DEVELOPING SEDIMENT QUALITY GUIDELINES FOR SOUTH AFRICA

PHASE 1: Identification of international best practice and applications for South Africa to develop a research and implementation framework.

Report to the
Water Research Commission

by

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EXECUTIVE SUMMARY

Rationale

The issue of sediment contamination in South African freshwaters has been largely ignored. Currently no sediment quality guidelines (SQGs) exist for freshwaters in this country. The objective of this project was to critically review SQG derivation methods being utilised internationally, and to identify specific factors that need addressing in order to derive and implement effective SQGs in South Africa.

Project aims

The aims for this project as detailed in the research contract with the Water Research Commission (WRC) were to:

1. Review international best practices and methods for derivation of sediment quality guidelines, identifying which types of contaminants the South African guidelines will represent, and whether sediment as a physical stressor be considered.
2. Determine the most appropriate derivation method for the South African sediment quality guidelines, taking into consideration whether the methodology for the sediment guidelines should be aligned with that used to derive water quality guidelines in South Africa and how local data (if any are available) should be incorporated into the derivation procedure.
3. Identify methods for generating locally relevant data with which to refine the guidelines for South Africa as a whole or for specific site assessments. This may include identifying sediment toxicity tests which may be developed or adapted for use with indigenous organisms.
4. Using information gathered from the above three aims, propose a research programme which will likely include 1) the development of sediment toxicity tests for indigenous organisms, 2) the actual derivation of the guidelines for South Africa, and 3) the development of a framework for their implementation.

Report structure

Aspects of each aim are discussed in different chapters within this report. Furthermore, as a result of decisions taken from the workshop, some aims will require further investigation.

Chapter 1 presents the rationale for this project, and the project's objective and aims.

Chapter 2 details a literature review of international approaches to developing and implementing SQGs and an overview of the complex physical and chemical characteristics of sediments (Aim 1).

Chapter 3 details the outcomes of a workshop held with South African scientists where the remaining issues from Aim 1 were discussed along with those from Aims 2 and 3.

Chapter 4 details a proposed future research framework (Aim 4).

Sediment environment

The sediment environment is complex with a multitude of interacting factors. Sediments are heterogeneous and sediment characteristics (e.g. size and chemical composition of the sediment particles, pH and redox potential of the overlying or interstitial water) that alter contaminant bioavailability to organisms vary over short distances, both laterally and vertically within sediments. Furthermore, sediment associated organisms have many different exposure routes, most of which are poorly understood. The accuracy of any derived sediment quality guideline will be limited by the ability to measure and incorporate these factors that account for bioavailability.

International methods for SQG derivation

There are three broad approaches to deriving SQGs: mechanistically based guidelines; empirically based guidelines and consensus based guidelines.

Mechanistic approach

The mechanistic approach assumes that the critical factor controlling sediment toxicity is the concentration of the contaminant in the sediment interstitial water. Methods include applying water quality guidelines (WQGs) directly to interstitial water contaminants, or after a partitioning coefficient had been applied, to the whole sediment sample. The equilibrium partitioning (EqP) approach was developed to take account of factors influencing chemical bioavailability in sediments. Approaches for nonionic organic and metal contaminants have been developed separately. For the organics the WQG for a particular chemical is multiplied by the partition coefficient of the chemical to organic carbon in the sediment to derive the SQG. For metals, partitioning is more complex and the key partitioning phase affecting toxicity is presence of sulfides. Consequently, the amount of sulfide (determined as acid volatile sulfide – AVS) in a sediment sample is compared to the metal concentration simultaneously extracted (SEM – simultaneously extracted metal). When AVS concentrations exceed the sum of the SEM concentrations, no acute toxicity is observed. However, the exceedance of the AVS concentrations by the SEMs does not necessarily result in mortality and for this reason the AVS method is used only to predict when sediment is not acutely toxic.

The advantages of the EqP method for deriving SQGs are: that the bioavailable fraction of the chemical in the bulk sediment sample is considered and is thus applicable across almost all sediment types; the guidelines are causally linked to a specific chemical; the SQGs are linked to the WQGs and therefore a level of protection can be specified, which is attractive to regulators; there are more toxicity data for water column exposures and thus the use of EqP approach allows for the incorporation of data from a wider range of species and; the approach is based on fundamental toxicological principles.

Disadvantages are that this method does not protect against additive, synergistic or antagonistic effects of other compounds or protect against bio-accumulative effects. Furthermore, EqP of polar organics can overestimate the bioavailable concentration because the binding of these organics to sediments can be enhanced by factors other than hydrophobicity. There is evidence that the SEM:AVS approach is less accurate at predicting nontoxicity and toxicity than selected empirical approaches, and that the guideline values of the empirical approaches were more sensitive than the SEM:AVS values.

Empirical approach

The empirical approach generally derives guidelines using data from biological responses to contaminated sediments (the exception is the use of background concentrations to set protective SQGs). These can be concentration-response data for known concentrations of single or mixed contaminants provided by spiked sediment toxicity tests (SSTTs) or biological response data obtained from field collected sediments or field surveys of benthic populations and/or communities. Generally, the derivation methods which employ empirical data utilise various percentiles of effects (ranked concentrations at which biological responses measured) and/or non-effects (ranked concentrations at which no biological response measured) databases to determine an effect level or benchmark. These "benchmark" values define ranges of concentrations that are associated with biological effects. Below the lower benchmark adverse biological effects are unlikely, above the upper benchmark adverse effects are a virtual certainty, and the range in between represents a gradient along which effects are increasingly likely. This is a way of dealing with the many uncertainties of the data on which the guidelines are based.

Guidelines developed utilising spiked sediment toxicity test data only have been applied to very few chemicals as there are very few possible benthic test organisms available. Although this approach considers causality (i.e. the specific chemical causing the biological effect is known) it does not consider bioavailability and is often criticized as being environmentally unrealistic as tests are conducted using hardy laboratory organism.

Guidelines derived from toxicity tests exposing the few available laboratory organisms to field collected sediments include the Sediment Effect Concentration and Logistic Regression Modeling methods. These approaches do not take account of causality or bioavailability and can also be criticized as being environmentally unrealistic.

A guideline derivation method using only concurrently measured community/population responses and contaminant concentration in field sediments is the Screening Level Concentration approach (this approach was specifically applied to polar organic chemicals, e.g. PAHs). In this case presence/absence of specific benthic biota is compared to the contaminant concentrations. Once again, this approach does not take account of causality or bioavailability, and sublethal biological effects were not measured.

Guidelines derived using a combination of toxicity tests and benthic community/population response data include the Apparent Effects Threshold (AET) approach, The National Oceanic and Atmospheric Administration (NOAA) approach and the Florida Department of Environmental Protection (FDEP) approach. Although the AET approach partially addresses bioavailability (organic contaminants are normalized to 1% organic carbon), the others did not. None of the approaches considers causality.

Using background concentrations to derive SQGs is based on the assumption that contaminant concentrations that are not higher than background are not hazardous. This approach was developed in response to uncertainty regarding the ecological realism of empirically and theoretically (EqP) derived SQGs because of the uncertainties of extrapolating data from the laboratory to the field, the impossibility of accounting for the sensitivities of all sediment dwelling organisms, and large variability of physical sediment characteristics over small geographical areas affecting the bioavailability of chemicals. A study has shown that overall metal concentrations in Norwegian marine sediments eliciting effects on biota were 3.6 times higher than background levels, and are much more conservative than current international SQGs, suggesting current SQGs are not adequately protecting marine sediment fauna. Disadvantages include: the approach has no biological effects basis, and it cannot be used for synthetic organic compounds, which should not be present in background sediments. In addition, measured concentrations exceeding background levels do not necessarily represent a hazard to organisms. However, background concentrations may be useful in screening other SQGs, highlighting where the guideline is within the range of background concentrations and thus should not be used to assess ecological hazard.

Lastly, the Consensus approach involves collating previously published SQGs in order to provide a unifying synthesis and attempt to address issues of bioavailability and causality. The SQG values were categorized into three groups according to their original narrative intent (i.e. likely or not likely to indicate hazardous biological effects). It was found that within each of the groups the SQG values agreed within a factor of two to four (depending on which SQGs were included). Field validation of the consensus guidelines have showed they can successfully predict sediment toxicity and benthic community perturbations at sites of PAH contamination.

Methods adopted by various countries

The USA has not adopted national SQGs and, although the USEPA is pursuing the EqP method, in practice various empirical methods are utilised by individual agencies. Guidelines adopted by Australia and New Zealand, Canada and Hong Kong are based on the empirical approach. Before the establishment of the European Union (EU), the French national guidelines and the guidelines of individual agencies in the UK, Italy and Germany utilised empirical data, while only the Netherlands derived national SQGs using EqP theory. Under the Water Framework Directive the EU are considering proposals on how to develop sediment environmental quality standards. No SQGs have been derived for South African freshwaters, however guidelines were derived for marine sediments utilizing the empirical approach.

Sediment quality guideline derivation and application in the South African context

There is a need for SQGs to be developed for South African freshwaters. The process, however, is a complex one needing interaction and collaboration among scientists, regulators and implementers. Consequently, this document raises issues that will need further investigation by particular working groups.

The aim of the SQGs should be explicitly stated and would in turn dictate the type of data utilised, the derivation and implementation methods employed. The philosophical approach to the SQGs should be based on the approach being developed for the revised water quality guidelines (WQGs) (i.e. a scenario-specific probabilistic risk assessment approach).

The application of the SQGs, from a regulatory point of view, is seen in two contexts: a) assessments (i.e. what can you say about how good or bad the situation is from a set of field collected data) – applicable to monitoring programmes and risk assessments; and b) setting sediment quality objectives (defining what is an acceptable risk to the aquatic environment and determining the sediment quality associated with that risk) – applicable to risk assessments and to ecological Reserve determinations when integrated into resource quality objectives. Within these contexts the SQGs will have to align to the current resource classification system in South Africa in order to be effectively implemented – an issue that is applicable to the proposed WQGs too.

The structure of the SQGs will be similar to that envisaged for the WQGs. The primary tool for facilitating the determination and use of these guidelines will be a software based decision support system (DSS) (still under development). The guidelines will comprise a three tier system:

Tier 1. Provides ‘generic’ guidelines values that are made available in the DSS and hard copy manuals. These guideline values will be conservative as the worst case scenario is assumed.
Tier 2. Allows for site specificity in specified contexts and is facilitated by the DSS, consequently there is more confidence in the derived value.
Tier 3. Full risk assessment providing scenario/site specific guideline value. Not facilitated by the DSS but will use information contained within the DSS information database.

The procedure for generating the guideline values will follow a probabilistic risk assessment process, starting at tier 3 and progressing to tier 1 (the exact procedural methods are still to be resolved). At tier 3 all exposure parameters that would affect the toxicity of the chemical to the organism are identified. At tier 2 only some mandatory parameters are required. Consequently, associated with the move down the tiers is a loss in confidence in the resultant guideline value and thus the need to be more conservative (through use of conservative data, perhaps inclusion of safety factors and/or reducing what is considered acceptable risk to the resource).

As a consequence of the scenario specific approach inherent in the proposed SQG structure at the tier 2 and 3 level, the method for deriving the SQG can be similar to the method for undertaking an assessment of sediment toxicity at a particular site or given a particular scenario.

The type and quality of data to be included in the DSS database for use in deriving the guidelines, and the derivation method itself need to be decided upon by a specialized focus group. There appears to be very little available toxicity test data utilising indigenous organisms or biological response data using standard organisms and South African field collected sediments. Consequently, the SQGs will rely heavily on international data. However, the generation of South African specific data should be prioritized (i.e. the development of benthic organism-based toxicity test, and the capacity to undertake these test in South Africa) and allowances made for the later inclusion of this data in the SGQ derivation process.

Possible further research framework
In order to develop SQGs that are scientifically defensible and applicable to South African water resource management strategies, three main issues requiring further investigation were identified.

1. There is a need to ensure that, the philosophical approach and implementation of the SQGs are aligned with those of the WQGs being currently revised for South Africa, and to determine if the DSS being developed for the WQGs is applicable for the SQGs.
2. Determine the most appropriate data and derivation method for South Africa.
3. Improve the capacity of organizations in South Africa to undertake sediment toxicity testing and analysis of contaminated sediments.

Research tasks and actions aimed at addressing these are detailed in Chapter 4.

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Abbreviations

AET	Apparent effects threshold
ANZECC	Australia and New Zealand Environmental and Conservation Council
ARMCANZ	Agriculture and Resource Management Council of Australia and New Zealand
ASTM	American Society for Testing and Materials
AVS	Acid volatile sulfide
BEDS	Biological effects database for sediments
CCME	Canadian Council of Ministers of the Environment
DEAT	Department of Environmental Affairs and Tourism (South Africa)
DSS	Decision support system
DWAF	Department of Water Affairs and Forestry (South Africa)
EC	Effective concentration
EC50	Median effect concentration
EEC	Extreme effects concentration
EQS	Environmental quality standards
EqP	Equilibrium partitioning
ERA	Environmental risk assessment
ERL	Effects range low (value)
ERM	Effects range medium (value)
ESB	EqP sediment benchmark
ESG	Equilibrium partitioning sediment guideline
EU	European Union
FCV	Final chronic value
FDEP	Florida Department of Environmental Protection
ISO	International Organisation for Standards
ISQV	Interim sediment quality value
LAET	Lowest apparent effects threshold
LC50	lethal concentration for 50% of test organisms
LRM	Logistic regression modelling
MEC	Median effects concentration
NOAA	National Oceanic and Atmospheric Association (USA)
OC	Organic carbon
OECD	Organisation of economic cooperation and development
PAH	Polycyclic aromatic hydrocarbons
PEC	Probable effects concentration
PEL	Probable effects level
SEC	Sediment effect concentration
SEM	Simultaneously extracted metals
SLC	Screening level concentration
SQG	Sediment quality guideline
SQT	Sediment quality triad
SSD	Species sensitivity distribution
SSLC	Species screening-level concentration
SSTT	Spiked sediment toxicity test
TEC	Threshold effects concentration
TEL	Threshold effects level
TIE	Toxicity identification evaluation
TPAH	Total polycyclic aromatic hydrocarbons
USEPA	United States Environmental Protection Agency

WFD	Water Framework Directive
WOE	Weight of evidence framework
WQGs	Water quality guidelines
WRC	Water Research Commission (South Africa)

CHAPTER 1: BACKGROUND

1.1 Rationale for the project

The management of water resources in South Africa, guided by the National Water Act (1998), places emphasis on the protection of the water resource as a whole (water quality and quantity, instream and riparian habitats, and instream and riparian biota) so as to ensure resources remain fit for use on a sustainable basis. This objective is effected through the implementation of the Reserve, and the establishment of various national monitoring programmes (e.g. National Toxicity Monitoring Programme, River Health Programme and others). Critical inputs into these activities include water quality guidelines and various aquatic biota health indices. The current South African water quality guidelines (DWAF, 1996) are being revised, yet these will still only reflect the effects of dissolved chemicals in the water column, while ignoring both chemicals associated with suspended and settled sediment and the sediments themselves (erosion/sedimentation is a very serious problem in South Africa). As particulate matter can act as a binding site for contaminants, many contaminants ultimately accumulate in sediments, from where they can be released into the water column and transported to uncontaminated sites (sediment as a nonpoint source) and thereby influence surface water quality (ANZECC and ARMCANZ, 2000). In addition, sediments can themselves act as environmental stressors either directly as a result of physical damage to aquatic biota and changes in habitat conditions, or indirectly through acting as a source of bioavailable toxins to benthic and burrowing biota which may come into contact with contaminated sediments Consequently, it is important to be able to identify situations where either the contaminants associated with sediments or the sediments themselves may represent a likely risk to ecosystem health and integrity. The development of sediment quality guidelines for South African freshwaters would therefore contribute an essential tool to the challenge of water resource management.

