Chapter 3

An Evaluation of Existing Approaches to Developing Numerical Sediment Quality Guidelines

3.0 Introduction

A variety of approaches have been devised to formulate sediment quality guidelines (SQGs). These approaches have been reviewed and summarized by Chapman (1989), Persaud *et al*. (1989), Beak Consultants. Ltd. (1987; 1988), EPA (1989a; 1992), Sediment Criteria Subcommittee (1989; 1990), and MacDonald *et al*. (1992). The discussion on the major approaches to the development of SQGs has been abstracted from these documents to provide a basis for recommending an appropriate procedure for Florida. The major approaches to developing SQGs are:

(i) Sediment Background Approach (SBA);
(ii) Spiked-Sediment Bioassay Approach (SSBA);
(iii) Equilibrium Partitioning Approach (EqPA);
(iv) Tissue Residue Approach (TRA);
(v) Screening Level Concentration Approach (SLCA);
(vi) Sediment Quality Triad Approach (SQTA);
(vii) Apparent Effects Threshold Approach (AETA); and,
(viii) Weight of Evidence Approach (WEA).

Discussions on each of the approaches has been divided into four main sections, including a brief description of the methodology, the major advantages and limitations of the approach, and the current uses of the approach. All of the approaches identified are directly applicable for deriving numerical SQGs. However, there are other procedures which are focused on site-specific assessment of sediment quality [e.g., the International Joint Commission sediment assessment strategy (IJC 1988), benthic community structure assessments, etc.]. These latter procedures are described by the EPA (1989a; 1992) and MacDonald *et al*. (1992).

3.1 Sediment Background Approach (SBA)

In this approach, sediment contaminant concentrations at a site (or a sedimentary stratum) that is being assessed are compared to the concentrations of those contaminants at sites that are considered to be representative of background (natural) conditions. Alternatively, historical records for a specific site or, more appropriately, data from sediment core profiles may be used to define background levels of specific contaminants. Using this approach, a site would be considered to be contaminated if the concentration of one or more contaminants exceeds the mean background concentration by a significant margin (e.g., two standard deviations or more). Application of this approach requires special care in choosing the location of sampling stations, in sample preparation, in sample analysis methodology, and in quality assurance/quality control (QA/QC).

The major advantage of this approach lies in its simplicity. It relies on measurements that can be made easily in most analytical laboratories, it provides a simple means of comparing monitoring program results with the guidelines (i.e., it yields chemical concentration values), it is specific to conditions at the site, it does not have extensive data requirements, and it does not require toxicity testing.

The major limitation of this approach is that no direct biological effects or bioavailability data are used for deriving guidelines. In addition, this approach only applies to major and trace elements, for which natural background concentrations can be identified from sediment core samples. The background concentrations of anthropogenically-derived organic contaminants should be zero, although it is well established that detectable concentrations of many of these contaminants occur due to the long range transport of atmospheric pollutants. Moreover, a variety of hydrocarbons may occur naturally in sediments that are affected by seepages from oil-bearing formations. While SQGs may be established at contemporary background levels, it is not clear whether or not these guidelines would be sufficiently protective of aquatic biota.

This approach has been used successfully at a number of locations in the United States and elsewhere in the world. In the Great Lakes, this approach was used by EPA Region V to develop a classification system for harbors (SAIC 1991) and to assess the applicability of SQGs for evaluating open-water disposal of dredged materials (Persaud and Wilkins 1976; Mudroch *et al*. 1986; 1988). Similarly, this approach has been used by the United States Geological Survey, EPA Region VI, Texas Water Quality Board, Virginia Water Control Board, Illinois Environmental Protection Agency, and several other agencies to establish informal guidelines for determining whether sediment contaminant concentrations exceed 'normal' levels (SAIC 1991).

Background levels of naturally-occurring substances vary significantly between areas. For this reason, SQGs developed using this approach specifically apply only to the areas that were considered in their development. However, the FDEP (Schropp *et al.* 1990) and others

(e.g., Loring 1991) have developed unique applications of the SBA which improve its overall reliability. These applications rely on normalization of metals levels to the concentration of a reference element, such as aluminum or lithium. Statistical analysis of data from numerous uncontaminated sites provides a means of establishing background levels of metals under a variety of conditions and, as such, a basis for identifying sites with anthropogenically-enriched levels of metals. The SBA alone is not sufficient for formulation of toxicity-based SQG values, but data on background concentrations of specific contaminants provides critical information for assessing the applicability of SQGs developed using other approaches and for formulating site-specific sediment quality objectives.

