a marine infaunal amphipod: Cadmium and its interaction with sewage sludge. *Marine Environmental Research* 18: 133-153.
- SWARTZ RC, COLE FA, SCHULTS DW and DEBEN WA (1986) *Ecological changes in the Southern California Bight near a large sewage outfall: benthic conditions in 1980 and 1983*. United States Environmental Protection Agency. Newport, Oregon.
- SWARTZ RC, SCHULTS DW, LAMBERSON JO, OZRETICH RJ and STULL JK (1991) Vertical profiles of toxicity, organic carbon, and chemical contaminants in sediment cores from the Palos Verdes shelf and Santa Monica Bay, California. *Marine and Environmental Research* 31: 215-225.
- SWARTZ RC, SCHULTS DW, OZRETICH RJ, LAMBERSON JO, COLE FA, DE WITT TH, REDMOND MS and FERRARO SP (1995) PAH: A model to predict the toxicity of polynuclear aromatic hydrocarbon mixtures in field-collected sediments. *Environmental Toxicology and Chemistry* 14: 1977-1987.
- SWARTZ RC (1999) Consensus sediment quality guidelines for polycyclic aromatic hydrocarbon mixtures. *Environmental Toxicology and Chemistry* 18: 780-787.
- TANNER PA, LEONG LS and PAN SP (2000) Contamination of heavy metals in marine sediment cores from Victoria Harbour, Hong Kong. *Marine Pollution Bulletin* 40: 769-779.
- TAYLOR SR and MCLENNAN SM (1981) The composition and evolution of the continental crust: rare earth element evidence from sedimentary rocks. *Philosophical Transactions of the Royal Society, London* 301: 381-399.
- THOMAS CA and BENDELL-YOUNG LI (1999) The significance of diagenesis versus riverine input in contributions to the sediment geochemical matrix of iron and manganese in an intertidal region. *Estuarine Coastal and Shelf Science* 48: 635-647.