1.2 Project objective

This report forms the first phase of a comprehensive research programme aimed at deriving and, ultimately, implementing sediment quality guidelines in South African freshwaters to complement existing water quality guidelines (WQGs) and water resource management strategies, monitoring programmes and tools.

This first phase will provide an overview of international approaches to developing and implementing sediment quality guidelines (SQGs); a discussion of the complex physical and chemical characteristics of sediments; details of a workshop undertaken with South African scientists regarding deriving and implementing SQGs in South Africa; and details further possible phases of this research programme

1.3 Project aims

The aims for phase 1 of this project, as detailed in the research contract with the Water Research Commission (WRC) are listed below:

1. Review international best practices and methods for derivation of sediment quality guidelines, identifying which types of contaminants the South African guidelines will represent, and whether sediment as a physical stressor be considered.

2. Determine the most appropriate derivation method for the South African sediment quality guidelines, taking into consideration whether the methodology for the sediment guidelines should be aligned with that used to derive water quality guidelines in South Africa and how local data (if any are available) should be incorporated into the derivation procedure.
3. Identify methods for generating locally relevant data with which to refine the guidelines for South Africa as a whole or for specific site assessments. This may include identifying sediment toxicity tests which may be developed or adapted for use with indigenous organisms.
4. Using information gathered from the above three aims, propose a research programme which will likely include: the development of sediment toxicity tests for indigenous organisms; the actual derivation of the guidelines for South Africa; and the development of a framework for their implementation.

1.4 Report structure

Aspects of each aim are discussed in different chapters within this report. Furthermore, as a result of decisions taken from the workshop, some aims will require further investigation in subsequent phases of this research programme.

Chapter 2 details a literature review of international approaches to developing and implementing SQGs and an overview of the complex physical and chemical characteristics of sediments. (Aim 1).

Chapter 3 details the outcomes of a workshop held with South African scientists where the remaining issues from Aim 1 were discussed along with Aims 2 and 3.

Chapter 4 details a proposed future research framework. (Aim 4).

CHAPTER 2: LITERATURE REVIEW

2.1 The complexities of the sediment environment

The term sediment generally refers to depositional material with grain sizes from sand, through silt to clay (Batley et al., 2005). The surface area of the particle affects the nature and number of binding sites for metal and organic contaminants. Fine sediments often have the highest concentration of contaminants on a dry weight basis as they have a higher relative surface area and thus increased density of sorption sites (Batley et al., 2005).

The chemical composition of a particle also affects its potential as a binding site. In sediments, decaying detrital organic particulate matter is distributed among mineral and amorphous particles. These organic particles, in addition to mineral particles whose surfaces can consist of iron and magnesium oxyhydroxides, sulfides, or are coated with organic material, bind with chemicals forming a particular 'phase' (Power and Chapman, 1992). The bioavailability of the sediment-associated contaminant is controlled by the phase in which it is found (e.g. sediments where metals are bound to sulfide and organics bound to organic carbon are less toxic). This contaminant concentration in the interstitial water is in equilibrium with the phase of the contaminant in the sediment (Di Toro et al., 1990; Di Toro et al., 1991). Redox potential and pH can affect the phase of metals associated with sediment (e.g. desorption and release to the interstitial water, or formation of sulfide and associated fixation and precipitation of trace metals onto sediment particle).

Sediments are naturally heterogenous. Besides the lateral spatial variability in particle size distribution and geochemistry (affecting pH and redox potential) caused by flow variations and underlying or upstream geology, there is also vertical zonation of pH, redox potential and chemical species caused by the oxygen gradient (Batley et al., 2005). The oxic zone is usually very thin (few millimeters in silty sediments and several centimeters in coarser sands) and is underlain by suboxic and then anoxic layers. Consequently, there are many different factors that can influence the bioavailability of a sediment associated contaminant, and these can vary considerably over a very limited distance, both horizontally and vertically.

The contaminant concentration in the interstitial water diffuses into the overlying water and vice versa depending on concentration gradient and sediment porosity (Batley and Maher, 2001). Consequently, organisms which inhabit, or are associated with the benthos, are potentially exposed to contaminated sediments through association with the sediment, the interstitial and/or overlying waters, and through sediment ingestion. The degree of toxicity will depend on which route of exposure to the contaminant the organism experiences (i.e. the bioavailability of the contaminant). Macroinvertebrates living in freshwater sediments live either in burrows, on the superficial layers of the sediment, or submerged in the sediment but with breathing apparatuses exposed to overlying water. Bately et al. (2005) suggested that the bioavailability of a contaminant to a burrowing organism cannot be compared with the concentration of the contaminant in the surrounding interstitial water as the burrow-dweller has the ability to create a microenvironment by irrigating the burrow with comparatively oxygen-rich overlying water and possibly changing the chemistry of water within the burrow. In addition, contaminant bioavailability can also be potentially affected by organism uptake kinetics (food selection; feeding rate; assimilation efficiency, etc.) and selective digestion (Batley et al., 2005).

The sediment environment is consequently very complex. The accuracy of any derived sediment quality guideline will be limited by the ability to measure and incorporate these factors that account for bioavailability (Wenning et al., 2005).

2.2 Methods of sediment quality assessment

The aim of sediment quality assessment for aquatic resources is usually to try answer the questions: is the sediment toxic; which substance in the sediment is causing the toxicity; what is the safe concentration for that chemical in the sediment and; what is the effect of contaminant mixtures on the toxicity in the sediment (McCauley et al., 2000)? Although it would be desirable to have one tool capable of answering all the above questions, this is not the present reality. Consequently, sediment quality assessment usually involves using some combination of the available methods. Six approaches are presented by McCauley et al. (2000):

- Sediment chemistry. Chemical constituents of field collected whole sediment and/or interstitial water samples from impacted and unimpacted/reference sites are measured and compared.
- Benthic community health. Measures such as presence/absence, abundance and diversity are recorded from benthic communities in impacted and unimpacted areas.
- Sediment toxicity tests. In the laboratory, selected organisms are exposed to contaminated field collected sediments to determine responses. In addition, the toxicity of specific chemicals can be determined using spiked sediment toxicity tests. In the field, caged organisms can be placed in contaminated and uncontaminated areas and responses measured.
- Bioaccumulation tests. Organisms exposed to contaminated sediments are assessed to determine if chemicals are accumulating within them.
- Toxicity identification and evaluation (TIE). This method is designed to characterise the cause of the toxicity by physically or chemically manipulating the interstitial water until the likely cause of toxicity is discovered.
- Sediment quality triad. Combines sediment chemistry analyses with sediment toxicity assessments and benthic community structure studies to better answer the questions posed above.

The above sediment quality assessment approaches, either individually or in combination, have been used as the basis for deriving SQGs.

2.3 International methods for SQG derivation

Scientists and regulatory authorities have investigated a wide variety of methods for deriving effective sediment quality guidelines (SQGs) (Table 2.1). These can be divided into three distinct categories based on the method of derivation: mechanistically based guidelines; empirically based guidelines and consensus based guidelines. The various guidelines have been derived for either fresh or saltwater environments and the methods employed are discussed in sections 2.3.1 to 2.3.3.

2.3.1 Mechanistic approaches for deriving sediment quality guidelines

The mechanistic approach assumes that the critical factor controlling sediment toxicity is the concentration of the contaminant in the sediment interstitial water (ANZECC, 2000). Water quality guidelines are applied to interstitial water contaminants directly, or after a partitioning coefficient had been applied, to the whole sediment sample. Partitioning models were developed to take account of factors influencing chemical bioavailability in sediments.

Direct measurement of interstitial water

Direct measurement of interstitial water compares the concentrations of contaminants in sediment interstitial waters with current regulatory water quality guidelines (WQGs) (the WQGs are derived from empirical data). This method is particularly applicable when interstitial waters represent the dominant phase in which the contaminant is found. This situation can arise if the contaminant was formed in this phase as a result of chemical and microbiological processes, and/or because of its high aqueous solubility (ANZECC, 2000). For example, ammonia is a potentially highly toxic naturally-occurring constituent of interstitial waters but is not generally considered a contaminant of concern in the regulation and management of sediments (ANZECC, 2000). Studies by Moore et al. (1997) and Whiteman et al. (1996) on the relationship between interstitial ammonia concentration and toxicity of sediment dwelling organisms suggests that the comparison of interstitial ammonia concentration with a WQG will provide an adequate prediction of potential harm (ANZECC, 2000).

Although there are arguments that benthic organisms are exposed to contaminants via other exposure routes, such as dermal absorption and ingestion of sediment particles, an analysis of the feeding habits of freshwater benthic species by Adams (1987) concluded that these species were not sediment ingesters, except for the oligochaetes (aquatic earthworms) and some chironomids that are both filter feeders and occasional sediment ingesters (Jones et al., 1997). In contrast to this, marine burrowing species frequently ingest sediment (Adams, 1987).

Maughan (1993) further argues that if the organism is in equilibrium with the interstitial water, then the concentration in the interstitial water would reflect the sum of all exposure routes. Therefore, an organism that has accumulated contaminants, through feeding, at a higher concentration than the equilibrium with interstitial water would re-establish the equilibrium by losing contaminants to the interstitial water. However, factors may influence whether the organism can establish equilibrium with the interstitial water. For example, diffusion within the interstitial water may limit transfer of desorbed compounds to the organism (Landrum and Robbins, 1990).

Difficulties associated with this approach include: there is no universally accepted method for extracting interstitial water from sediment; and interstitial water is difficult to extract from sediment without potentially altering the toxicity of the pore water (Maughan, 1993).

Estimation of interstitial water concentrations: sediment/water equilibrium partitioning approach

The equilibrium partitioning (EqP) method for deriving SQGs is based on the theory that "the bioavailable fraction of contaminants in sediments cause biological effects, and that bioavailability is a function of the partitioning of chemicals between sediments, interstitial water and the benthic organisms" (McCauley et al., 2000, p137). The EqP SQG is defined as the concentration of the contaminant in the sediment that is in equilibrium with the water quality guideline in the interstitial water (ANZECC, 2000). The EqP methods for organic compounds and metals are different due the different interactions of these compounds with other chemical compounds within the sediment. Various EqP approaches have been developed and investigated in the USA (DiToro et al., 1991; Ankley et al., 1996) and in the Netherlands (Van der Kooij et al., 1991).

Nonionic organics

DiToro et al. (1991) showed that the organic carbon (OC) content in sediments dominates the partitioning of organic compounds between sediment particles and interstitial water, affecting their concentration in the interstitial water. The higher the OC in the sediment, the greater the sediment's capacity to adsorb or bind the nonionic organic compound and consequently the lower the bioavailable concentration. Thus in order to derive the SQG (sometimes referred to as the equilibrium partitioning sediment guideline – ESG) the WQG for the particular chemical is multiplied by the partition coefficient (K_{oc}) for the particles' OC and the mass fraction of the OC (f_{oc}) (EPA, 1993b):

$$SQG = K_{oc} \times f_{oc} \times WQG$$

Where the K_{oc} is unavailable, it is estimated by the octanol-water partition coefficient K_{ow} of the chemical for sediments using the following equation (Di Toro, 1985):

$$\mathrm{Log}_{10}(K_{oc})=0.00028 + 0.983 \log_{10}(K_{ow})$$

The EqP approach has four major assumptions (EPA, 1993a): (1) partitioning of the organic chemical between OC and interstitial water is stable at equilibrium: The EqP approach can only be used on sediments with >0.2% OC (dry weight). At f_{oc} <0.2, second order effects such as particle size and sorption to nonorganic mineral fractions become more important (EPA, 1993b);
(2) the sensitivities of benthic species and species tested to derive WQGs, predominately water column species, are similar: Di Toro et al. (1991) demonstrated, using toxicity test data for amphipods, that 100% mortality occurred when the ratio of interstitial water concentration to water-only LC50 exceeded a value of 1. This implied that benthic organisms were as sensitive as water-column organisms (ANZECC, 2000). In addition, the US EPA (1993b) concluded that the sensitivities of benthic species are sufficiently similar to those of water column species to tentatively permit the use of WQGs for the derivation of SQGs;
(3) the levels of protection afforded by the WQGs are appropriate for benthic organisms: if their sensitivities are similar the protection afforded by WQGs should be adequate;
(4) exposures are similar regardless of feeding type or habitat: As discussed under the section describing direct measurement of interstitial water, Maughan (1993) argues that if the organism is in equilibrium with the interstitial water, then the concentration in the interstitial water would reflect the sum of all exposure routes, regardless of feeding type or habitat.

The advantages of the EqP method for deriving SQGs are: that the bioavailable fraction of the chemical in the bulk sediment sample is considered and thus is applicable across sediment types (as long as these sediments possess >0.2% organic carbon); the guidelines are causally linked to a specific chemical; the SQGs are linked to the WQGs and thus level of protection is specified and is thus attractive to regulators; furthermore, because there are more toxicity data for water column exposures, the use of EqP approach allows for the incorporation of data from a wider range of species and; the approach is based on fundamental toxicological principles (McCauley et al., 2000).

Disadvantages are that this method does not protect against additive, synergistic or antagonistic effects of other compounds or protect against bio-accumulative effects. Furthermore, EqP of polar organics can overestimate the bioavailable concentration because the binding of these organics to sediments can be enhanced by factors other than hydrophobicity (Batley and Maher, 2001). Consequently, these guidelines should not be used as pass fail standards (McCauly et al., 2000).