3.2 Spiked-Sediment Bioassay Approach (SSBA)

The SSBA to generating SQGs relies on empirically generated information on the responses of test organisms to specific contaminant challenges, generally under laboratory conditions. In this procedure, clean sediments are spiked with known concentrations of contaminants (individually or in mixtures) to establish definitive cause and effect relationships between chemical concentrations and biological responses (i.e., mortality, reductions in growth or reproduction, physiological changes, etc.). The SSBA has been used successfully with various types of sediments, generally for single contaminants or relatively simple mixtures of contaminants (e.g., Cairns *et al*. 1984; McLeese and Metcalfe 1980; Swartz *et al*. 1985; 1988; 1989). Typically, numerical SQGs are derived by applying a safety factor to the lowest observed effect level from a study on the most sensitive species (MacDonald and Smith 1991); however, a variety of other specific procedures may be employed.

The major advantage of this method is that it is suitable for all classes of chemicals and most types of sediments. In addition, it has the capability to produce precise dose-response data pertaining to toxic chemicals. It can also account for factors that are thought to control the bioavailability of these substances, such as total organic carbon (TOC) and acid volatile sulphide (AVS; EPA 1990). As such, guidelines derived using spiked-sediment bioassay data should be highly defensible.

The major disadvantage associated with the implementation of this method for deriving SQGs is that spiked-sediment bioassays have only been conducted on a few species of aquatic biota and with only a limited number of substances (i.e., cadmium, copper, a few pesticides, and several PAHs). Therefore, the existing database would support the derivation of numerical SQGs for only a few contaminants. Significant expansion of this database (i.e., to include the range of substances that are expected to occur in coastal sediments) will require substantial resources and these are not likely to be available to state agencies. In addition, uncertainties associated with spiking procedures, equilibration periods, and factors controlling the bioavailability of the substances may limit the interpretation of the results of spiked-sediment bioassays.

In addition to their potential role in deriving numerical SQGs, data developed using this approach are fundamental for evaluating the applicability of guidelines that have been developed using other approaches. For example, Environment Canada has recently developed a formal protocol for developing SQGs and data from spiked-sediment bioassays play an important role in this process (Smith and MacDonald 1994). Likewise, Hansen *et al.* (1993e) used spiked-sediment bioassay data to evaluate the applicability of the sediment quality criteria that have been developed for fluoranthene. Similarly, Outridge *et al.* (1992) used data from spiked-sediment bioassays to evaluate the applicability of SQGs for cadmium derived using the weight of evidence approach (Smith and MacDonald 1994) .

3.3 Equilibrium Partitioning Approach (EqPA)

The water-sediment EqPA has been one of the most studied and evaluated approaches used to develop SQGs (primarily for non-polar hydrophobic organic chemicals) in the United States (Pavlou and Weston 1983; Bolton *et al*. 1985; Kadeg *et al*. 1986; Pavlou 1987; Di Toro *et al.* 1991). This approach is based on the assumption that the distribution of contaminants among different compartments in the sediment matrix (i.e., sediment solids and interstitial water) is predictable based on their physical and chemical properties. It also assumes that continuous equilibrium exchange between sediment and interstitial water occurs. This approach has been supported by the results of sediment toxicity tests, which indicate that positive correlations exist between the biological effects observed and the concentrations of non-polar organic contaminants measured in the interstitial water.

In the EqPA, water quality criteria developed for the protection of marine organisms are used as the basis of the SQGs [termed sediment quality criteria (SQC) by the EPA] derivation process. As such, the water quality criteria formulated for the protection of water column species are assumed to be applicable to benthic organisms (Di Toro *et al*. 1991). Sediment quality guidelines are calculated using the appropriate water quality criteria (usually the marine final chronic values) in conjunction with the sediment/water partition coefficients for the specific contaminants. The calculation procedure for non-ionic organic contaminants is as follows:

$$\mathbf{SQG = Kp\ FCV}$$

where:

SQG = Sediment quality guideline (in μg/kg);
Kp = Partition coefficient for the chemical (in L/kg); and,
FCV = Final chronic value (in μg/L).