- THOMPSON B and LOWE S (2004) Assessment of macrobenthos response to sediment contamination in the San Francisco Estuary, California, USA. *Environmental Toxicology and Chemistry* 23: 2178-2187.
- TORRES RJ, ABESSA DMS, SANTOS FC, MARANHÃO LA, DAVANSO MB, DO NASCIMENTO MRL and MOZETO AA (2009) Effects of dredging operations on sediment quality: contaminant mobilization in dredged sediments from the Port of Santos, SP, Brazil. *Journal of Soils and Sediments* 9: 420-432.
- TUREKIAN KK and WEDEPOHL KH (1961) Distribution of the elements in some major units of the earth's crust. *Geological Society of America Bulletin* 72: 175-192.
- USACE (United States Army Corps of Engineers) (1998) *Use of Sediment Quality Guidelines (SQGs) in Dredged Material Management*. Dredging Research Technical Note EEDP-04-29.
- USACE (United States Army Corps of Engineers) (2006) *Sediment evaluation framework for the Pacific Northwest*.
- USACE (United States Army Corps of Engineers) (2008) *Dredged material management program*. Prepared by Dredged Material Management Office, US Army Corps of Engineers, Seattle District.
- USEPA (1989) *Evaluation of the Apparent Effects Threshold (AET) Approach for Assessing Sediment Quality*. Report of the Sediment Criteria Subcommittee, Science Advisory Board. SAB-EETFC-89-027.
- USEPA (United States Environmental Protection Agency) (1992) *Marine benthic community structure assessment. Sediment classification methods compendium*. EPA 813/R-92/006. Washington, District of Columbia.
- USEPA (United States Environmental Protection Agency) (1997) *The incidence and severity of sediment contamination in surface waters of the United States. Volume 1: National sediment quality survey*. EPA 823-R-97-006, Office of Science and Technology, Washington, DC
- USEPA (United States Environmental Protection Agency) (1998) *EPA's Contaminated Sediment Management Strategy*. EPA-823-R-98-001.
- USEPA (2003) Procedures for the derivation of equilibrium partitioning sediment benchmarks (ESBs) for the protection of benthic organisms: PAH mixtures. U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC. EPA/600/R-02/013.
- USEPA (2005) Procedures for the derivation of equilibrium partitioning sediment benchmarks (ESBs) for the protection of benthic organisms: Metal mixtures (cadmium, copper, lead, nickel, silver and zinc). U.S. EPA, Office of Research and Development, Washington, DC. EPA-600-R-02-011.
- VAN DEN BERG GA, MEIJERS GGA, VAN DER HEIJDT LM and ZWOLSMAN JGG (2001) Dredging-related mobilization of trace metals: a case study in the Netherlands. *Water Research* 35: 1979-86.
- VEINOTT G, PERRON-CASHMAN S and ANDERSON MR (2001) Baseline metal concentrations in coastal Labrador sediments. *Marine Pollution Bulletin* 42: 187-192.
- VETHAAK AD and WESTER PW (1996) Diseases of flounder *Platichthys flesus* in Dutch coastal and estuarine waters with particular reference to environmental stress factors. II. *Liver histopathology*. *Diseases of Aquatic Organisms* 26: 99-116.
- VIDAL DE and BAY SM (2005) Comparative sediment quality guidelines performance for predicting sediment toxicity in southern California, USA. *Environmental Toxicology and Chemistry* 24: 3173-3182.
- VIVAN JM, DI DOMENICO D and DE ALMEIDA TCM (2009) Effects of dredged material disposal on benthic macrofauna near Itajaí Harbour (Santa Catarina, South Brazil). *Ecological Engineering* 35: 1435-1443.
- WARREN LA, TESSIER A and HARE L (1998) Modelling cadmium accumulation by benthic invertebrates in situ: The relative contributions of sediment and overlying water reservoirs to organism cadmium concentrations. *Limnology and Oceanography* 43: 1442-1454.
- WEDEPOHL KH (1995) The composition of the continental crust. *Geochimica et Cosmochimica Acta* 59: 1217-1232.
- WEISBERG SB, WILSON HT, HEIMBUCH DG, WINDOM HL and SUMMERS JK (2000) Comparison of sediment metal:aluminum relationships between the eastern and Gulf Coasts of the United States. *Environmental Monitoring and Assessment* 61: 373-385.
- WENNING RJ and INGERSOLL CG (2002) *Use of sediment quality guidelines and related tools for the assessment of contaminated sediments*. Executive Summary of a SETAC Pellston Workshop. 2002, 17 August-22 August; Fairmont (MT), USA. SETAC, Pensacola (FL), USA.
- WENNING RJ, BATLEY GE, INGERSOLL CG, MOORE DW (2005) *Use of sediment quality guidelines and related tools for the assessment of contaminated sediments*. Pensacola, FL, USA: SETAC Press.
- WETHERINGTON JD, STANLEY BH, ADAMS KO and, SCHWER RF (2005) An evaluation of logistic regression models for predicting amphipod toxicity from sediment chemistry. *Environmental Toxicology and Chemistry* 24: 2691-2700.
- WHOMERSLEY P, WARE S, REES HL, MASON CE, BOLAM T, HUXHAM M and BATES HD (2008) Biological indicators of disturbance at a dredged material disposal site in Liverpool Bay, UK: an assessment using time-series data. *ICES Journal of Marine Science* 65.
- WINDOM H, SCHROPP S, CALDER F, RYAN J, SMITH R, BURNEY L, LEWIS F and RAWLINSON C (1989) Natural trace metal concentrations in estuarine and coastal marine sediments of the south-eastern United States. *Environmental Science and Technology* 23: 314-320.
- WORD JQ, ALBRECHT BB, ANGHERRA ML, BAUDO R, BAY MS, DI TORO DM, HYLAND JL, INGERSOLL CG, LANDRUM PF, LONG ER, MEADOR JP, MOORE DW, O'CONNOR TP and SHINE JP (2002) Predictive ability of sediment quality guidelines. In: *Use of sediment quality guidelines and related tools for the assessment of contaminated sediments*, Wenning RJ, Batley GE, Ingersoll CG, Moore DW (Eds). SETAC Press Pensacola, FL.
- WSEDEC (Washington State Department of Ecology) (1995) *Sediment Management Standards*. Chapter 173-204 WAC.
- ZHUANG Y, ALLEN HE and FU G (1994) Effect of aeration of sediment on cadmium binding. *Environmental Toxicology and Chemistry* 13: 717-24.
- ZOUMIS T, SCHMIDT A, GRIGOROVA L and CALMANO W (2001) Contaminants in sediments: remobilisation and demobilisation. *Science of the Total Environment* 266: 195-202.

29. Appendices

Appendix 1

CHAPTER 8

MARINE AND COASTAL POLLUTION CONTROL

Dumping permits

- 71.** (1) A person who wishes to dump at sea any waste or other material must—
- (a) apply in writing to the Minister in the form stipulated by the Minister for a dumping permit that authorises the waste or other material to be loaded aboard a vessel, aircraft, platform or other structure and to be dumped at sea; and
 - (b) pay the prescribed fee.
- (2) When deciding an application for a dumping permit contemplated in subsection (1), the Minister must have regard to—
- (a) the Waste Assessment Guidelines set out in Schedule 2;
 - (b) any coastal management programme applicable in the area;
 - (c) the likely environmental impact of the proposed activity;
 - (d) national legislation dealing with waste;
 - (e) the interests of the whole community;
 - (f) transboundary impacts and international obligations and standards; and
 - (g) any other factors that may be prescribed.
- (3) The Minister may not grant a dumping permit that authorises the dumping of any waste or other material, other than—
- (a) dredged material;
 - (b) sewage sludge;
 - (c) fish waste, or material resulting from industrial fish processing operations;
 - (d) vessels and platforms or other man-made structures at sea;
 - (e) inert, inorganic geological material;
 - (f) organic material of natural origin; or
 - (g) bulky items primarily comprising iron, steel, concrete and similarly nonharmful materials for which the concern is physical impact, and limited to those circumstances where such wastes are generated at locations, such as small islands with isolated communities, having no practicable access to disposal options other than dumping at sea.
- (4) The Minister may not issue a dumping permit if—
- (a) the waste or other material proposed for dumping contains—
 - (i) levels of radioactivity greater than as defined by the International Atomic Energy Agency and adopted by the contracting parties to the Protocol to the Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter adopted on 7 November 1996; or
 - (ii) material which is capable of creating floating debris or otherwise contributing to the pollution of the marine environment and which could be removed from the material proposed for dumping;
 - (b) dumping the waste or other material in question—
 - (i) is likely to cause irreversible or long-lasting adverse effects that cannot satisfactorily be mitigated;
 - (ii) would cause a serious obstacle to fishing or navigation;
 - (iii) would prejudice the achievement of any coastal management objective contained in a coastal management programme;
 - (iv) would be contrary to the obligations of the Republic under international law; or
 - (v) would be contrary to the interests of the whole community.
- (5) A dumping permit must be issued for a specified period of not more than two years but may be renewed once for a period of not more than two years.