Metals
The application of the EqP method to inorganic chemicals is hampered by the complex partitioning of metals between a number of different phases. Trace metals can be adsorbed at particle surfaces, bound to particulate sulfide, organic carbon and iron hydroxide or dissolved in interstitial waters (Jones et al., 1997). The uptake (and therefore effects) of sediment-associated contaminants is largely a function of bioavailability (Jones et al., 1997). And bioavailability is further influenced by sediment-water partitioning relationships, organism physiology (uptake rates from waters and assimilation rates from particulates) and, organism feeding and behaviour (e.g. feeding selectivity and burrow irrigation) (Simpson and Bately, 2007) (the above applies to organic constituents too).

A key partitioning phase controlling metal activity and therefore toxicity in the sediment-interstitial water system are sulfides, particularly iron monosulfide, FeS. The sulfides bind with divalent metals cadmium, copper, nickel, lead and zinc forming insoluble sulfide complexes and therefore limiting biological availability (Ankley et al., 1996). These sulfides are readily decomposed by dilute acid and thus termed acid volatile sulfides (AVS). In the laboratory, this process is used as part of a sediment toxicity assessment method. The metal concentration that is simultaneously extracted is termed the simultaneously extracted metal (SEM). When AVS concentrations exceed the sum of the SEM concentrations, no acute toxicity is observed (Ankley et al., 1996). However, the exceedance of the AVS concentrations by the SEMs does not necessarily result in mortality, for this reason, the AVS method is used only to predict when a sediment is not acutely toxic (Jones et al., 1997).

A comparison of the ability of the SEM:AVS approach and two empirical approaches (effects range-low/effects range-median and apparent effects threshold – see section 2.3.2 pg 9-11) to predict the toxicity of sediment associated trace metals by Long et al. (1998b) revealed that the empirical approaches were more accurate at predicting nontoxicity and toxicity. Furthermore, the guideline values of the empirical approaches were more sensitive than the SEM:AVS values.

Although EqP methods consider the bioavailability of the toxicant and provide causality (the effect of the single chemical is known), more research is required to fully understand the role of other binding phases in bioavailability. At present, only five metals (Cd, Cu, Pb, Ni, and Zn) can be evaluated using AVS. Further limitations include: concerns regarding the relevance of the AVS method to chronic toxicity effects (longer term or community effects) (Jones et al., 1997), reported occurrences where the SEM component has been over estimated resulting in sediments not being identified as non toxic when they should have been (Simpson et al., 1997), and the method’s inability to account for the presence or possible toxicity of metals other than the five divalent metals (Long et al., 1998b).

2.3.2 Empirical approaches for deriving sediment quality guidelines

The empirical approach generally derives guidelines using data from biological responses to contaminated sediments. These can be concentration-response data for known concentrations of single or mixed contaminants provided by spiked sediment toxicity tests (SSTTs). These tests provide highly defensible data indicating unequivocal causation between biological effect and chemical concentration. Biological response data can also be obtained from field surveys. Contaminant concentrations in a sediment sample obtained in the field are matched with biological effects associated with that sample (Long et al., 1998b). The biological responses can be community or population responses measured in the field, or a multitude of endpoints measured using bioassays in the laboratory. The contaminant concentration/s within the sediment sample are determined from dry weight whole-sediment. Consequently, the total concentration of contaminant is measured and not the bioavailable fraction (which can be affected by sediment characteristic such as grain size, and presence of organic carbon and sulfides). Thus it is possible to bias a no-effects data set downwards or upwards depending on whether data are either predominantly from sandy sediment with low contaminant concentrations (but with relatively higher bioavailable fraction), or from silty sediments where contaminant concentrations are expected to be higher (but with relatively lower bioavailable toxicity)(Batley and Maher, 2001). The more recent approach of measuring acid soluble metal concentrations instead of determining total metal concentrations is considered a better measure of bioavailability (Batley and Maher, 2001). The use of field survey data for deriving guidelines relies on the premise that relationships between sediment chemistry and biological effects will emerge during the analysis of large datasets from a wide range of geographic locations (Long et al., 1998b).

Sediment toxicity tests

Benthic organisms are exposed to reference sediments spiked in the laboratory with known amounts of single chemicals or mixtures (Jones et al., 1997). Mortality or sublethal effects are recorded, and concentration-response relationships are determined (Chapman, 1989). The guideline is derived from these concentration response relationships. A major advantage to this approach is that it follows the methods used to develop WQGs internationally; therefore, the procedure and rationale are technically acceptable and legally defensible (Chapman, 1989). Unfortunately, this method has been applied to relatively few contaminants, usually in isolation, and to a very limited number of benthic species (ANZECC, 2000). Furthermore, these data are often criticized as not being environmental realistic as they were conducted in laboratories using hardy laboratory reared organisms (ANZECC, 2000).

Interstitial water toxicity tests

It is also possible to extract interstitial water for use in standardized toxicity tests, which will generate data for guideline derivation. Toxicity identification evaluation (TIE) procedures can then be used to characterize, identify, and confirm the toxic components of a complex aqueous solution. However, TIE procedures may be difficult and costly (Maughan, 1993). Furthermore, as mentioned previously, there is currently no universally accepted method for extracting interstitial water from sediment and the extraction procedure can potentially alter the toxicity of the interstitial water (Maughan, 1993).

Screening level concentration (SLC)
This derivation method proposed by Neff et al. (1986) was specifically applied to non-polar organic chemicals (e.g. PAHs) and was one of the first attempts at deriving SQGs from an effects database. Using field data, this approach compares chemical concentrations in the sediment with presence/absence of specific benthic biota. In describing the method, Swartz (1999, pg 780) states: "A cumulative frequency distribution of stations at which a particular species was present was plotted against the organic carbon (OC)-normalised concentration of an individual PAH to derive the species screening level concentration (SSLC). The SSLC was defined as the concentration at the 90th percentile of this frequency distribution. The SSLCs for a large number of species were then plotted in another frequency distribution. The SLC was defined as the individual PAH concentration above which 95% of the SSLCs were found."

This approach has been used by the Ontario Ministry of the Environment to develop SQGs using data from a range of local sediments and benthic biota (Persaud et al., 1990). In addition to organic chemicals, guidelines were derived for selected metals and metalloids too. Disadvantages of the SLC approach is that causality is not determined; bioavailability not resolved, biological effects not measured (McCauley et al., 2000).

Apparent effects threshold (AET)
The AET method was first applied to sediments from Puget Sound, Washington, USA (Beller et al., 1986; Barrick et al., 1988). Effects data were generated by conducting sediment toxicity tests with amphipods, oyster larvae and Microtox test kit on field-collected sediments from potentially impacted and non-impacted sites. In addition, the abundance of benthic infauna was measured in the field at each of the sampled sites (Swartz, 1999). Impacted sites were identified by statistically significant adverse biological effects (these biological effects were obtained from the toxicity tests and field collected data). Using only the non-impacted sites, the lowest AET (LAET) was determined as the highest detected concentration measured in sediments from these non-impacted sites (ANZECC, 2000). Guidelines were derived for a number of metals and organics (Barrick et al., 1988). Organics were normalized to 1% organic carbon.

The advantages of this approach are that it provides non-contradictory evidence of biological effects and has an empirical foundation (ANZECC, 2000; McCauley et al., 2000). Disadvantages include: it is site specific; fails to separate effects of combined (mixtures) and single compounds (i.e. causality not determined); bioavailability not resolved; false negatives and positives will occur (ANZECC, 2000, McCauley et al., 2000) and; may err towards underprotection as the frequency of adverse effects at lower concentrations may be relatively high, but just not "always" or statistically higher (Chapman and Mann, 1999).

Sediment effect concentration
Sediment effect concentrations (SECs) were developed using data generated from specific toxicity tests of field-collected sediments (Ingersoll et al., 1996; USEPA, 1996). The toxicity test endpoints were *Hyalella azteca* 10d and 28d survival, growth and maturation, and *Chironomus riparius* 10d and 14d survival and growth. These data came from a freshwater sediment database of several surveyed areas in the USA. The derivation procedure used to produce the effects range and effects level values for individual chemicals followed method used by the National Oceanic and Atmospheric Administration and the Florida Department of Environmental Protection. Effects data only were utilised, and samples which showed toxic effects were included in the effects

database if the concentration of each toxic sample exceeded the mean of the non-toxic samples for the entire database. The SEC values for non-organics were derived on a dry weight basis and organic carbon-normalised for organics (thus bioavailability was partially addressed). Causality was not determined. SEC values were derived for total metals, SEMs, PCBs and PAHs (Batley et al., 2005).

Logistic regression modeling

The aim of logistic regression modeling (LRM) is to establish a probability of adverse effect as a function of sediment chemical concentration (Field et al., 1999, 2002). Data were obtained from a marine sediment database with matching measured sediment contaminant concentrations and effects data from sediment toxicity tests with marine amphipods. Data quality screening similar to that used by Long et al. (1995), MacDonald et al. (1996) and the USEPA (1996) was used. Only toxic sample data (effects data) were used in the regression (a sample was toxic if it was statistically different from a negative control and had less than 90% survival). Individual regression models for 37 chemicals (10 metals, 22 PAHs, total PCB and 4 pesticides) were combined into a single model to estimate probability of toxicity of a sample (Batley et al., 2005). Bioavailability and causality were not addressed.

National Oceanic and Atmospheric Administration (NOAA) approach

The NOAA annually collects and chemically analyses sediment samples from marine and estuarine locations throughout the United States. In order to interpret this chemical data, i.e. estimate a “safe” concentration (a concentration below which biological effects are not likely) and thus enable geographic indication of areas where adverse effects on benthic biota were likely (NOAA, 1999), a set of guidelines was proposed by Long and Morgan (1990). These initial guidelines were derived using a database of matching chemical concentrations and biological effects from saltwater and freshwater sediment sampling sites. The biological effects data were composed predominantly of benthic community analyses (endpoints such as abundance and indices of species richness) and bioassays of field collected samples in which mixtures of chemicals were encountered, but also included limited bioassays of reference sediments spiked with specific chemicals and data from EqP models (Long et al., 1998a). The guidelines were revised by Long et al. (1995) using an updated, higher quality database in which only saltwater data were included. SQGs were derived for nine trace elements, total PCBs, two pesticides, 13 polycylic aromatic hydrocarbons (PAHs), and three classes of PAH.

The guidelines were derived using the effects data within the database (i.e. data for which endpoints showed adverse biological effects). The effect data for each substance/chemical were ranked in order of ascending concentration and the 10th and 50th percentiles identified. The 10th percentile value was named the Effects Range-Low (ERL) and indicated a concentration below which adverse effects rarely occurred. The 50th percentile was named the Effects Range-Median (ERM), indicating concentrations above which effects frequently occur (NOAA, 1999). Long et al. (1998a) undertook a field validation of the ERL and ERM guideline values using toxicity test data (amphipod survival and a range of sublethal tests, e.g. clam, urchin and abalone embryo development – only the amphipod survival bioassay was conducted on all samples) from independent field-collected sediments (i.e. data that were not used in the derivation of the guidelines). They found that the ERL guidelines indicated 11% false negatives (toxicity observed when not expected) in the amphipod survival toxicity tests. For the ERMs , the percentage of samples that were highly toxic increased with the number of chemical specific guidelines exceeded (when one chemical’s ERM was exceeded 23%

of samples were toxic, but once 11 or more ERMs are exceeded probability of toxicity is greater than 85%). Percentage toxicity was always higher when the more sensitive sublethal toxicity tests were considered (Long et al., 1998a). A later study (Long et al., 2000), which further investigated the predictive ability of the NOAA SQGs using only acute (lethal) amphipod toxicity test data confirmed that these SQGs provided reasonably accurate estimates of chemical concentrations that were either non-toxic or toxic in laboratory bioassays. In a similar study investigating the predictability of freshwater SQGs, Ingersol et al. (1996) reported that Type I and Type II errors ranged between 5-30% for most substances tested in field collected samples from numerous studies.

The NOAA use these guidelines to identify spatial patterns of contamination, spatial scales of contamination and to rank and prioritize sites of concern and chemicals of concern (NOAA, 1999). However, there are a number of limitations associated with these guidelines: the guidelines were derived using dry weight sediments and therefore do not account for the potential effects of geochemical factors in sediments that may influence contaminant availability; they were not intended for predicting toxic effects from bioaccumulation; the data used to derive the guidelines are from soft sediments and therefore not applicable to all sediment types (although the database does represent data from a wide range of geographic locations within the US); importantly the ERLs and ERMs are not toxicological thresholds and consequently cannot be used for predicting nontoxic or toxic conditions respectively (NOAA, 1999; O'Connor, 2004) but rather identifying and prioritizing sites of concern and; causality is not determined (the effects data from field collected sediment samples is ascribed to all chemicals present within the sample) (McCauley et al., 2000).

Florida Department of Environmental Protection (FDEP) approach

The FDEP approach is similar to the NOAA approach and was first developed by MacDonald (1993). MacDonald et al. (1996) expanded and revised the database used by Long et al. (1995) to produce the Biological Effects Database for Sediments (BEDS) and applied the FDEP derivation method to it. The BEDS database allowed for the careful screening of spiked sediment bioassays and co-occurring field data matching sediment chemistry and biological effects prior to inclusion in the database. Data that were originally expressed as organic carbon-normalised were converted to dry weight assuming 1% organic carbon, which was consistent with the average organic carbon reported in other studies included in the BEDS database (Batley et al., 2005). Effects and non-effects data were ranked in order of ascending concentration. "Guidelines were calculated as the threshold effect level (TEL) and the probable effect level (PEL) for each chemical. The TEL was calculated as the square root of the product (i.e. the geometric mean) of the lower 15th percentile concentration of the effect data set and the 50th percentile concentration of the no-effect data set. The PEL was calculated as the square root of the product (i.e. the geometric mean) of the 50th percentile concentration of the effect data set and the 85th percentile concentration of the no-effect data set. The TEL represents the upper limit of the range of sediment chemical concentrations that is dominated by no-effect data entries. Within this range concentrations of sediment-associated chemicals are not considered to represent significant hazards to aquatic organisms. The PEL represents the lower limit of the range of chemical concentrations that is usually or always associated with adverse biological effects. The geometric mean is used to account for the uncertainty in the distribution of the data sets (Sokal and Rohlf, 1981)" (CCME, 1995, pg 24). Guidelines for 31 chemicals/compounds were derived.