Currently, this procedure is considered to be appropriate for deriving SQGs for non-ionic organic substances, such as PAHs, polychlorinated benzenes, biphenyls, dioxins, and furans,

and many pesticides (EPA 1991). To date, EPA has developed SQGs for five substances, including fluoranthene, acenaphthene, phenanthrene, endrin, and dieldrin (Hansen *et al.* 1993a, b, c, d, e). For these substances, TOC normalization may provide a basis for predicting toxicity to aquatic organisms (Swartz *et al.* 1990). In addition, the role of AVS in determining the bioavailability of metals is also under investigation (Di Toro *et al*. 1989), and efforts are currently under way to establish normalization procedures for metals as well (Di Toro *et al.* 1992). Di Toro *et al.* (1991) have also noted that porewater dissolved organic carbon (DOC) levels may influence the bioavailability of hydrophobic compounds; however, the nature of this relationship has not been fully established.

One of the principal advantages of this approach is that it is applicable to a wide variety of aquatic systems because it considers the site-specific environmental variables that are thought to control the bioavailability of sediment-associated contaminants (i.e., TOC and AVS). In addition, this approach is practical for implementation with a broad suite of substances because it requires only existing water quality criteria and contaminant sediment/water partition coefficients to support the derivation of SQGs. Confidence in the validity of this approach is further enhanced because the EqP theory upon which this approach is based is well developed, it has already been used in various regulatory and remedial action applications, and it provides a consistent basis for identifying the severity of sediment contamination (EPA 1989a).

However, there are a number of limitations to this approach which may restrict its applicability for deriving numerical SQGs. Specifically, SQGs developed using the EqPA do not explicitly address possible synergistic, antagonistic or additive effects of contaminants. In addition, the technical basis for developing SQGs for metals is still under development. Furthermore, the interim SQGs for non-ionic chemicals apply only to sediments that have significant organic carbon contents ($\geq$ 0.5 percent), yet the relationship between toxicity of fluoranthene and TOC levels has only been quantitatively established at low levels of TOC (i.e., < 0.5%; Swartz *et al.* 1990).

Another disadvantage of the EqPA is related to the limited number of reliable partition coefficients available for many priority contaminants. For example, the 95% confidence interval associated with the K_{oc} of endrin spans more than two orders of magnitude (EPA 1991). This variability in the estimate of the partition coefficient generates considerable uncertainty in any SQGs derived using these data. Furthermore, *in situ* sediments are seldom, if ever, at equilibrium and are likely to achieve steady state conditions only rarely. Several other limitations of the approach were identified by Di Toro *et al.* (1991), all of which are considered to restrict the application of SQGs developed using the EqPA (Sediment Criteria Subcommittee 1989).

In spite of its limitations, the EqPA has been selected by the EPA as a primary basis for deriving SQGs and the EPA has expended considerable effort in the development of the technical basis of the approach (Di Toro *et al*. 1991). While the initial review by the Science

Advisory Board (SAB) was not very positive (Sediment Criteria Subcommittee 1989), the subsequent review commended EPA for its progress towards reducing the uncertainties associated with the approach (Sediment Quality Subcommittee 1992). However, the SAB recommended that SQGs derived using the EqPA should be field validated to reduce uncertainty, expressed as ranges to facilitate sediment assessments, and further tested to improve the method. This approach has been used primarily in the United States, however, the applicability of the approach for deriving SQGs has also been evaluated by several other jurisdictions [i.e., Canada (MacDonald *et al*. 1992), Ontario (Persaud *et al*. 1990) and the Netherlands (Van Der Kooij *et al*. 1991)].