Emergency dumping at sea

- 72.** (1) The Minister may in relation to any application for a dumping permit referred to in section 71 dispense with any prescribed procedure, including any consultation and public participation processes, if—
- (a) the dumping at sea of a quantity of any particular waste or other material is necessary to avert an emergency that poses an unacceptable risk to the environment or to human health or safety; and
 - (b) there is no other feasible solution.
- (2) Before issuing a permit in the circumstances contemplated in subsection (1), the Minister must consult with—
- (a) any foreign state that is likely to be affected by the proposed dumping at sea; and
 - (b) the International Maritime Organisation.
- (3) The Minister must—
- (a) as far as reasonably possible in the circumstances, follow any recommendations received from the International Maritime Organisation when imposing permit conditions regarding the procedures to be followed in conducting the loading or dumping at sea of the relevant quantity of waste or other material; and
 - (b) inform the International Maritime Organisation of any action taken under this section within a reasonable period.

National action list

- 73.** (1) The Minister must progressively and subject to available resources, develop a national action list to provide a mechanism for screening waste and other material on the basis of their potential effect on human health and the marine environment.
- (2) The national action list must—
- (a) be developed in accordance with the Waste Assessment Guidelines set out in Schedule 2; and
 - (b) contain the prescribed information.

Appendix 2

SCHEDULE 2

(Section 71)

GUIDELINES FOR THE ASSESSMENT OF WASTES OR OTHER MATERIAL THAT MAY BE CONSIDERED FOR DUMPING AT SEA ("the Waste Assessment Guidelines")

GENERAL

1. This Schedule sets out guidelines for reducing the necessity for dumping at sea in accordance with Schedule II to the Protocol to the Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matters adopted on 7 November 1996.

WASTE PREVENTION AUDIT

2. The initial stages in assessing alternatives to dumping at sea should, as appropriate, include an evaluation of—
 - (a) the types, amounts and relative hazard of wastes generated;
 - (b) details of the production process and the sources of wastes within that process; and
 - (c) the feasibility of the following waste reduction or prevention techniques:
 - (i) product reformulation;
 - (ii) clean production technologies;
 - (iii) process modification;
 - (iv) input substitution; and
 - (v) on-site, closed-loop recycling.
3. In general terms, if the required audit reveals that opportunities exist for waste prevention at its source, an applicant for a permit is expected to formulate and implement a waste prevention strategy, in collaboration with the relevant local, provincial and national agencies, which includes specific waste reduction targets and provision for further waste prevention audits to ensure that these targets are being met. Permit issuance or renewal decisions must assure compliance with any resulting waste reduction and prevention requirements.
4. For dredged material and sewage sludge, the goal of waste management should be to identify and control the sources of contamination. This should be achieved through implementation of waste prevention strategies and requires collaboration between the relevant local, provincial and national agencies involved with the control of point and non-point sources of pollution. Until this objective is met, the problems of contaminated dredged material may be addressed by using disposal management techniques at sea or on land.

CONSIDERATION OF WASTE MANAGEMENT OPTIONS

5. Applications to dump wastes or other material must demonstrate that appropriate consideration has been given to the following hierarchy of waste management options, which implies an order of increasing environmental impact:
 - (a) re-use;
 - (b) off-site recycling;
 - (c) destruction of hazardous constituents;
 - (d) treatment to reduce or remove the hazardous constituents; and
 - (e) disposal on land, into air and in water.
6. The Minister will refuse to grant a permit if it is established that appropriate opportunities exist to re-use, recycle or treat the waste without undue risks to human health or the environment or disproportionate costs. The practical availability of other means of disposal should be considered in the light of a comparative risk assessment involving both dumping at sea and the alternatives.

CHEMICAL, PHYSICAL AND BIOLOGICAL PROPERTIES

7. A detailed description and characterisation of the waste is an essential precondition for the consideration of alternatives and the basis for a decision as to whether a waste may be dumped. If a waste is so poorly characterised that a proper assessment cannot be made of its potential impacts on health and the environment, that waste may not be dumped.

Characterisation of the wastes and their constituents must take into account—

- (a) origin, total amount, form and average composition;
- (b) properties: physical, chemical, biochemical and biological;
- (c) toxicity;
- (d) persistence: physical, chemical and biological; and
- (e) accumulation and biotransformation in biological materials or sediments.

ACTION LIST

8. In selecting substances for consideration in the Action List referred to in section 73, the Minister will give priority to toxic, persistent and bioaccumulative substances from anthropogenic sources (e.g., cadmium, mercury, organohalogens, petroleum hydrocarbons, and, whenever relevant, arsenic, lead, copper, zinc, beryllium, chromium, nickel and vanadium, organosilicon

compounds, cyanides, fluorides and pesticides or their by-products other than organohalogens). An Action List can also be used as a trigger mechanism for further waste prevention considerations.