Advantages and limitations associated with this approach are similar to the NOAA approach.

Background concentrations

This approach is seen as a simple screening method (Jones et al., 1997). The assumption is that concentrations that are not higher than background are not hazardous. Jones et al. (1997) see two major disadvantages: the approach has no biological effects basis, and it cannot be used for synthetic organic compounds, which should not be present in background sediments. In addition, measured concentrations exceeding background levels do not necessarily represent a hazard to organisms (Batley and Maher, 2001). However, background concentrations may be useful in screening other SQGs, highlighting where the guideline is within the range of background concentrations and thus should not be used to assess ecological hazard.

Bjorgesaeter and Gray (2008) question the ecological realism of empirically (field survey data and toxicity tests) and theoretically (EqP) derived SQGs because of the uncertainties of extrapolating data from the laboratory to the field, the impossibility of accounting for the sensitivities of all sediment dwelling organisms, and large variability of physical sediment characteristics over small geographical areas affecting the bioavailability of chemicals. They hypothesized that setting a SQG of 4-times background concentration would give sufficient protection to sediment fauna from metal contamination. In order to test this, they used field data (biological presence/absence and associated metal concentrations) collected at different sediment types and depths on the Norwegian Continental Shelf to determine threshold levels eliciting effects (apparent effects threshold approach) and compared these against background levels from reference sites. They found large variations in naturally occurring metal concentrations and in the threshold levels eliciting effects on the fauna at different sediment types and depths suggesting SQGs should not be universally applied. Their results showed that the overall metal concentrations eliciting effects on biota were 3.6-times higher than background levels, and were much more conservative than current international SQGs, suggesting current SQGs are not adequately protecting marine sediment fauna.

2.3.3 Consensus approach for deriving sediment quality guidelines

The consensus approach was developed by Swartz (1999) in order to provide a unifying synthesis of existing SQGs, to reflect causal rather than correlative effects, and to account for effects of contaminant mixtures. Swartz (1999) focused on polycyclic aromatic hydrocarbons (PAHs) in marine sediments in response to the issue of PAHs occurring in the field as a complex mixture of covarying compounds. The consensus approach addresses what Swartz terms the “mixture paradox”: “a SQG derived from accurate, experimental determination of toxicological effects caused by an individual PAH compound (e.g. through spiked sediment experiments) will greatly underestimate ecological effects in the field that are associated with the SQG, but actually caused by the PAH mixture. As a corollary, an SQG derived from the correlation of ecological effects with the concentration of an individual PAH in field-collected sediment will greatly overestimate the effects actually caused by the single compound.” (Swartz, 1999, pg 782).

The method involved collating previously published SQGs for total PAH compounds (TPAH). These SQG values were categorized into three groups according to their original narrative intent (i.e. likely or not likely to indicate hazardous biological effects).

Within each of the groups the SQG value agreed within a factor of four (when TEL and PEL were included – probably because they are derived from a combination of effects and no-effects data) and within in a factor of two when TEL and PEL were removed. The consensus SQG for TPAH within in each category was derived as the arithmetic mean of various SQGs to produce the threshold effects concentration (TEC), median effects concentration (MEC) and extreme effects concentration (EEC). TPAH concentrations below the TEC are unlikely to cause adverse effects on benthic ecosystems, while the EEC indicates virtual certainty of adverse effects. The range between the TEC and EEC represents a gradient along which effects are increasingly more probable (Swartz, 1999).

Swartz (1999) justifies the validity of the consensus approach by pointing out that the similar grouping of the concentrations derived by the various guidelines into threshold, median and extreme effects is unlikely to be coincidental. Furthermore, field validation of the consensus guidelines showed they successfully predicted sediment toxicity and benthic community perturbations at sites of PAH contamination.

The consensus method is not simply collating available SQGs and calculating an average value. The consensus stems from the idea that if different methods for deriving SQGs result in quantitatively similar concentrations, then the validity of the result is greatly enhanced. Only then is the calculation of a consensus guideline justified (Batley et al., 2005).

MacDonald and co-workers applied the consensus-based approach to total polychlorinated biphenyls in marine sediments (2000a) and to 28 chemical compounds for freshwater sediments (2000b). In the freshwater sediment study a TEC and probable effect concentration (PEC) were derived as a geometric means of published SQGs from each group/category of narrative intent (the PEC is analogous to the MEC)(MacDonald et al., 2000b). The TEC and PEC were evaluated, revealing most of the TECs (21 of 28) provided an accurate basis for predicting absence of sediment toxicity, and similarly, most PECs (16 of 28) provided an accurate basis for predicting sediment toxicity. MacDonald et al. (2000b) applied the derived guidelines in a quotient-based framework to get around the problem of the average consensus value obscuring conceptual differences among guidelines (Batley et al., 2005).

The advantages of the consensus approach are that it addresses causality, bio-availability and mixture effects, and is based on the strengths of other methods (McCauley et al., 2000). McCauley et al. (2000) feel the disadvantages of this approach are that the methods are not standardized and the theory not well understood.

TABLE 2.1 Summary of sediment quality guideline derivation methods

SQG	Citation	Data source	Test endpoints	Route of exposure	Test duration	Lab vs. field	Bioavailability considered	Causality	Mixtures
NOAA approach (ERL and ERM)	Long & Morgan, 1990; Long et al., 1995	Matching chemistry and biological effects data, mainly from toxicity tests of field collected sediments, some benthic survey and EqP data too	Predominantly lethality, wide range of test organisms	Whole sediment and pore water	Variable	Mixed	No	Not addressed	Considered implicitly, not explicitly
FDEP approach (TEL & PEL)	MacDonald et al., 1996	As for NOAA approach	Predominantly lethality, wide range of test organisms	Whole sediment and pore water	Variable	Mixed	No	Not addressed	Considered implicitly, not explicitly
SLC	Neff et al., 1986	Matching chemistry and presence / absence of benthic biota	Benthic organism occurrence	In situ	Long-term	Field	Partly addressed, normalised to organic carbon	Not addressed	Considered implicitly, not explicitly
AET	Barrick et al., 1988	Matching chemistry and biological effects data, mainly from toxicity tests of field collected sediments, some benthic surveys	Predominantly lethality, some sub-lethal (growth, abnormality), benthic community endpoints, Microtox	Varied with endpoint, whole sediment, elutriate, in situ	Variable	Mixed	Partly addressed, normalised to organic carbon	Attempts to address causality, success uncertain	Considered implicitly, not explicitly
SEC	Ingersoll et al., 1996	Matching chemistry and biological effects data from toxicity tests of field collected sediments	Lethality, growth and reproduction	Whole sediment	Variable	Lab	No	Not addressed	Considered implicitly, not explicitly
LRM	Field et al., 1999,2002	Matching chemistry and amphipod toxicity response data of field collected sediments	Amphipod mortality	Whole sediment	10d	Lab	No	Not addressed	Considered implicitly, not explicitly
EqP	Di Toro et al., 1991; Van der Kooij et al., 1991, Ankley et al., 1996	Acute and chronic toxicity data from water column tests	Acute and chronic toxicity from wide range of test organisms	Ingestion and pore water	Variable	Lab	Organics: normalised to organic carbon; metals: AVS:SEM ratio used to address metal bioavailability	Yes	Not considered except for guidelines developed explicitly for PAH mixtures
Consensus	Swart, 1999	SQG values derived by other methods	Combines guidelines with range of endpoints	Varied	Variable	Mixed	Partly addressed, normalised to organic carbon	Combines guidelines with range of causal connection	Combines guidelines with range of mixture consideration

2.4 Methods adopted by various countries

USA

The various empirical methods discussed in section 2.3.2 were developed and used within the USA. However, no national SQGs have been adopted, although the US EPA has decided on pursuing the EqP approach (USEPA, 2003a; 2003b; 2003c; 2003d; 2004a; 2004b).

Australia and New Zealand

The ANZECC & ARMCANZ unveiled risk-based SQGs in 2000. The guidelines were primarily based on the NOAA effects database only (updated by Long et al., 1995), although if a more appropriate or realistic guideline for a particular type of chemical was identified, it was adopted instead. It was felt that the inclusion of no-effects data may include a more serious bias to the guidelines through the addition of abnormally low values from coarser grain size sediments (ANZECC & ARMCANZ, 2000). However, it was acknowledged that while the use of effects only data was acceptable for the setting of interim guidelines, this approach was limited and more accurate guidelines would require more realistic, better quality data.

The interim SQGs for most metals were derived using the NOAA effects-only database. Exceptions are Hg (the ERM value was adopted from the Hong Kong guidelines), As (the NOAA value was below background values in Australia and thus the value determined for ocean disposal (ANZECC, 1998) was used) and Cu where the ERM value was adopted from the Hong Kong guidelines). The organics guidelines were derived using varied approaches: interstitial water toxicity testing (e.g. for ammonia, nitrate and nitrite); EqP approach (tributyltin), BEDS (lindane) and NOAA effects-only database (most remaining organics).

The updated NOAA database (Long et al., 1995) incorporated only saltwater data, but is applied to Australian and New Zealand freshwaters. It has been shown, however, that the freshwater guidelines developed in Canada show good agreement ($r = 0.98$) with marine-based guidelines (Environment Canada, 1995; Smith et al., 1996). Nevertheless, in the future, separate databases for fresh and saline waters are desired (ANZECC and ARMCANZ, 2000).

Canada

Canada utilised data from the BEDS (MacDonald et al., 1996) (combined with some EqP data and spiked sediment toxicity tests results), to produce separate fresh and saline water databases. The derivation procedure was adopted from the FDEP approach to produce a TEL which, after the application of a safety factor, was used as an interim SQG (CCME, 1995). Separate interim SQGs were developed for fresh and saline waters.

Hong Kong

For most contaminants, NOAA data and derivation procedures were used to derive an interim sediment quality value (ISQV) (Chapman et al., 1999). Two sets of ISQVs were determined: ISQV-low (equivalent to ERL) and ISQV-high (equivalent to ERM). There were some exceptions, for the ISQV-low, all metals except Hg and As used previous Hong Kong criteria (as no effects had been observed to date using these criteria). For ISQV-high, Hg and Ni used previous Hong Kong criteria, while remaining metals were derived using NOAA data.

Europe

Before the legislation of the Water Framework Directive for the European Union, individual countries had to lesser and greater extents developed their own SQGs. The United Kingdom, Italy and Germany did not have common national regulations for management of contaminated sediments, but some individual agencies within those countries were using some form of guideline (Babut et al., 2005). In France, a preliminary set of criteria was completed in 1999. The method utilised was the FDEP approach to derive TELs and PELs using BEDS for trace elements and a few organic compounds (DDE, dieldrin, fluoranthene, lindane and polychlorinated biphenyls). Equilibrium partitioning was utilised for the majority of organic substances (Babut et al., 2005). And in the Netherlands, the national SQGs for almost all constituents were derived using EqP theory.

More recently, there was an attempt under the Water Framework Directive to ensure all member states of the European Union implement mandatory sediment environmental quality standards (EQSs). Proposals were made on how to develop these sediment EQSs. However, the European Commission was subsequently advised (Bonnomet and Alvarez, 2006 cited in Crane and Babut, 2007) that the complexities and uncertainties in setting and monitoring legally binding sediment environmental quality standards were too great to be reliable. Consequently, in the Daughter Directive released in 2006 (EC, 2006) which proposed EQSs for water, sediment EQSs were not included. The debate within Europe continues regarding the possibility of deriving legally defensible, environmentally relevant sediment EQSs, or using SQGs at a lower tier risk assessment framework within the Water Framework Directive (Crane and Babut, 2007).

South Africa

The Benguela Current Large Marine Ecosystem Programme developed sediment quality guidelines for marine coastal zone associated with the Benguela current. Effect and no-effect data utilized were from BEDS (FDEP) and the derivation method used was adapted from the Canadian approach (CSIR, 2006) (Figure 2.1). The Department of Environmental Affairs and Tourism have developed guidelines in order to comply with the Convention of Marine Pollution by Dumping of Wastes and Other Matter (London Convention). The guidelines have no regulatory status. Newman and Watling (2007) discovered that documentation of the derivation methods used in the DEAT guidelines is limited and thus the actual derivation method used is difficult to ascertain, but that it appears a “middle of the road” approach based on several other countries’ approaches. There are no freshwater sediment guidelines.

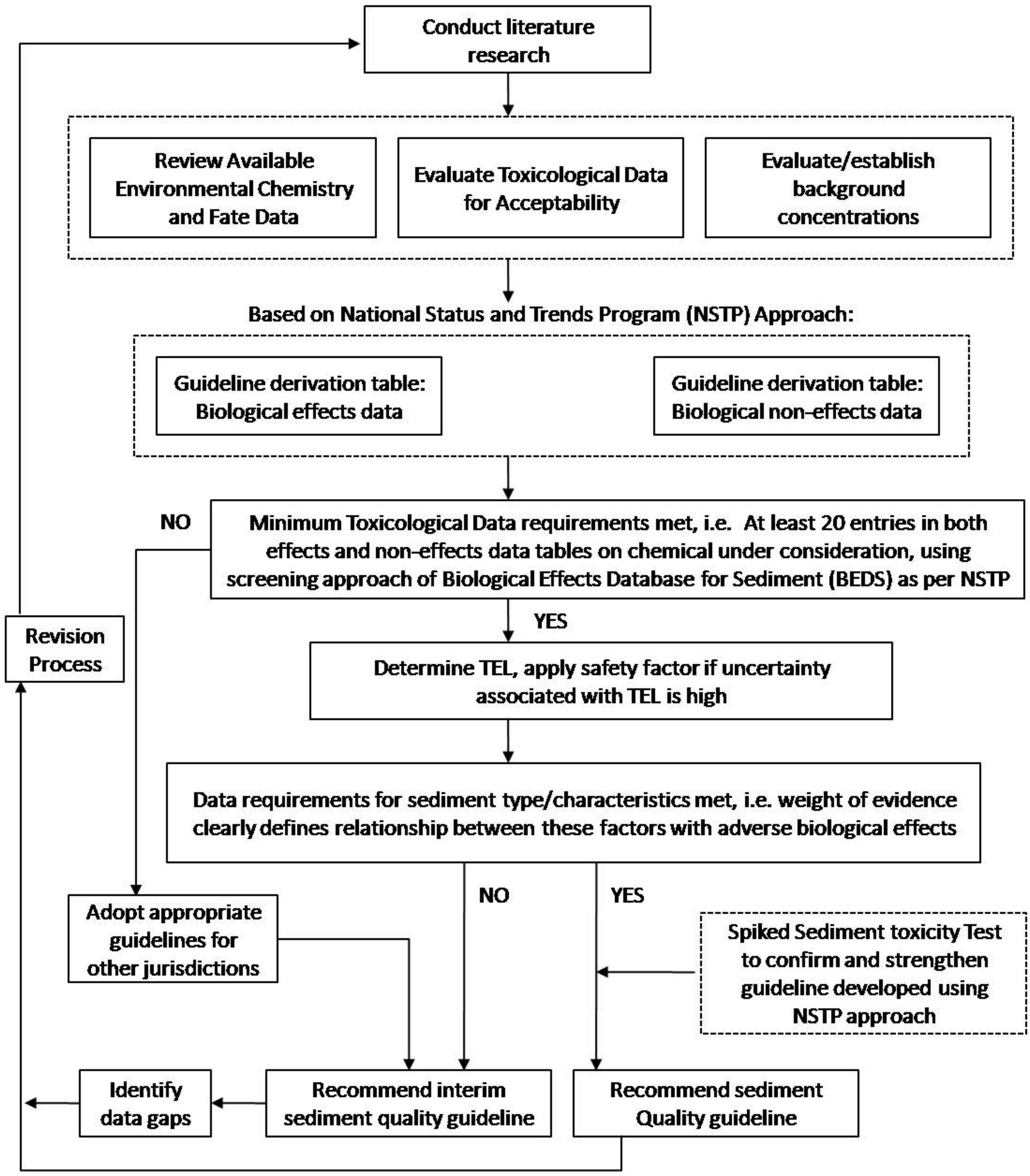

FIGURE 2.1 Derivation protocol used for BCLMEP (CSIR, 2006) (adopted from Canadian approach (CCME, 1995)