3.4 Tissue Residue Approach (TRA)

The TRA (which is also known as the biota-water-sediment equilibrium partitioning approach) involves the establishment of tolerable sediment concentrations for individual chemicals or classes of chemicals by determining the chemical concentrations in sediments that are predicted or observed to result in acceptable tissue residues. This process necessitates the development of relationships between concentrations of contaminants in sediments and contaminant residue levels in aquatic biota. In addition, relationships between contaminant residues in aquatic biota and adverse effects on consumers of these species must be established. Several methods are available to derive guidelines for levels of contaminants in the edible tissues of aquatic biota (see MacDonald 1991; Walker and MacDonald 1993).

The principal advantage of this approach lies in its simplicity. Sediment quality guidelines may be derived directly from tissue residue guidelines for the protection of human health or wildlife consumers of aquatic biota, if acceptable sediment to biota bioaccumulation factors (BAFs) are available. The other main advantage of this approach is that it explicitly considers the potential for bioaccumulation of persistent toxic substances.

The chief disadvantage of this approach, apart from those cited for the EqPA, is that tissue residue guidelines for the protection of wildlife have not been developed and residue-based dose-response relationships have not been established for most contaminants (EPA 1989a; 1992). Therefore, SQGs would generally be developed from tissue residue guidelines applicable to the protection of human health. While guidelines, so developed, would adequately address human health concerns, other components of the ecosystem may not be adequately protected (e.g., marine mammals with high daily consumption rates of aquatic organisms). Recently, a protocol for deriving numerical tissue residue guidelines for the protection of wildlife has been developed (Walker and MacDonald 1993) and tissue residue guidelines for dioxins and furans have been derived (MacDonald 1993). Subsequently, SQGs for the protection of wildlife were formulated for these substances (MacDonald 1993).

This approach has been used on several occasions to develop water quality guidelines for the protection of human health (most notably for DDT, Hg, and PCBs). In addition, sediment contamination limits for 2,3,7,8 tetrachlorodibenzo-*p*-dioxin (T_4CDD) have been established for Lake Ontario on the basis of fish tissue residues (Endicott *et al*. 1989; Cook *et al*. 1989). Using a risk assessment approach, EPA (1993) has also derived SQGs for mammalian and avian wildlife species for this substance. The applicability of this approach for deriving SQGs is supported by data which demonstrate that declines in DDT residues in fish and birds (since its use was banned) are strongly correlated with declining concentrations of this substance in surficial sediments in the Great Lakes and Southern California Bight. As such, this approach is a logical companion for the effects-based approaches to deriving SQGs.

3.5 Screening Level Concentration Approach (SLCA)

The SLCA (Neff *et al*. 1986) is a biological effects-based approach that is applicable to the development of SQGs for the protection of benthic organisms. This approach uses matching biological and chemistry data collected in field surveys to calculate a screening level concentration (SLC). The SLC is an estimate of the highest concentration of a contaminant that can be tolerated by a pre-defined proportion of benthic infaunal species.

The SLC is determined through the use of a database that contains information on the concentration of specific contaminants in sediments and on the occurrence of benthic organisms in the same sediments. First, for each benthic organism for which adequate data are available a species screening level concentration (SSLC) is calculated. The SSLC is determined by plotting the frequency distribution of the contaminant concentrations over all of the sites at which the species occurs (information from at least ten sites is required to calculate a SSLC). The 90th percentile of this distribution is taken as the SSLC for the species being investigated. The SSLCs for all of the species, for which adequate data are available, are compiled as a frequency distribution to determine the concentration that 95% of the species can tolerate (i.e., the 5th percentile of the distribution). This concentration is termed the screening level concentration of the contaminant.

The advantages of the SLCA include its versatility and reliance on information which is generally available. It can be used to develop guidelines for virtually any contaminant for which analytical methods are available and SLCs are based on effects on organisms that are resident in marine environments. Therefore, SLCs can be adapted to local conditions by including only data on resident species.

The SLCA relies on several assumptions that may limit its applicability for SQG derivation. First, this approach assumes that the distribution of benthic organisms is related primarily to the levels of the contaminant measured in the sediments. The effects of other factors, including unmeasured contaminants, habitat composition (i.e., grain size, water current

velocity, salinity gradient, etc.), and interspecific interactions are not explicitly considered. However, some of these may be accounted for in the data analysis. Second, the approach assumes that adverse biological effects of a contaminant are manifested only by the absence of species from a particular site. Information on dose/response relationships, which may be assembled using data on population levels or sublethal effects, are largely ignored. Furthermore, the SLCA assumes that the available database includes concentrations of the contaminant over the full range of tolerance of the species.