9. The Action List must specify an upper level and may also specify a lower level. The upper level should be set so as to avoid acute or chronic effects on human health or on sensitive marine organisms representative of the marine ecosystem. Application of an Action List will result in three possible categories of waste:
 - (a) wastes which contain specified substances, or which cause biological responses, exceeding the relevant upper level shall not be dumped, unless made acceptable for dumping at sea through the use of management techniques or processes;
 - (b) wastes which contain specified substances, or which cause biological responses, below the relevant lower levels should be considered to be of little environmental concern in relation to dumping at sea; and
 - (c) wastes which contain specified substances, or which cause biological responses, below the upper level but above the lower level require more detailed assessment before their suitability for dumping at sea can be determined.

DUMP-SITE SELECTION

10. The Minister will require at least the following information before deciding whether or not to approve a site for dumping at sea:
 - (a) the physical, chemical and biological characteristics of the water-column and the seabed;
 - (b) the location of amenities, values and other uses of the sea in the area under consideration;
 - (c) the assessment of the constituent fluxes associated with dumping at sea in relation to existing fluxes of substances in the marine environment;
 - (d) the economic and operational feasibility; and
 - (e) any relevant coastal management objectives.

ASSESSMENT OF POTENTIAL EFFECTS

11. *Assessment of potential effects should lead to a concise statement of the expected consequences of the sea or land disposal options, i.e., the "Impact Hypothesis". It provides a basis for deciding whether to approve or reject the proposed disposal option and for defining environmental monitoring requirements.*
12. The assessment for dumping at sea must integrate information on waste characteristics, conditions at the proposed dump-site or dump-sites, fluxes, and proposed disposal techniques and specify the potential effects on the environment, human health, living resources, amenities and other legitimate uses of the sea. It must define the nature, temporal and spatial scales and duration of expected impacts based on reasonably conservative assumptions.
13. An analysis of each disposal option must be considered in the light of a comparative assessment of the following concerns: human health risks, environmental costs, hazards, (including accidents), economics and exclusion of future uses. If this assessment reveals that adequate information is not available to determine the likely effects of the proposed disposal option then this option may not be considered further. In addition, if the interpretation of the comparative assessment shows the dumping at sea option to be less preferable, a permit for dumping will not be given.
14. Each assessment must conclude with a statement supporting a decision to issue or refuse a permit for dumping at sea.

MONITORING

15. Monitoring is used to verify that permit conditions are met—compliance monitoring—and that the assumptions made during the permit review and site selection process were correct and sufficient to protect the environment and human health—field monitoring. It is essential that such monitoring programmes have clearly defined objectives.

PERMIT AND PERMIT CONDITIONS

16. A decision to issue a permit will only be made if all impact evaluations are completed and the monitoring requirements are determined. The conditions of the permit must ensure, as far as practicable, that adverse effects are minimised and the benefits maximised. A dumping permit issued must contain data and information specifying—
 - (a) the types and sources of materials to be dumped;
 - (b) the location of the dump-site(s);
 - (c) the method of dumping at sea; and
 - (d) monitoring and reporting requirements.
17. The Minister will review permits for dumping at sea at regular intervals, taking into account the results of monitoring and the objectives of monitoring programmes. Review of monitoring results will indicate whether field programmes need to be continued, revised or terminated and will contribute to informed decisions regarding the continuance, modification or revocation of permits. This provides an important feedback mechanism for the protection of human health and the marine environment.

Appendix 3²

GUIDELINES FOR THE MANAGEMENT OF DREDGED SPOIL IN SOUTH AFRICAN COASTAL WATERS

1. Introduction

Dredging activities can have a variety of negative effects on marine organisms, from habitat disturbance for benthic communities in the dredged area, to physical smothering or chemical contamination of those on the disposal site. Inappropriate selection of disposal sites can also result in interference with fishery activities, recreation and navigation. It is therefore important that dredging activities be properly managed, both by limiting dredging to what is strictly necessary, and by controlling the use and/or disposal of dredged materials.

2. Legal considerations

South Africa is a contracting party to the London (Dumping) Convention, which it ratified in 1978. The main provisions of the Convention were incorporated into local legislation in the form of the Dumping at Sea Control Act 73 of 1980, which came into force on 23 April, 1982. With respect to the dumping of dredged spoil into marine or estuarine waters, the London Convention:

- a. Prohibits the dumping of dredged material containing Annex I³ substances, unless:
 - the physical, chemical conditions will ensure that they are "rapidly rendered harmless"; or
 - they are present only as "trace contaminants"; or
 - in the case of radioactive materials, at "de minimis" levels.

In addition, they should not make edible organisms unpalatable or endanger human health or that of domestic animals.

- b. Requires that dredged materials containing Annex II⁴ substances in more than "significant amounts" be issued special permits which prescribe "special care" measures and/or other limiting conditions.
- c. In addition, any permits issued for dumping of dredged spoil must only be granted after an assessment of the factors contained in Annex III⁵.