CHAPTER 3: SEDIMENT QUALITY GUIDELINE DERIVATION AND APPLICATION IN THE SOUTH AFRICAN CONTEXT

3.1 Introduction

A workshop of aquatic and sediment scientists was held to identify the issues involved in deriving and implementing SQGs in South Africa (Appendix A). It was recognized that the process of deriving and implementing SQGs, so as to complement current aquatic resource management, will be complex and require considerable effort. Consequently, this chapter details the issues raised and current thinking of South African aquatic specialists, and further investigation, consultation and decisions regarding these issues are needed.

3.2 Sediment quality guideline derivation and application: Issues raised

The need for SQGs for South African freshwaters
Freshwaters are managed to protect aquatic ecosystem functionality while allowing sustainable development. Management involves setting objectives for the various components of the water resource (biological organisms, water chemistry, geomorphology, etc.) that will ensure a particular level of aquatic ecosystem function or ecological health. Sediment is an important component that can affect aquatic ecosystem functioning. Sediments are a sink for many potentially harmful anthropogenic contaminants, which may be released and become toxic when conditions within the resource change (e.g. changes in pH, redox potential, perturbations, presence/absence of organic carbon and sulfides). Furthermore, sediment may have a negative physical effect on biota and/or alter their habitat. Thus, there is a real need to understand the effect of sediment and sediment associated contaminants on aquatic organisms in order to effectively manage aquatic ecosystem function. There are four options for assessing sediment quality:

1. Decision making based only on background levels of the contaminant
2. Decision making based only on biological testing
3. Decision making based only on SQGs
4. Decision making based on a combination of the above 3

Chapman et al. (1999) feel the best option is no. 4 as each of the first 3 options are useful and important, but individually have some serious drawbacks. Thus a combined approach provides the most reasonable information for decision making. Most jurisdictions now follow this approach.

Workshop attendees agreed that there was a need for SQGs to be developed for implementation in South Africa. The ultimate goal is the development of inclusive chemical-specific standards for the aquatic resource as a whole (which would incorporate water, sediment and riparian habitat compartments). Unfortunately, this is not feasible at present and thus separate SQGs should be developed.

The application of the SQGs, from a regulatory point of view, is seen in two contexts:
a) assessments (i.e. what can you say about how good or bad the situation is from a set of field collected data) – applicable to monitoring programmes and risk assessments; and b) setting sediment quality objectives (defining what is an acceptable risk to the aquatic environment and determining the sediment quality associated with that risk) – applicable to ecological Reserve determinations and risk assessments.

What will be the aim of the SQGs?
This will need to be explicitly stated. It was discussed at the workshop, but not resolved and will require further deliberation of the following: Protect what, when, for how long? The aim of the 1996 WQGs for aquatic ecosystems (DWAF, 1996) was to achieve protection of aquatic ecosystems from adverse effects (Roux et al., 1999). A possible aim for the SQGs could be to protect all forms of aquatic life during all forms of aquatic life cycles from adverse effects for an indefinite period of exposure to substances associated with the sediment. The 'adverse effects' would have to be defined, and would need to be linked to the various resource classifications; such is vital for the determination and implementation of the ecological Reserve. However, the data required to derive an environmentally realistic and sustainable development-friendly guideline based on local derived data and knowledge are insufficient at present. The predominant data available are chemical concentration / biological response data from field collected sediments. Unfortunately, guidelines derived using these types of data cannot confidently predict a specific level of toxicity for a specific chemical in the sediment of a site. Rather, these types of guidelines identify sites of potential contaminant concern that require further investigation.

What will be the philosophical approach?
It is envisaged that the philosophical approach to be adopted for deriving and implementing the SQGs will be based on the approach being developed for the revision of the water quality guidelines (WQGs) in South Africa (DWAF, 2008), which is currently underway. SQGs developed by other jurisdictions generally consist of a list of chemical specific concentrations. These are not site (or scenario) specific and tend to be misused as those applying them are sometimes unaware of the underlying assumptions made in deriving the values. By adopting an explicit probabilistic risk assessment approach (where all available and/or applicable hazard and exposure information is taken into consideration) the person applying the SQG (e.g. water resource manager) is able to make an informed decision as they understand what is at risk, what data has been used or assumptions made in determining that risk, and what the resultant confidence in the SQG value should be.

Aligning SQGs with all resource management tools
This will need further attention from a focus group and is also an issue for current revision of the WQGs. However, it should be noted that in order to derive suitable risk assessment endpoints there will need to be a further interpretation of the current classification system which does not clearly delineate what that endpoint is/should be. This is related to what sort of endpoint (e.g. organismal lethality, sub-organism effects, population level effects, etc.) and the extent of those effects (e.g. 1%, 5% or 10% of an SSD) are used as risk assessment endpoints.

How should the SQG be structured?
It is envisaged that the guideline structure will be similar to that being developed for the revised WQGs. The proposed WQGs structure is succinctly conveyed in DWAF (2008). Some of ideas presented in that document have been included in this document (extracts are presented in a different font):

Many people's idea of SQGs are of a table of numbers that define concentration ranges within which specified biological effects can be expected for specific chemicals. The approach envisaged for both the sediment and the revised WQGs requires the guidelines to be much more than this:

"The guidelines" include all of the following:

- The numerical values that define ranges within which specified effects, or degrees of effect, can be expected;
- The narrative description of those effects (e.g. lethality in the test organism population; changes in community structure);
- The description of the sampling, sample preparation, and analytical procedures upon which the numerical values are based;
- The general information on the constituents (like chemical, physical, and toxicological properties) – the so-called "hazard description";
- The general information on such issues as occurrence in the environment and general behaviour (like typical fate and transport); and
- Mitigation (treatment) options that provide guidance on what can be done about specific problems.

"The guidelines" therefore include all the information that may be useful to the reader, particularly in supporting informed decision making relating to water resource management.

The primary tool facilitating the determination and use of the guidelines should be a software decision support system (DSS). This should be complemented with a set of hard copy manuals that at least present generic values and supporting information (DWAF 2008, pg 42). A more detailed description of the DSS is presented in DWAF (2008).

Details regarding the specific type of data to be used in the derivation process (e.g. spiked sediment toxicity test data or field collected cause-effect data), the source of the data (international or local or combination) and the required quality of the data were not discussed in detail at the workshop. These details will need to be the topic of further investigation by a focus group in subsequent phases of the project.

The sediment quality guidelines are envisaged to comprise of a three tier system:

Tier 1. Provides 'generic' guidelines values that are made available in the DSS and hard copy manuals. These guideline values will be conservative as the worst case scenario is assumed.

Tier 2. Allows for site specificity in specified contexts and is facilitated by the DSS, consequently there is more confidence in the derived value

Tier 3. Full risk assessment. Not facilitated by the DSS but will use information contained within the DSS information database.

As a consequence of the scenario specific approach inherent in the proposed SQG structure at the Tier 2 and 3 level, the method for deriving the SQG is similar to the method for undertaking an assessment of sediment toxicity and applying site specific factors of a particular site or given a particular scenario.

The procedure for generating the guideline values will follow a probabilistic risk assessment process, starting at Tier 3 and progressing to Tier 1 (the exact procedural methods are still to be resolved). Associated with the move down the tiers is a loss in confidence in the resultant guideline value and thus the need to be more conservative (through use of conservative data, perhaps inclusion of safety factors and/or reducing what is considered acceptable risk to the resource). A description of the possible derivation procedure is as follows:

- At Tier 3 all exposure parameters that would affect the toxicity of the chemical to the organism are identified (these parameters were discussed by workshop attendees and a tentative list produced – see Table 3.1). In addition to the exposure parameters, hazard characterisation (identifying and/or generating relevant toxicity data) has to be undertaken (workshop attendees discussed the various options available for measuring the effect of toxicants on organisms – see Table 3.3). The data required for the exposure parameters and hazard characterization must be generated if not available.
- If this is not desirable or possible, then a Tier 2 assessment is undertaken instead. At this level, there are mandatory exposure parameters (for which data are required) and preferable exposure parameters (if no data are available, assumptions can be made). Ideas on what these should perhaps be were proposed at the workshop (Table 3.2).
- If there are not sufficient data to meet the requirements of a Tier 2 assessment, and no option to collect or generate data from the field or in the laboratory then a Tier 1 assessment is undertaken. At this level, assumptions are made for all parameters and consequently the guideline value will be conservative due to the application of safety factors and/or conservative methodology. Thus, although there may be low confidence in the Tier 1 guideline value as a result of the assumptions made in its derivation, at least the person applying the guideline is aware of the assumptions and the level of confidence as the 'record of decision' detailing the derivation process and data utilized will be explicit.

Although on most occasions managers applying the SQGs will probably use the Tier 1 values it is critical to have the option to derive a guideline value that represents greater confidence and scenario specificity at Tiers 2 and 3. In addition the tiered system shows that the thinking has been done at the higher tiers and the implications of using the value at the lower tier are explicit.

The output of the SQG derivation process will be different depending on the application:
For assessments: each sediment quality variable for which concentrations are specified by the user should have a quantitative estimate of risk (e.g. expressed as a percentage) and a corresponding narrative description of the fitness (e.g. excellent, good, fair, unacceptable) associated with the guideline value.

For sediment quality objective setting: each sediment quality variable of interest specified by the user should have a corresponding generic value (for comparison purposes) and a site-specific value.

TABLE 3.1 Potential exposure assessment parameters for a Tier 3 SQG derivation

Parameter	Comment
Organic carbon content	Provides binding site for contaminants
AVS/SEM ratio (acid volatile sulfide / sequentially extracted metal)	Method for assessing bioavailability of metals
Sediment oxygen demand	Implications for bioavailability of contaminants
Particle size distribution	Water flow will affect particle size distribution, dictating habitat availability, which in turn dictates organisms found at a site. As different organisms have different uptake routes, susceptibility to a chemical stressor will vary. In addition, surface area affects the available binding sites for contaminants
pH	Affects desorption and release to interstitial water of sediment bound metals
Alkalinity / carbonate	Affects bioavailability of sediment bound contaminants
Chloride concentration	
Hardness / calcium and magnesium	Affects bioavailability of sediment bound contaminants
Redox potential	Affects desorption and release to interstitial water of sediment bound metals
Ammonia (NH3)	Although a constituent it will affect other constituents
Total dissolved solids (TDS)	
Flow	Will affect bioavailability and spatial distribution of contaminants either directly or indirectly
Anthropogenic nanoparticles	Impossible to quantify or qualify as a hazard at the moment, but is included as an exposure parameter
Historical profile of the catchment	Provides clues on possible contaminant exposure and geochemical characteristics of sediment. Sedimentation transport and settling rates will potentially affect bioavailability of contaminants
Bioaccumulation	Toxicity may be attributable to bioaccumulated toxin concentration instead of toxin's external environmental concentration
Routes of exposure / sediment ingestion rate	Affects bioavailability of sediment bound contaminants
Background concentrations	Deriving a guideline value using 3.6 X background conc has been shown to be protective (Bjorgesaeter and Gray, 2008)

TABLE 3.2 Potential exposure assessment parameters required for Tier 2 SQG derivation

Mandatory	Preferable (can make assumptions)
pH	Organic carbon
Flow	Redox potential
Particle size distribution	Alkalinity / carbonate
AVS/SEM (for metals)	Bioaccumulation
	Hardness / calcium and magnesium
	Background concentrations

Ideally, how should the SQGs be developed?
In order to derive the ideal SQG one would require the right data and derivation method.

Data:
Need cause and effect data (high quality spiked sediment toxicity test (SSTT)data that takes into account the antagonistic and synergistic effects of contaminant mixtures, and issues of bioavailability under different sediment conditions) from wide variety of organisms (including ecologically sensitive and important species). However, in reality, the ideal type of data do not exist in sufficient quantities to allow for derivation, not only just for South African species, but world wide too.

So what types of data do exist? The majority of data available for SQG derivation are correlative, i.e. cause and effect between a specific chemical concentration and a biological effect is not proved, but rather a correlation observed. These data are generated from simultaneously field-collected chemical and biological data. The biological data can be a community response measured in the field, or a specific toxicological endpoint measured in the lab using the field collected sediment sample. The biological response is ascribed to the concentrations of chemicals measured in the sediment sample. Advantages of these data are that: they account for antagonistic and synergistic toxic effects of chemicals in mixtures; and there are many such data available (based on North American field studies). Disadvantages are: causation not proved (no proof that biological effect observed was caused by chemical concentration measured); do not sufficiently account for bioavailability (the concentration measured from the field sample is on a dry weight basis and factors such as particle size, pH of the overlying water and presence of sulfides and organic carbon are not always measured or taken account of), and data are specific to the site where the sample was taken (due to factors affecting bioavailability changing spatially).

Can SSTT data be generated? There are standard methods available and development and refinement of South African specific tests should encouraged. However, to generate the number of data points required for all contaminants of concern would be a great challenge and expense and take a very long time. Many jurisdictions have adopted interim SQGs using what correlative data are available and made provision for updating guidelines values when SSTT data become available.

The ideal derivation method:
Needs to define, with a high degree of confidence, thresholds of ecosystem effect/health for various concentrations (e.g., on a basic level: effect and no-effect concentrations. Or more intricately: concentrations associated with percentage likelihood of effect – 90%, 80%, etc.) to allow for application within a management framework for resource protection (e.g. the South African resource classification system). Linear regression models using SSTT data in the form of a species sensitivity distribution (SSD) meet the requirement of an ideal derivation method.

However, the derivation method chosen depends on the toxicity data available/used: As correlative field based concentrations often have considerable overlap of effect and no-effect data associated with them, the derivation approach used does not attempt to define specific effect concentration, but rather ranges of concentrations where certain biological effects may be expected.

The equilibrium partitioning (EqP) method can also be used to generate SQGs. The advantages of this method are attractive: the bioavailable fraction of the chemical in the bulk sediment sample is considered and thus is applicable across sediment types (as long as these sediments possess >0.2% organic carbon); the guidelines are causally linked to a specific chemical; the SQGs are linked to the WQGs and thus level of protection is specified and is thus is attractive to regulators and; the approach is based on fundamental toxicological principles). However, there are some considerable disadvantages: this method does not protect against synergistic or antagonistic effects of other compounds or protect against bio-accumulative effects; critics argue that not enough is known about the role of other binding phases during the partitioning process from sediment to interstitial water; at present, only five metals (Cd, Cu, Pb, Ni, and Zn) can be evaluated using the AVS EqP method (SQGs for organics using the EqP methods are more successful) and; the method assumes sensitivities of benthic organisms to contaminants are similar to that of organisms living in the overlying water.

Consequently, this topic needs considerable further investigation by a focus group in subsequent phases of the project. However, it was noted that using a probabilistic approach allows the use of a weight of evidence approach utilizing all derivations methods.

Type and quality of toxicity data to be included in DSS database?
This will need to be fully investigated by a specialized focus group. The issue of data quality control is important. It was emphasized that a preoccupation with local/indigenous data at the expense of international data should be avoided. International data should be used if it is applicable to SA conditions (e.g. particle size mineralogy, organic carbon). If applicability of data is questionable, it could still be used as a reference point to guide the derivation process.

It is envisaged that there will be a database of suitable quality toxicity data associated with the DSS. Issues to be dealt with by the focus group will be: establishing rules for determining the suitability of toxicity data for inclusion in the database and establishing a protocol for including new data when it becomes available. The type of toxicity data to be included in the data base (e.g. matching chemical concentrations and biological effects from field collected samples or laboratory SSTT data) will need to be decided on in further phases of this project.

In terms of utilising laboratory spiked sediment toxicity test data, delegates discussed the possible application of SSDs for the purpose of guideline derivation. Concerns were raised regarding the undesirable flexibility in the tails of the Burr distribution used by the Australians in their water quality guideline derivations. The addition of data points in a Burr distribution is said to alter (to an unacceptable level) the lower end of the distribution, e.g. the 5^{th} percentile which is often used in guideline derivation. A possible alternative for investigation is the Pareto distribution, which is not as flexible in the tails, but consequently does not always fit the data as well as the Burr distribution.

Another issue for further investigation is the use of Bayesian statistics when SSDs are updated with new information.

A proposal for the use of SSDs and SSTT data in the tiered SQG guideline approach was discussed:

At Tier 3: the full risk assessment will generate hazard data which will feed into the SSD and update the hazard data obtained from the DSS database.
At Tier 2: The hazard data from the DSS database are used as is and the required exposure parameters are noted. The output is a scenario specific concentration/guideline with probability estimates (i.e. some measure of confidence). Perhaps some form of safety factor could be applied.
At Tier 1: Only the hazard data from the DSS database are use in an SSD. A more conservative approach is adopted and safety factors adopted.

What are the most appropriate methods for hazard characterisation?
There are few whole-sediment toxicity tests developed for freshwater benthic organisms (Table 3.3). Standard methods have been developed for *Hyalella azteca* and *Chironomus riparius*, *Chironomus tentans*, *Tubifex tubifex* and *Hexagenia* sp. in North America and Europe (Table 3.3). There are a number of toxicity tests that were originally developed for aqueous samples but have been adapted for testing sediment extracts (pore water/interstitial water, elutriates). Examples of such tests that have been standardised at ISO level or national level include crustacean *Daphnia magna* (ISO, 1996; ISO, 2000a; Environment Canada, 2000); crustacean *Ceriodaphnia dubia* (Environment Canada, 1992; AFNOR, 2000a); rotifer *Brachionus caliciflorus* (AFNOR, 2000b); algaes *Pseudokirchnerilla subcapitata*, *Desmodesmus subspicatus* and *Selenastrum capricornutum* (ISO, 2004); duckweed *Lemna minor* (ISO, 2005a); bacteria *Vibrio fischeri* (ISO, 2007a); bacteria *Pseudomonas putida* (ISO, 1995); genotoxicity using umu-test (ISO, 2000b), AMES-test (ISO, 2005b), amphibian larvae (ISO, 2006) and cell line (ISO, 2008); fish eggs (ISO, 2007b).

TABLE 3.3 Benthic organism-based sediment toxicity tests

Organism	Test duration	Endpoint	Standardised	References
Hyalella azteca (Amphipod)	28 d	Survival and growth	AFNOR (2003); ASTM (1997, 2002); USEPA (2000a); Environment Canada (1997a)	Schubauer-Berigan et al., 1993; Call et al., 2001a, 2001b
Chironomus riparius (Diptera)	7 or 10 d, 28 d	Survival and growth Emergence	AFNOR (2004); ASTM (1997, 2002); Environment Canada (1997b);	Bettinetti and Provini (2002); Girling et al., (2000); De Haas et al., (2004);
Chironomus tentans (Diptera)	10 d 65 d	Survival Partial life cycle	ASTM (1997, 2002); USEPA (2000a); Environment Canada (1997b)	Call et al., 2001a, 2001b
Tubifex tubifex (Oligochaete)	28 d	Survival and reproduction		Reynoldson et al. (1991); Bettinetti and Provini (2002); Pateris et al., (2003)
Hexagenia sp. (mayfly nymph)	10 d 21 d	Survival Partial life cycle	ASTM (2002);	Prater and Anderson 1977; Giesy et al., 1990

Adapted from Ireland and Ho (2005); Hansen et al., 2007

Toxicity tests specific to South African conditions should be developed or adapted from international tests and validated. In particular, sublethal endpoints should be investigated at lower levels of biological organisation (e.g. chironomus mouth part deformation, various sub-organismal biomarkers). Currently, Resource Quality Services of the DWAF are culturing a *Chironomus* species (yet to be identified to species level) and are regularly undertaking toxicity tests using field collected sediments. In addition, The Council for Geosciences intends on developing sediment toxicity testing capacity within its laboratories.

Are there local sediment toxicity data available and how to include them?

There appears to be very little local sediment toxicity data available. The University of Johannesburg has recently investigated the application of the sediment quality triad in order to determine the site specific hazard of freshwater and marine sediments. Toxicity tests conducted on field collected freshwater sediments utilised indigenous *Chironomus* sp. and *Daphnia* as bioindicators (Wepener pers comm.). The bioindicator used for the marine sediments was *Grandidierella lignorum*. The above toxicity test methods are detailed in a PhD manuscript currently being completed. As part of the same investigation, an MSc study investigated methods of determining the bioavailability of metal in the field collected sediment. This study is also currently being completed.

Regarding sediment chemistry assessments in South Africa, a number of studies have investigated metal concentrations in sediments of local rivers and dams. Fatoki and Awofolu (2003) determined Cd and Zn concentrations in sediment from the Buffalo, Keiskamma, Tyume and Mthatha rivers and in the Sandile and Mthatha dams. Awofolu et al. (2005) determined Cd, Pb, Co, Zn, Cu and Ni sediment concentrations in the Tyume River. Awofolu and Fatoki (2003) also investigated sediment concentrations of organochlorine pesticides DDT, chlordane, hexachlorobenzene, heptachlor and endosulfan in the Buffalo, Keiskamma, Tyume and Swartkops rivers and in Sandile Dam. In the Western Cape, Berg River sediments have been analysed for Al and Fe by Jackson et al. (2007) and for Al, Fe, Mn, Zn, Cu, Ni and Pb in the Plankenburg and Diep rivers by Jackson et al. (2009). Thawley et al. (2004) investigated Zn and Cd concentrations on the sediments of three tributaries of the Limpopo River. Sequential metal extraction techniques have been used to determine the concentrations of 12 heavy metals in sediment from the Vaal Dam System (Gouws and Coetzee, 1997) and from the Nyl River flood plain (Greenfield et al., 2007). Wepener et al. (2000) applied EqP method in order to derive site specific copper and zinc quality criteria for the Olifants and Selati rivers.

In the case of marine sediments, Fatoki and Mathabatha (2001) determined the distribution of Zn, Cd, Cu, Fe, Mn and Pb in sediments from the East London and Port Elizabeth harbours, while Wepener and Vermeulen (2005) investigated Al, Cr, Cu, Fe, Mn and Zn sediment concentrations in the Richards Bay Harbour. Newman and Watling (2007) generated baseline concentrations for selected metals found in marine sediments along the south-eastern Cape coast, and compared these to guidelines derived by the Department of Environmental Affairs and Tourism (DEAT)(undated). According to Newman and Watling (2007), the DEAT guidelines were developed in order to ensure that South Africa was compliant with the Convention of Marine Pollution by Dumping of Wastes and Other Matter (London Convention). The guidelines have no regulatory status though. Newman and Watling (2007) further discovered that documentation of the derivation methods used in the DEAT guidelines is limited and thus the actual derivation method used is difficult to ascertain, but that it appears a "middle of the road" approach based on guidelines from several other countries. Their assessment of the DEAT guidelines using baseline metal concentrations found that the lower guideline value (Special Care Level) was inappropriate for chromium and possibly nickel at the sites assessed along the south-eastern Cape coast. The authors concluded that locally relevant guidelines needed to be established, and might need to be regionally focused as background concentration levels of metal vary regionally. Their observations are relevant to the derivation of freshwater sediment guidelines for South Africa.

Are guidelines based on equilibrium partitioning (EqP) methods acceptable in a regulatory framework?
The many uncertainties associated with EqP were noted. However, the consensus from DWAF representatives at the workshop was that if the scientists were confident of results using EqP then there would be no objection from the regulators.

Should SQGs for freshwater and saltwater sediments be derived from separate datasets?
Further discussion is needed. It was noted that crustaceans were more sensitive than other invertebrates in both fresh and saltwater, and thus the origin of the organism might be less important than the type of organism. Perhaps representative taxonomic groups need to be specified for inclusion in the hazard assessment. A further point raised suggested that organics may have higher bioavailability in saltwater.

The database used to derive the Australian and New Zealand sediment quality guidelines incorporated only saltwater data, but has been applied to freshwaters too. Justification is based on studies which show that the freshwater guidelines developed in Canada show good agreement ($r = 0.98$) with marine-based guidelines (Environment Canada 1995, Smith et al., 1996). Nevertheless, it has been expressed by Australia and New Zealand authorities that in the future separate databases for fresh and saline waters are desired (ANZACC and ARMCANZ, 2000).

There are international databases available that have separate freshwater and saltwater data (e.g. NOAA sediment toxicity database – SEDTOX, and Environment Canada).

Possible sediment quality guidelines implementation approaches
The possible implementation approach for SQGs in South Africa will be dependant on the approach adopted for the revised WQGs. An example of an implementation approach for guidelines derived using matching biological effects and contaminant concentrations in field collected sediments is presented in Table 3.4. The guidelines are “benchmark” values which define ranges of concentrations that are associated with biological effects. This is a way of dealing with the many uncertainties of the data on which the guidelines are based and fits in with a risk based approach: i.e. SQGs are not pass or fail standards (legally defensible), but if exceeded are used to trigger action or further investigation (decision support system – DSS) at a site specific level. It is also a way of estimating the probability of adverse biological effects.

The ANZECC & ARMCANZ (2000) have developed two DSS frameworks to assess the risk of sediments to ecosystem function, depending on the type of potential contaminant. The structures of both are very similar, but vary in the types of factors controlling bioavailability that need to be considered. The total concentration of the sample is compared to the SQG value. If the measured concentration is below the trigger value risk to the environment is low. If the measured concentration is exceeded, it is compared to background concentrations (taking into account grain size) and factors controlling bioavailability. In the case of metals these include: acid-soluble metal analysis, acid volatile sulfides, characteristics of the pore water (salinity and pH) and speciation. In the case of organics these include organic carbon content and once again characteristics of the pore water (salinity and pH)(Figures 3.1 and 3.2)(Batley and Maher, 2001). If the concentration still exceeds the guidelines value, then toxicity testing should be undertaken as the ultimate means of assessing sediment toxicity (toxicity testing can be undertaken at any stage of the risk assessment process, however high costs usually result in it being used as a measure of last resort)(Batley and Maher 2001). Toxicity testing is used to demonstrate an absence of toxicity, however if toxicity is observed it cannot be necessarily attributed to a specific contaminant if the sediment sample consists of a mixture of chemicals. In this case, toxicity identification and evaluation procedures would need to be implemented (Batley and Maher, 2001).

An example of the how these DSSs for ANZECC (2000) can fit within a larger management approach is show in Figure 3.3.