The implementation of the Convention with respect to dumping of dredged spoil thus requires the interpretation of the clauses "rapidly rendered harmless", "trace amounts" "de minimis levels" and "significant amounts". While these issues are the subject of ongoing investigation and debate at the international level, the objective of this document is to establish guidelines for their interpretation in the South African situation with a view to providing a basis for the assessment and management of dredged spoil disposal in our coastal waters. To begin with, it is useful to review existing interpretations.

a. "Rapidly rendered harmless"

In terms of Interim Guidelines adopted and amended at various Consultative Meetings of the contracting parties to the London Convention, "rapidly rendered harmless" is taken to mean: "if tests of the waste or other matter proposed for dumping, including tests on the persistence of the material, show that the substances can be dumped so as not to cause acute or chronic effects or bioaccumulation in sensitive marine organisms typical of the marine ecosystem at the disposal site. A persistent substance should not be regarded as "harmless" except when it is present as a "trace contaminant".

The Interim Guidelines also outline test procedures to be used in making such assessments.

b. "Trace amounts"

The above-mentioned Interim Guidelines, while they do not give any recommendations on what levels of contaminants should be regarded as "trace amounts", do give a list of what should not be considered as trace contaminants. These include:

- a) If they are present in otherwise acceptable wastes or other materials to which they have been added for the purpose of being dumped;
- b) if they occur in such amounts that the dumping of the wastes or other materials could cause undesirable effects, especially the possibility of chronic or acute toxic effects on marine organisms or human health whether or not arising from their bioaccumulation in marine organisms and especially in food species; and
- c) if they are present in such amounts that it is practical to reduce their concentration further by technical means."

c. "Significant amounts"

The 8th Consultative Meeting of the London Convention agreed on the following interpretations:

² Minor alterations to the original document have been made, mainly in terms of grammar and layout.

³ Annex I substances include organohalogen compounds, mercury and mercury compounds, cadmium and cadmium compounds, persistent plastics and other synthetic materials, crude oil, fuel oil, heavy diesel oil, lubricating oils, hydraulic fluids, radioactive wastes or matter, and materials produced for biological or chemical warfare.

⁴ Annex II substances include arsenic, lead, copper and its compounds, zinc and its compounds, organosilicon compounds, cyanides, fluorides, pesticides and their by-products, beryllium, chromium, nickel, vanadium, and bulky metallic wastes.

⁵ Criteria to be considered include the characteristics and composition of the matter, the characteristics of the dumping site and method of disposal, the possibility of effects on other uses of the sea and on marine life, and the practical availability of alternative, land-based methods of treatment or disposal.

Pesticides (other than those covered by Annex I) and their by-products, and lead and lead compounds.

0.05% or more by weight in the waste or other matter.

Arsenic, copper and its compounds, zinc and its compounds, organosilicon compounds, cyanides, and fluorides.

0.1% or more by weight.

d. "De Minimis" Levels

The concept of "de minimis" levels for radioactive materials was introduced after the Resolution by Contracting Parties at the Consultative Meeting in November 1993 to permanently phase out the dumping of radioactive waste at sea - except for waste in which radioactive material was present only at "de minimis" levels. The International Atomic Energy Agency are currently working on producing recommendations as to what these levels should be.

3. Action levels

The interpretations outlined above do little to provide concrete criteria against which sediments can be assessed for permitting purposes - at least for Annex I substances - and essentially allow national authorities total discretion over what is acceptable or not. Their application is also dependent to a large extent on complex assessment procedures involving both chemical analysis and bioassays. For many contracting parties such testing is simply impracticable, and, especially since the general acceptance of the Waste Assessment Framework⁶, there has been a move to establish "Action Levels" which would streamline assessment procedures. It is proposed that such "Action Levels" form an integral part of the assessment of dredged spoil in South Africa. Table 1 below therefore reviews "Action Levels" as adopted by other contracting parties, while Table 2 (in section 4) makes proposals for "Action Levels" for South Africa.

Table 1

ANNEX I SUBSTANCES

	Hong Kong ⁷	Iceland ⁸	Ireland ⁹	Germany ¹⁰	Norway ¹¹	Canada ¹²	Quebec	Ontario
<u>Heavy metals</u> (ppm dry weight)								
Cadmium	1.0-1.5	0.5-1.5	4.5	2.5-12.5	1.0-10.0	0.6-3.0	5	0.1
Mercury	0.7-1.0	0.25-1.25	1.0	1.0-5.0	0.6-5.0	0.75-1.5	0.3	0.3
<u>Organohalogen compounds</u>								
PCB's		0.38					0.05	0.05
Organochlorine							0.05	
Hexachlorobenzene							0.05	
<u>Persistent plastics</u>						4% by volume, suitably comminuted		
<u>Oils</u>						Any quantity that yields less than or equal to 10 ppm of n-hexane-soluble substances.	1000 ppm (dry weight)	1500 ppm
<u>Persistent plastics</u> : Canada: 4% by volume, suitably comminuted								
<u>Oils</u> Canada: Any quantity that yields less than or equal to 10 ppm of n-hexane-soluble substances. Quebec: 1000 ppm (dry weight) Ontario: 1500 ppm								

⁶ The WAF is a practical procedure which has been provisionally adopted by the Contracting Parties to the London Convention for the purposes of managing wastes in compliance with the terms of the Convention by providing a set of technical protocols for the evaluation of wastes and associated circumstances.