TABLE 3.4 SQG benchmarks define ranges of chemical concentrations and associated probability of observing biological effect. Adapted from CCME (1995)

SQG benchmarks	Potential biological effects at these concentrations	Risk	Possible management options
<ERL/TEL/TEC	Rarely associated with biological effects	Little toxicological concern	Protect existing sediment quality conditions and continue monitoring
>ERL/TEL/TEC <ERM/PEL/EEC	Occasionally associated with biological effects	Potential toxicological concern	Although adverse biological effects are possible within this range of concentrations, their occurrence and severity are difficult to predict on an a priori basis as site specific conditions are likely to control bioavailability. Thus further site specific investigations are needed: determine background concentrations; and undertake biological assessments
>ERM/PEL/EEC	Frequently associated with biological effects	Significantly hazardous to exposed organisms	Site specific investigation needed. Undertake biological assessments to determine nature and extent of effects. Undertake remedial action if possible

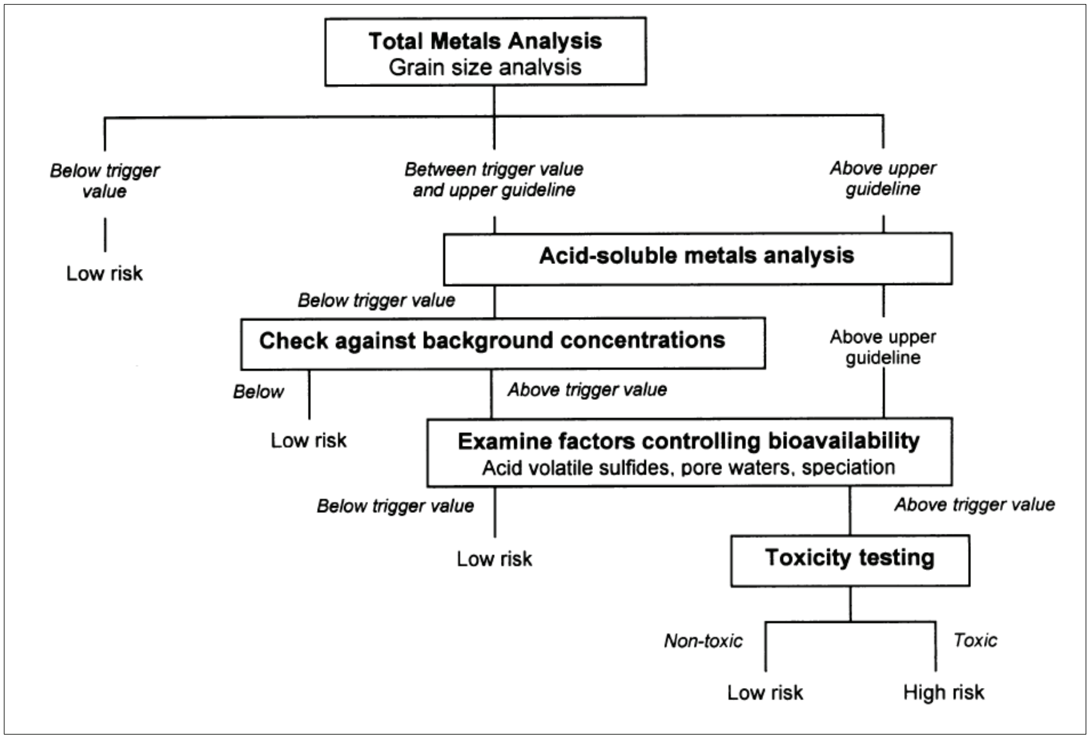

FIGURE 3.1 Sediment quality decision support system for metals in Australia and New Zealand (Batley and Maher, 2001)

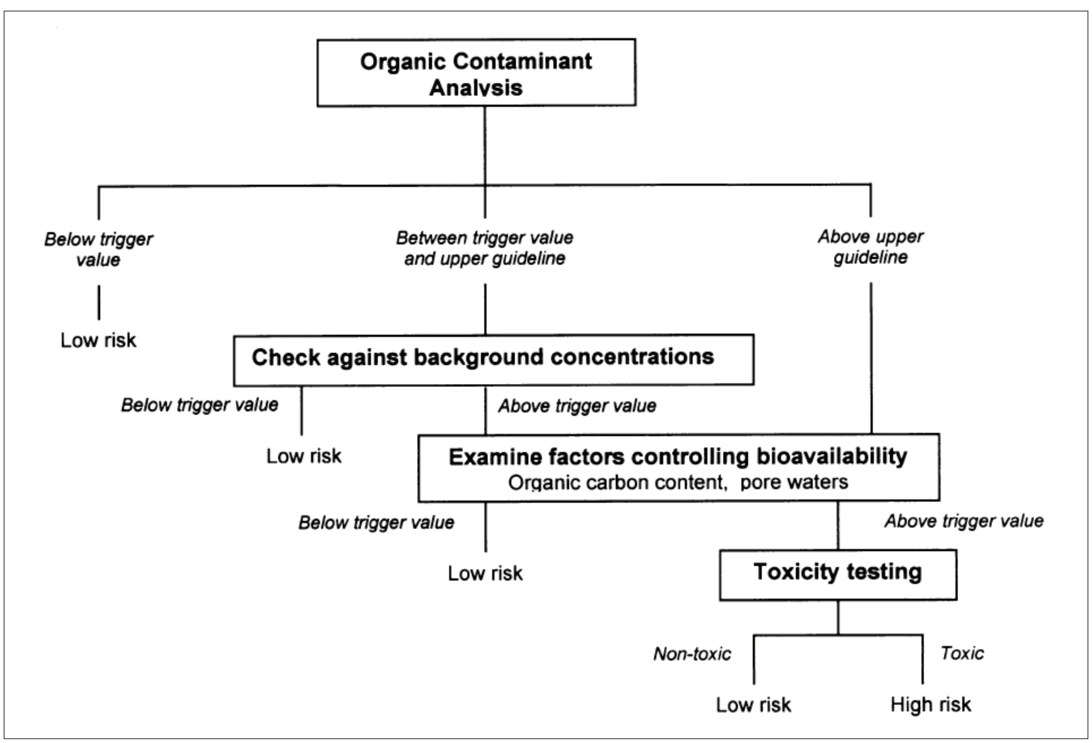

FIGURE 3.2 Sediment quality decision support system for organic contaminants in Australia and New Zealand (Batley and Maher, 2001)

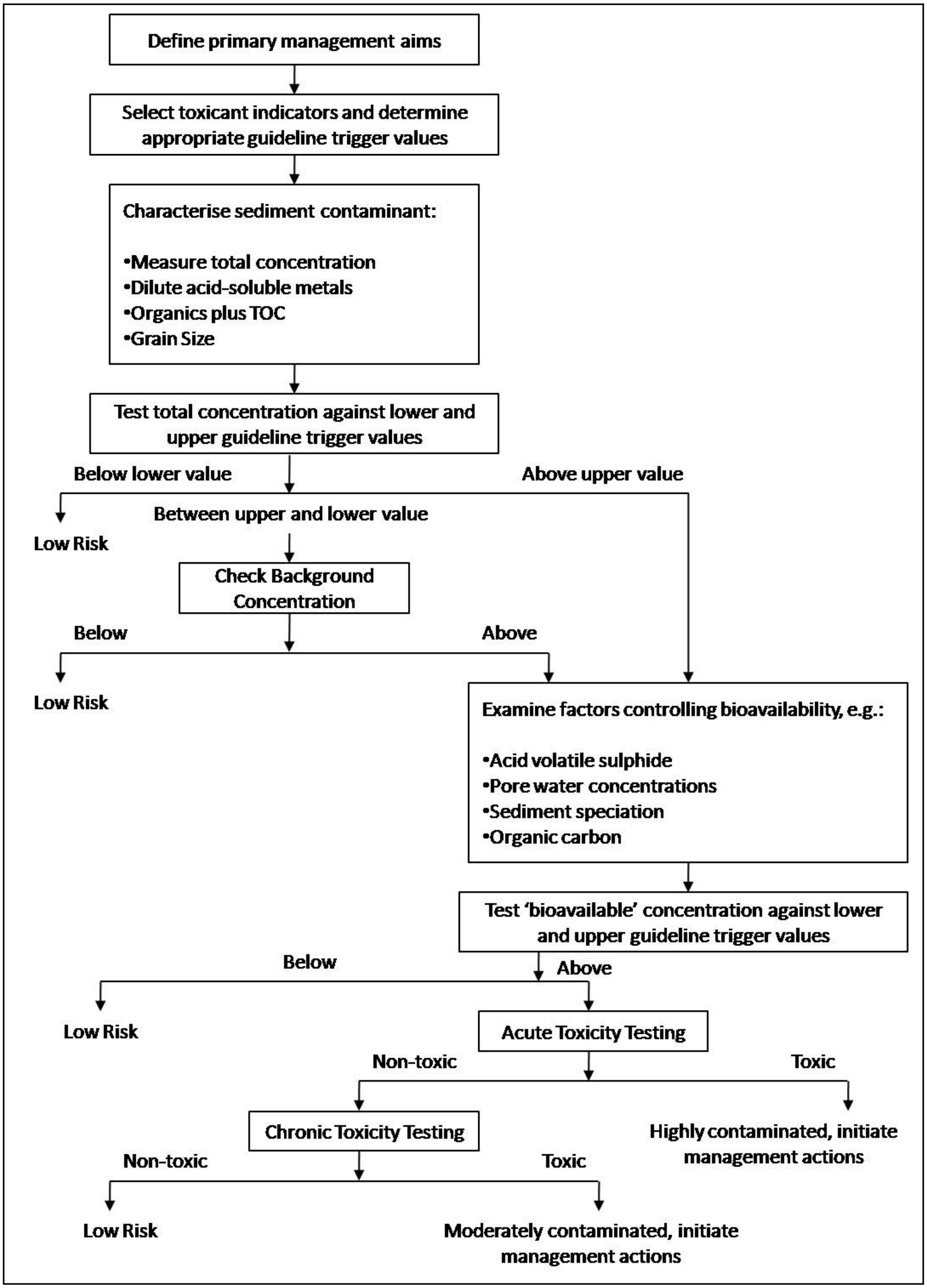

FIGURE 3.3 Application of sediment quality guidelines in Australia and New Zealand as part of monitoring programmes (CSIR, 2006).

Sampling methodology
Acceptable sampling and analytical methods used to generate field data which are then compared with the SQGs to derive a risk value should be stipulated in order to ensure that they align with the assumptions made in deriving the guidelines. Concerns were raised that variability in sediment chemistry at the meter scale was enough to be of concern.

Standards for sediment sampling have been developed and should be undertaken in accordance with ISO 5667-16 (Hansen et al., 2007). The USEPA (2001) have also developed methods for collection, storage and manipulation of sediments for chemical and toxicological analysis. Bioaccumulation methodology is detailed in USEPA (2000b).

Should suspended or settled sediments be analysed:
Unresolved. There was a feeling that they should be analysed together, but there are significant methodological challenges in attempting this.

Should sediment as a physical stressor be included in SA SQGs?
Unresolved. It was suggested that sediment as a physical stressor in the water column should perhaps should be dealt with by the WQGs. Fine sediment in large amounts could be a stressor for epibenthic organisms too though.

Which chemicals require guideline derivation for South African SQGs?
This topic will need the attention of a focus group. However some suggestions were made in the workshop. It was suggested that organics would probably be determined by the list being developed for the revised water quality guidelines (should also include anthracene and endocrine disrupting compounds). All elements being dealt with in the water quality guidelines should be included in the SQGs for the sake of continuity. Potential metals could be determined by identifying industries functioning within South Africa – suggestions made included: aluminium, manganese, cadmium, lead, chromium, uranium, copper, tungsten, cobalt. Semi metals like arsenic and selenium were suggested.

CHAPTER 4: POSSIBLE FUTURE RESEARCH FRAMEWORK

4.1 Introduction

In the course of producing this research report it is evident that a number of issues require considerable more attention.

- The alignment of the SQGs with the water quality guidelines (WQGs) being currently revised for South Africa was seen as important by workshop attendees. The philosophical approach to the derivation of the WQGs has in most part been decided upon, however the derivation procedure for WQGs for aquatic ecosystems, the assessment of data integrity for use in derivation and the physical structure of the WQGs (in the form of a software decision support system) is yet to be finalised. The WQG revision project will address the above issues in the coming two years and consequently it would be important for the development of the SQGs if the project team could participate in this process.
- Workshop attendees recognized that considerably more discussion was required before a decision regarding the most appropriate biological data for deriving the SQGs could be confidently made.
- Furthermore, it was also recognized that considerably more discussion was required before a decision regarding the most appropriate derivation method for the SQGs could be made.

This chapter presents a possible future research framework that will address the above issues and lead to the development of SQGs for South Africa that are scientifically defensible, reflect the current international state-of-the-art regarding assessment of contaminated sediments, and are correctly aligned with water resources management strategies in South Africa (e.g. resource classification and Reserve determination).

4.2 Research framework

Issue 1	Research task/s	Possible actions
Philosophical approach, structure and implementation of SQGs	1. Ensure that the philosophical approach of the SQGs continues to be aligned with that of the proposed WQGs. 2. Determine if the physical structure of the WQGs (in the form of the software decision support system) can be applied to the SQGs.	Attend the WQG revision meetings to ensure that the philosophical approach, structure of the guidelines and principles of implementation of the new SQGs are aligned with the WQGs

Issue 2	Research task/s	Possible actions
The most appropriate data and derivation method for South Africa	1. Collate all (freshwater and saltwater) potential sediment toxicity data (local and international). This process will be independent of the rules for data suitability or rules for data use in deriving the SQGs (these will be determined separately). This will be a reference source for available and potential data and will provide an overview of the types and availability of (useful) data. 2. Through consensus with relevant	Appropriate specialists will need to be identified for the formation of focus groups who will address the proposed research tasks

	experts, determine the most appropriate SQG derivation method for South Africa. 3. Identify or develop a data integrity/suitability assessment system for sediment toxicity data before their application in a derivation method. 4. Collate a list of all SQG values in use internationally. These will form a reference against which SQG values derived for South Africa can be compared and assessed. 5. Derive guidelines for a small number of chemicals using alternate data and derivation methods. Assess the guideline values derived theoretically (i.e. against internationally accepted guideline values or hazard assessments for that chemical), or empirically (i.e. with field collected data from reference and impacted sites)	

Issue 3	Research task/s	Possible actions
Improve the capacity of organizations in South Africa to undertake sediment toxicity testing and analysis of contaminated sediments	1. Experiment with the development of indigenous benthic organism-based sediment toxicity tests. 2. Develop capacity to analyse sediment samples for contaminants	Identify organisations which have ability to develop and strength capacity in respect of the identified tasks

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Appendix A. Workshop report for WRC project K8/793 : Developing sediment quality guidelines for South Africa – Phase 1.

Date: 19 November 2008
Time: 09:30–15:30
Venue: Water Research Commission, Marumati Building, Pretoria

Present:

Gordon, Andrew	Institute for Water Research, Rhodes University (Chair)
Bollmohr, Silke	Department of Water Affairs and Forestry
Glass, Jenny	Council for Geoscience
Jooste, Dr Sebastian	Department of Water Affairs and Forestry resource quality services
Jordaan, Wikus	Council for Geoscience
Oberholzer, Paul	CSIR (until 11:00)
Shaddock, Bridget	Golder
Thwala, Melusi	Department of Water Affairs and Forestry
Wade, Dr Peter	Council for Geoscience
Wepener, Victor	Department of Zoology, University of Johannesburg
Arnold, Robyn	Write Connection (Scribe)

Apologies:
Kempster, Dr Phillip
Mitchell, Dr Steve
Muller, Nikite
Oelofse, Suzan
Rossouw, Nico

The need for sediment quality guidelines:

It was agreed that there was a need for SQGs to be developed for implementation in South Africa. The ultimate goal is the development of inclusive chemical-specific standards for the aquatic resource as a whole (which would incorporate water, sediment and riparian habitat compartments). Unfortunately, this is not feasible at present and thus separate SQGs should be developed.