⁷ Sediments containing Annex I substances in quantities higher than the lower level, special care techniques must be applied. For those containing levels higher than the upper level, there must be effective isolation from the environment.

⁸ Sediments with Annex I substances higher than the levels indicated in the table are prohibited from dumping.

⁹ Sediments with Annex I substances higher than the levels indicated are prohibited from dumping.

¹⁰ Sediments containing substances higher than the lower level given in the Table require Special Permits, while those with levels above the upper limit require an Environmental Impact Assessment.

¹¹ Sediments with levels of substances above the lower levels shown in the Table require "special care" procedures, while those with levels above the upper limits are prohibited from dumping.

¹² For Schedule I substances under the Ocean Dumping Control Act (1975 source - may be out of date).

ANNEX II SUBSTANCES

	Hong Kong	Iceland	Ireland	Germany	Norway	Quebec	Ontario
Heavy metals (ppm dry weight)							
Arsenic	-	2.5-5.0	-	30-150	80-1000	3	8
Chromium	50-80	400-1200	-	150-750	300-5000	70	25
Copper	55-65	500-1000	-	40-200	150-1500	30	25
Lead	65-75	5-50	-	100-500	120-1500	20	50
Nickel	35-40	30-600	-	50-250	130-1500	-	25
Zinc	150-200	250-1000	-	350-1750	650-1000	80	100
Cyanide	-	-	-	-	-	-	0.1

4. Guidelines for the issuing of permits for dumping of dredged spoil in South African waters

Once an application has been received for the dumping of dredged spoil, it needs to be evaluated in terms of our obligations as a Contracting Party to the London Convention and the various technical criteria which have been established in terms of that Convention. In order to facilitate this process, these criteria have been built in to the decision-making procedure shown below, while the various assessment procedures are detailed in Section 5.

Decision-making procedure

1. Is the dredging operation strictly necessary?

No	Reject application
Yes	2
2. Is the dredging part of a new development, or is it maintenance dredging?

New	3
Maintenance	4
3. Is the site contaminated or uncontaminated? (See Section 5A for preliminary assessment procedure)

Uncontaminated	5
Contaminated	7
4. Is the site contaminated or uncontaminated? (See Section 5B for assessment procedure)

Uncontaminated	5
Contaminated	7
5. Can the sediment be used for an alternative purpose? (See Section 5C)

Yes	No permit
No	6
6. Will dumping at the proposed site lead to interference with other activities? (See Section 5D)

Yes	No permit
No	Issue General Permit
7. What is the level of contamination? (See Section 5E)

"Trace" or "insignificant" contamination	5
Moderately contaminated	8
Highly contaminated	11
8. Can the sediment be used for an alternative purpose? (See Section 5C)

Yes	No permit
No	9
9. Is the disposal of the material at the proposed site likely to result in harmful effects to marine organisms or to human health? (See Section 5F)

Yes	10
No	Issue Special Permit
10. Can the material be effectively isolated from the environment using "Special Care" techniques? (See Section 5G)

Yes	Issue Special Permit with appropriate conditions [#]
No	11

[#] Conditions in these applications must include a commitment to investigating and addressing the source of contamination i.e. it should be made clear that permits for contaminated sediments will not be made on an ongoing basis, and that a programme must be established to reduce or eliminate the problem at source.

Special permits - with specific conditions - may be issued for moderately contaminated sediments if:

- biological testing indicates that the impacts outside of the disposal site will be minimal
- marine disposal is considered to be the disposal option of least detriment to the environment
- there is a commitment on the part of the applicants to the implementation of a source reduction programme

11. Are land treatment and/ or disposal options likely to be more or less harmful to the environment as a whole? (See Section 5H)

More harmful

Issue special permit

Less harmful

No permit (recommend land disposal)

5. Assessment Procedures/Management Techniques

A: Preliminary assessment of dredging site for new development

The first step is to evaluate the status of the site to be dredged on the basis of existing information.

- * Are there any technical reports available on the sediments of the area concerned?
 - i) If so, do they suggest that the area is contaminated or uncontaminated. If contaminated, then a full chemical assessment must be undertaken.
 - ii) If the reports indicate no contamination, then the question must be asked as to whether the position is likely to have changed since the publication of the report/s?
- * Has there been any industrial development or establishment of any other potential source of pollutants in the vicinity of the site since the report? Other potential sources include rivers which may be carrying pollution loads from sources upstream. If so, then a full chemical assessment must be undertaken. If not, then the site can be assumed to be uncontaminated.
- iii) If there are no reports, then information must be obtained as to whether there is any industrial development or other potential source of pollutants in the vicinity of the site. Other potential sources include rivers which may be carrying pollution loads from sources upstream. If so, then a full chemical assessment must be undertaken. If not, then the site can be assumed to be uncontaminated.

B: Assessment of maintenance dredging operation

The first step again should be an evaluation of existing information.

- i) If the sediments in the area have been tested and demonstrated to be clean (against the criteria outlined in Section 5E below) within the last 5 years, and no new potential sources of contaminants have been established, then the sediments can be considered to be uncontaminated.
- ii) On the other hand, if the sediments in the area have either never been tested, or they have previously been shown to be contaminated, then they must be tested using the procedures outlined in Section 5E.