Topics discussed at the workshop focused particularly on:

1. the philosophical approach for deriving the guidelines;
2. what the basic structure of the guideline will look like and;
3. identifying issues for further investigation by focus groups in future phases of the project

1. Philosophical approach

A new approach to deriving SQGs is envisaged. It is based on the approach being developed for the revision of the water quality guidelines in South Africa (DWAF 2008), which is currently underway. It was felt that a new approach was needed as previous SQGs developed by other jurisdictions were not site (or scenario) specific and were misused as those applying them were unaware of the underlying assumptions made in deriving the values. By adopting an explicit probabilistic risk assessment approach (where all available and/or applicable hazard and exposure information is taken into consideration) the person applying the SQG (e.g. water resource manager) is able to make an informed decision as they understand what is at risk, what data has been used or assumptions made in determining that risk, and what the resultant confidence in the SQG value should be.

The application of the SQGs, from a regulatory point of view, was seen in two contexts: a) assessments (i.e. what can you say about how good or bad the situation is from a set of field collected data); and b) setting sediment quality objectives (defining what is an acceptable risk to the aquatic environment and determining the sediment quality associated with that risk).

2. Guideline structure

As the SQG structure is envisaged to be similar to that being developed for the revised water quality guidelines, some of what was discussed during the workshop has been succinctly conveyed in DWAF (2008). Consequently, some of ideas presented in that document have been included in this document (extracts are presented in a different font):

Many people's idea of sediment quality guidelines are of a table of numbers that define concentration ranges within which specified biological effects can be expected for specific chemicals. The approach envisaged for both the sediment and the revised water quality guidelines requires the guidelines to be much more than this:

"The guidelines" include all of the following:

- The numerical values that define ranges within which specified effects, or degrees of effect, can be expected;
- The narrative description of those effects (e.g. lethality in the test organism population; changes in community structure);
- The description of the sampling, sample preparation, and analytical procedures upon which the numerical values are based;
- The general information on the constituents (like chemical, physical, and toxicological properties) – the so-called "hazard description";
- The general information on such issues as occurrence in the environment and general behaviour (like typical fate and transport); and
- Mitigation (treatment) options that provide guidance on what can be done about specific problems.

"The guidelines" therefore include all the information that may be useful to the reader, particularly in supporting informed decision making relating to water resource management.

The primary tool facilitating the determination and use of the guidelines should be a software decision support system (DSS). This should be complemented with a set of hard copy manuals that at least present generic values and supporting information. A more detailed description of the DSS is presented in DWAF (2008).

Details regarding the specific type of data to be used in the derivation process (e.g. spiked sediment toxicity test data or field collected cause-effect data), the source of the data (international or local or combination) and the required quality of the data were not discussed in detail at the workshop. These details will need to be the topic of further investigation by a focus group in subsequent phases of the project.

The sediment quality guidelines are envisaged to comprise of a three tier system:

Tier 1. Provides 'generic' guidelines values that are made available in the DSS and hard copy manuals. These guideline values will be conservative as the worst case scenario is assumed.

Tier 2. Allows for site specificity in specified contexts and is facilitated by the DSS, consequently there is more confidence in the derived value

Tier 3. Full risk assessment. Not facilitated by the DSS but will use information contained within the DSS information database.

The procedure for generating the guideline values will follow a probabilistic risk assessment process, starting at tier 3 and progressing to tier 1 (the exact procedural methods are still to be resolved). Associated with the move down the tiers is a loss in confidence in the resultant guideline value and thus the need to be more conservative (through use of conservative data, perhaps inclusion of safety factors and/or reducing what is considered acceptable risk to the resource). A description of the possible derivation procedure is as follows:

- At tier 3 all exposure parameters that would affect the toxicity of the chemical to the organism are identified (these parameters were discussed by workshop attendees and a tentative list produced – see Table 1). In addition to the exposure parameters, hazard characterisation (identifying and/or generating relevant toxicity data) has to be undertaken (workshop attendees discussed the various options available for measuring the effect of toxicants on organisms – see Table 3). The data required for the exposure parameters and hazard characterization must be generated if not available.
- If this is not desirable or possible, then a tier 2 assessment is undertaken instead. At this level, there are mandatory exposure parameters (for which data are required) and preferable exposure parameters (if no data are available, assumptions can be made). Ideas on what these should perhaps be were proposed at the workshop (Table 2).
- If there are no data to meet the requirements of a tier 2 assessment, and no option to collect or generate data from the field or in the lab then a tier 1 assessment is undertaken. At this level, assumptions are made for all parameters and consequently the guideline value will be conservative due to the application of safety factors and/or conservative methodology. Thus, although there may be low confidence in the tier 1 guideline value as a result of the assumptions made in its derivation, at least the person applying the guideline is aware of the assumptions and the level of confidence as the 'record of decision' detailing the derivation process and data utilized will be explicit.

Although on most occasions people applying the SQGs will probably use the Tier 1 values it is critical to have the option to derive a guideline value that represents greater confidence and scenario specificity at Tiers 2 and 3. In addition the tiered system shows that the thinking has been done at the higher tiers and the implications of using the value at the lower tier are explicit.

The output of the SQG derivation process will be different depending on the application.
For assessments: each sediment quality variable for which concentrations are specified by the user should have a quantitative estimate of risk (e.g. expressed as a percentage) and a corresponding narrative description of the fitness (e.g. excellent, good, fair, unacceptable) associated with the guideline value.
For sediment quality objective setting: each sediment quality variable of interest specified by the user should have a corresponding generic value (for comparison purposes) and a site-specific value.

3. Issues for further investigation

3.1 Hazard assessment method

In addition to the need for further work regarding the most appropriate methods for hazard characterization (Table 3), the method to be employed in the hazard assessment for each tier will need to be investigated by a focus group. It is envisaged that there will be a database of suitable quality toxicity data associated with the DSS. Issues to be dealt with by the focus group will be: establishing rules for determining the suitability of toxicity data for inclusion in the database; establishing a protocol for including new data when it becomes available; and determining a method for utilizing the toxicity data in a hazard assessment. Delegates discussed the possible application of species sensitivity distributions (SSDs) for the purpose of hazard assessment. Concerns were raised regarding the undesirable flexibility in the tails of the Burr distribution used by the Australians in their water quality guideline derivations. The addition of data points in a Burr distribution is said to alter (to an unacceptable level) the lower end of the distribution, e.g. the 5^{th} percentile which is often used in guideline derivation. A possible alternative for investigation is the Pareto distribution, which is not as flexible in the tails, but consequently doesn't always fit the data as well as the Burr distribution.

Another issue for further investigation is the use of Bayesian statistics when SSDs are updated with new information.

Table 1 – Potential exposure assessment parameters for tier 3 assessment

Parameter	Comment
Organic carbon content	Provides binding site for contaminants
AVS/SEM ratio (acid volatile sulfide / sequentially extracted metal)	Method for assessing bioavailability of metals
Sediment oxygen demand	Implications for bioavailability of contaminants
Particle size distribution	Water flow will affect particle size distribution, dictating habitat availability, which in turn dictates organisms found at a site. As different organisms have different uptake routes, susceptibility to a chemical stressor will vary. In addition, surface area affects the available binding sites for contaminants
pH	Affects desorption and release to interstitial water of sediment bound metals
Alkalinity / carbonate	Affects bioavailability of sediment bound contaminants
Chloride concentration	
Hardness / calcium and magnesium	Affects bioavailability of sediment bound contaminants
Redox potential	Affects desorption and release to interstitial water of sediment bound metals
Ammonia (NH3)	Although a constituent it will affect other constituents
Total dissolved solids (TDS)	
Flow	Will affect bioavailability and spatial distribution of contaminants either directly or indirectly
Anthropogenic nanoparticles	Impossible to quantify or qualify as a hazard at the moment, but is included as an exposure parameter
Historical profile of the catchment	Provides clues on possible contaminant exposure and geochemical characteristics of sediment
Bioaccumulation	Toxicity may be attributable to bioaccumulated toxin concentration instead of toxin's external environmental concentration
Routes of exposure / sediment ingestion rate	Affects bioavailability of sediment bound contaminants
Background concentrations	Deriving a guideline value using 3.6 X background conc has been shown to be protective (Bjorgesaeter and Gray 2008)

Table 2 – Potential exposure assessment parameters required for tier 2 assessment	
Mandatory	**Preferable (can make assumptions)**
pH	Organic carbon
Flow	Redox potential
Particle size distribution	Alkalinity / carbonate
AVS/SEM (for metals)	Bioaccumulation
	Hardness / calcium and magnesium
	Background concentrations

Table 3 – Hazard characterisation	
A discussion was held regarding what the most applicable toxicity tests, endpoints (in terms of a) biological organization (subcellular, individual, population, community), b) length of exposure) and test organisms. It was realized that this issue will require considerable further attention by a focus group of experts in subsequent phases of the project. Issues or synergistic and antagonistic toxicity effects should be addressed. However, the following was noted:	
Potential test organism	**Associated endpoint(s)**
Chironomus	Mortality (lethal); mouth part deformation (sublethal)
Microbes/bacteria	Growth inhibition - Luminescence bacteria (Microtox) Community structure
Epibenthic diatoms	Community structure
Various epibenthic invertebrates	Endpoints can be generated in short-term lethality tests in the laboratory, longer term sublethality tests in micro/mesocosms, and from observations in the field. The endpoints measured can be mortality, biochemical, behavioural, and community responses such as indices (Shannon Wiener diversity index) and multivariate statistics and other statistics (Bray Curtis index of similarity)

A proposal for the use of SSDs in the tiered SQG guideline approach was discussed:
At Tier 3: the full risk assessment will generate hazard data which will feed into the SSD and update the hazard data obtained from the DSS database.
At Tier 2: The hazard data from the DSS database are used as is and the required exposure parameters are noted. The output is a scenario specific concentration/guideline with probability estimates (i.e. some measure of confidence). Perhaps the 5^{th}% of the SSD is used as the guideline.
At Tier 1: Only the hazard data from the DSS database are use in an SSD. A more conservative approach is adopted and possibly safety factors adopted or perhaps the 1^{st}% of the SSD used instead of the 5^{th}% for example.

3.2 What is the aim of SQGs?

This needs to be explicitly stated. It was discussed, but not entirely resolved, at the workshop and will require further deliberation of the following: Protect what, when, for how long? Possibly to protect all forms of aquatic life during all forms of aquatic life cycles for an indefinite period of exposure to substances associated with the sediment?

3.3 Which chemicals require guideline derivation for South African SQGs?

This topic will need the attention of a focus group. However some suggestions were made in the workshop. It was suggested that organics would probably be determined by the list being developed for the revised water quality guidelines (should also include anthracene and endocrine disrupting compounds). All elements being dealt with in the water quality guidelines should be included in the SQGs for the sake of continuity. Potential metals could be determined by identifying industries functioning within South Africa – suggestions made included: aluminium, manganese, cadmium, lead, chromium, uranium, copper, tungsten, cobalt. Semi metals like arsenic and selenium were suggested.

3.4 Type and quality of toxicity data to be included in DSS database?

This will be fully investigated by a specialized focus group. The issue of data quality control is important. It was suggested that a preoccupation with local/indigenous data at the expense of international data should be avoided. International data should be used if it is applicable to SA conditions (e.g. particle size mineralogy, organic carbon). If applicability of data is questionable, it could still be used as a reference point to guide the derivation process.

3.5 Should SQGs for freshwater and saltwater sediments be derived from separate datasets?

Further discussion is needed. It was noted that crustaceans were more sensitive than other invertebrates in both fresh and saltwater, and thus the origin of the organism might be less important than the type of organism. Perhaps representative taxonomic groups need to be specified for inclusion in SSD hazard assessment. A further point raised suggested that organics may have higher bioavailability in saltwater.

3.6 Which derivation method to use?

Needs to be fully investigated by a focus group. However, it was noted that using a probabilistic approach allows the use of a weight of evidence approach utilizing all derivations methods.

3.7 Are guidelines based on equilibrium partitioning (EqP) methods acceptable in a regulatory framework?

The many uncertainties associated with EqP were noted. However, from a regulatory point of view, if the scientists are confident of results using EqP there would be not objection from the regulators.

3.8 Should suspended or settled sediments be analysed:

Unresolved. There was a feeling that they should be analysed together, but there are significant methodological challenges in attempting this.

3.9 Should sediment as a physical stressor be included in SA SQGs?

Unresolved. It was suggested that sediment as a physical stressor in the water column should perhaps should be dealt with by the WQGs. Fine sediment in large amounts could be a stressor for epibenthic organisms too though.

3.10 Application of SQGs to dredging risk assessments

Was briefly discussed. It was noted that the disturbance of sediment by means of dredging is not classified as a water 'use' (like waste discharge) and thus is not covered by the National Water Act, which regulates water users.

3.11 Sampling methodology

Acceptable sampling and analytical methods used to generate field data which are then compared with the SQGs to derive a risk value should be stipulated in order to ensure that they align with the assumptions made in deriving the guidelines. Concerns were raised that variability at meter scale was enough to be of concern.

3.12 Aligning SQGs with resource classification system

Unresolved. This will need the attention of a focus group and is also an issue for current revision of the WQGs. However, it should be noted that in order to derive suitable risk assessment endpoints there will need to be a further interpretation of the current classification system which does not clearly delineate what that endpoint is/should be. This is related to what sort of endpoint (e.g. organismal lethality, sub-organism effects, population level effects, etc.) and the extent of those effects (e.g. 1%, 5% or 10% of an SSD) are used as risk assessment endpoints.

3.13 Aligning the SQGs and WQGs

It is anticipated that they will be closely aligned. Details of how this will work are unresolved; however they could potentially be developed concurrently.

Reference

BJORGESAETER A and GRAY JS (2008) Setting sediment quality guidelines: A simple yet effective method. *Marine Pollution Bulletin* **57** 221-235.

DWAF (2008) *Development of SA risk-based water quality guidelines: Phase 1. Needs assessment & philosophy*. Report No N/0000/SRQ0107. ISBN: 978-0-621-38380-5. Resource Quality Services, Department of Water Affairs and Forestry, Pretoria.