C: Beneficial Uses

Dredged spoil - particularly if it is uncontaminated - can be used for a variety of purposes, including:

- i) Engineering purposes:
 - Land creation
 - Land improvement
 - Offshore berms
 - Capping
 - Replacement fill
 - Beach nourishment (currently accounts for some 70% of dredged spoil in South Africa, primarily in Durban and Richards Bay).
- ii) Agricultural/ Product uses
 - Topsoil
 - Aquaculture
 - Construction materials
- iii) Environmental enhancement
 - Wetland creation
 - Upland habitat
 - Fisheries improvement

The harbour authorities should actively investigate opportunities to make constructive use of dredged spoil.

D: Dumpsite selection

The number of dumpsites used should be limited as far as possible. In selecting dumpsites, the following issues should be taken into consideration:

- i) Dumping should not lead to interference with other beneficial uses including shipping, fishing, recreation, mariculture etc.
- ii) Dumping should not impact on ecologically important sites such as spawning grounds, feeding grounds, or habitats of vulnerable or endangered species.
- iii) Dumping should not take place in areas where the physical characteristics make it vulnerable to the build-up of pollutants which could lead to eutrophication, oxygen depletion, blanketing of the seafloor etc.

- iv) Where wave and current movement at the dumpsite is limited, there should be a dispersive style of disposal to facilitate the spread of dumped materials.
- v) Sediments in the dredged material and the receiving area should be similar as far as possible.

E: Chemical Characterisation of Sediments

This process should provide a general description of the sediments, as well as an indication of the level of contamination by Annex I and Annex II substances. The results can then be compared to the "Action Levels" to categorise the sediments into one of three groups as follows:

- i) Trace or insignificant contamination
- ii) Moderately contaminated
- iii) Highly contaminated

The steps to be followed are:

a) Sampling

Sampling of sediments from the proposed dredging site should represent the vertical and horizontal distribution and variability of the material to be dredged. Samples should be spaced so as to identify between non-contaminated and contaminated locations. For further details of sampling procedures, see Section 6A.

b) General description

- total volume to be dredged, as well as a breakdown of estimated volumes from each area of the harbour.
- visual determination of sediment characteristics (clay-silt/sand/gravel/boulder) (for contaminated sites a proper grain-size analysis might be required).
- total organic carbon (?).

c) Chemical Analysis: List of contaminants to be measured.

At the time of a first application for dumping from a specific harbour area - or at the time of the first application after the introduction of these guidelines - the sediments should, as a minimum, be analysed for the following Annex I and Annex II substances:

Annex I Substances

- organohalogen compounds : PCB's, organochlorine, hexachlorobenzene
- mercury and mercury compounds
- cadmium and cadmium compounds
- persistent plastics
- petroleum hydrocarbons
- radioactive materials (?)
- materials produced for biological or chemical warfare (?)

Annex II Substances

- arsenic
- lead
- copper and its compounds
- zinc and its compounds
- beryllium
- chromium
- nickel
- vanadium
- organosilicon compounds
- cyanides
- fluorides
- pesticides and their by-products
- bulky metallic wastes (?)

In addition, where there are known sources of particular pollutants not listed here, such pollutants should be included in the analyses. (Need to be more specific in some cases e.g. for pesticides). Analytical techniques to be utilised are outlined in Section 6B.

Should the initial survey indicate the presence of only a limited number of these contaminants, then the list could be limited accordingly in discussion with SFRI, and on a case by case basis.

d) Comparison with Action Levels

Sediments will be considered to be uncontaminated (trace or insignificant contamination) if they contain levels of Annex I and Annex II substances less than the lower of those levels shown in Table 2 below:

Table 2

ANNEX I SUBSTANCES

Suggested Action/Prohibition levels (based on values from the international literature as shown in the Tables below):

Range(ppm)	Action level	Prohibition level
Cadmium	1.5-10.0	>10,0
Mercury	0.5-5.0	>5.0
or for a combined level of these two	1.0-5.0	>5.0
Organohalogens	0.05-0.1	>0.1
Oils	1000-1500	>1500
Persistent plastics: 4% by volume, suitably comminuted		
Radioactive materials: to be determined by the IAEA		

ANNEX II SUBSTANCES

Suggested Special care/Prohibition levels

Range(ppm)	Special care	Prohibition level*
Arsenic	30-150	>150 (1000)
Chromium	50-500	>500
Copper	50-500	>500 (1000)
Lead	100-500	>500 (500)
Nickel	50-500	>500
Zinc	150-750	>750 (1000)
or a combined level of these substances:	50-500	>500 (1000)
Cyanides	0.1	(1000)
Fluorides		(1000)
Organosilicon compounds		(1000)
Pesticides		(500)

* According to the agreement at the 8th Consultative Meeting, significant amounts of these substances were $\geq 0.1\%$ by weight, or 0.5% by weight for lead and pesticides.

F: Biological testing

While chemical analysis may reveal the presence of various contaminants in sediments, this does not necessarily mean that the sediments will result in biological harm. The contaminants may, for example, be present in a form in which they are not "available" to organisms i.e. they are biochemically inert. The next step for moderately contaminated sediments, therefore, would be to test for such bioavailability. Such tests should measure for acute and chronic toxicity to selected marine organisms, as well as the potential for bioaccumulation. Details of testing procedures to be used are outlined in Section 6C.

G: "Special Care" Techniques

Some of the "special care" techniques reportedly used by other countries include:

- * careful selection of dump sites, especially for highly polluted sediments
- * silt screens to avoid the spread of fine particulate matter for moderately polluted sediments
- * effective isolation from the environment for highly polluted sediments by dumping into specialised marine disposal pits and capping with clean sediment.

H: Comparative assessment of land-based alternatives**6. Sampling Procedures and Analytical Techniques**

General: Laboratories that perform the analyses should be "accredited". The accreditation procedure would take the form of an intercalibration exercise to be set up by the Pollution Division of Sea Fisheries Research Institute, with SFRI supplying a set of reference sediments. This would be repeated every 3 years.

A: Sediment sampling

Sampling of sediments from the proposed dredging site should represent the vertical and horizontal distribution and variability of the material to be dredged. Samples should be spaced so as to identify between non-contaminated and contaminated locations. Sampling should be undertaken using a barrel core as described in Loring and Rantala (1992). (see attached).

B: Analytical techniques (chemical)**Heavy metals**

Sample preparation: Teflon bomb acid digestion as described in Loring and Rantala (1992) Analysis:

- Mercury: Cold vapour atomic absorption (Loring and Rantala, 1992).
- Cadmium, Lead, Copper, Zinc, Beryllium, Chromium, Nickel, Vanadium: Flame and graphite furnace Atomic Absorption.
- (Loring and Rantala, 1992).
- Arsenic: Hydride generation Atomic Absorption

Hydrocarbons

Two tests should be conducted: one for total hydrocarbons, and one for polycyclic aromatics (PAHs).

- Extraction: One phase CHCL₃-MeOH Bligh and Dyer method (White et al, 1979).
- Separation: Aliphatic hydrocarbons and PAH by silicic acid column chromatography (Leeming and Nichols, 1988).
- Total hydrocarbons: IR ?
- Gas chromatography (Nichols et al, 1988).
- PAH analysis: Gas chromatography - Mass Spectrometry using 2,3 benzfluorine as an internal standard(Nichols et al, 1988).

Organohalogens

Plastics

C: Biological testing procedures

- i) Acute toxicity
- ii) Chronic toxicity
- iii) Bioaccumulation

7. References

- Leeming,R. & Nichols,P.D. (1991). An integrated scheme to analyse naturally occurring and pollutant organic constituents in urban sewage. Proceedings Bioaccumulation Workshop. Sydney.
- Loring,D.H. & Rantala,R.T.T. (1992). Manual for the geochemical analyses of marine sediments and suspended particulate matter. Earth Science Reviews 32: 235 -283.
- Nichols,P.D., Volkman,J.K., Palmisano,A.C., Smith,G.A. and White,D.C. (1988) Occurrence of an isoprenoid C₂₅ diunsaturated alkene and high neutral lipid content in Antarctic sea-ice diatom communities. J.Phycology 24: 90 - 96.
- White,D.C., Bobbie,R.J., Herron,J.S., King,J.D. and Morrison,S.J. (1979) Biochemical measurements of microbial biomass and activity from environmental samples. IN Native Aquatic Bacteria: Enumeration, Activity and Ecology. (Eds) J.W. Costerton and R.R.Colwell. pp. 69 - 81. (ASTM STP 695).

Appendix 4

Modified National Action List for South Africa (the original National Action list is presented in Appendix 3)

Range(ppm)	Action level	Prohibition level
Cadmium	1.5-10.0	>10.0
Mercury	0.5-5.0	>5.0
or for a combined level of these two	1.0-5.0	>5.0
Arsenic	30-150	>150 (1000)
Chromium	50-500	>500
Copper	50-500	>500 (1000)
Lead	100-500	>500 (500)
Nickel	50-500	>500
Zinc	150-750	>750 (1000)
or a combined level of these substances:	50-500	>500 (1000)

Explanatory notes on application:

1. Once the levels of contamination in the sediments have been determined through chemical analysis, these are compared with the Action Levels contained in the above List (presented as $\mu\text{g.g}^{-1}$ dry weight sediment).
2. A decision on whether or not to require biological testing, or to prohibit disposal of the sediment at sea, is determined as follows:
 - a) If none of the metals measured exceed the Action Levels, then no biological testing is required, and the material can be dumped.
 - b) If the Action Levels for both Annex I metals (Cd and Hg) are exceeded, or the combined level of Cd and Hg is $>5 \mu\text{g.g}^{-1}$, then biological testing is required.
 - c) If the Action Level for either of the Annex I metals, and two or more of the Annex II metals are exceeded, then biological testing is required.
 - d) If the Action Levels of three or more Annex II metals are exceeded, and the total of Annex II metals is $>500 \mu\text{g.g}^{-1}$, then biological testing is required.
 - e) If the combined level of Annex II metals is $>1\ 000 \mu\text{g.g}^{-1}$, then biological testing is required.
 - f) If either of the Prohibition Levels for the Annex I metals is exceeded, or if the Prohibition Level of two or more of the Annex II metals is exceeded, dumping will not be allowed.