Another major limitation of the SLCA is that it is not possible to establish a direct cause/effect relationship between any one contaminant and the benthic biota. Since single contaminants are rarely present in field situations, observed effects (presence or absences of biota) are usually dependant on the entire mixture of chemicals. Therefore, SLCs are based on *associations* between chemical concentrations and biological effects. In addition, sampling procedures may selectively bias the results of the analysis (e.g., dredge sampling may be biased towards sessile species).

Additional limitations of the SLCA are largely related to the magnitude of its information requirements. Calculation of a SLC requires information on contaminant concentrations in sediments from at least ten sites (some scientists suggest that twenty is more appropriate; e.g., Chapman 1989) and on the distribution of at least twenty species. For many contaminants, these data may not be available. Therefore, development of SQGs could require the design and implementation of a potentially costly data collection program. The SLC calculated for a particular contaminant is highly dependent on the quality and quantity of data available. Assessment of the database is difficult without *a priori* information on the sensitivities of affected species. Therefore, it is difficult to determine how much confidence can be placed on the resultant SLC.

Neff *et al.* (1986) originally developed the SLCA to derive numerical SQGs for non-polar organic contaminants in freshwater and marine sediments in the United States. The values for marine sediments were subsequently recalculated using a database that had been further verified to eliminate questionable data (Neff *et al.* 1987). While this approach appeared promising during its developmental stages, it has not been used to any significant extent in recent years. However, Ontario (Persaud *et al*. 1990) has developed a procedure for deriving numerical SQGs that relies on the strengths of this approach (i.e., lowest effect and severe effect levels are derived). Using this procedure, Ontario has developed provincial SQGs for 10 metals (Jaagumagi 1990a), PCBs, and 9 organochlorine pesticides (Jaagumagi 1990b).

3.6 Sediment Quality Triad Approach (SQTA)

The SQTA was originally developed to support site-specific assessments of sediment quality (Long and Chapman 1985; Long 1989). However, the information collected in support of

the SQTA has also been used as a basis for developing SQGs (Chapman 1986). The SQTA is based on correspondences between three measures: sediment chemistry, sediment bioassays, and *in situ* biological effects. Data on sediment chemistry and other (physical) characteristics are collected to assess the level of contamination at a particular site and to document other factors that could influence the distribution and abundance of benthic species. The results of sediment bioassays provide information that may be used to evaluate the toxicity of the contaminants that are present in bed sediments. Measures of *in situ* biological effects, such as benthic infaunal community structure and histopathological abnormalities in benthic fish species, provide information on alterations of resident communities that may be related to sediment chemistry. Integration of these three components provides comprehensive information which may be used to evaluate and rank the priority of the areas that have been surveyed. Also, they can be used to formulate site-specific sediment quality objectives; however, SQGs are not developed that would be applicable on a regional or national basis.

The major advantage of the SQTA is that it integrates the data generated from the three separate measurements and, thereby, facilitates the separation of natural variability in biotic characteristics from variability due to the toxic effects of contaminants. For example, variability in benthic community composition may be due to the presence of contaminants in sediments or it may be related to differences in other aspects of habitat quality (i.e., grain size, depth, etc.). The triad approach provides a basis for distinguishing between these effects; however, it cannot be used to establish cause and effect relationships. Other advantages of this approach are that it may be used for any measured contaminant, it may incorporate information on both acute and chronic effects, and it does not require information on specific processes governing interactions between organisms and toxic contaminants. Integrating the three data types provides a weight of evidence approach to guidelines development.

The major limitations of the SQTA are as follows (Chapman 1989): statistical criteria have not been developed for use with the triad; rigorous criteria for determining single indices for each of the separate measurements have not been developed; a large database is required; it is generally used to develop guidelines for single chemicals, and as such the results can be strongly influenced by the presence of unmeasured toxic contaminants that may or may not co-vary with the measured chemicals; sample collection, analysis, and interpretation is labor-intensive and costly; and, the choice of a reference site is often made without adequate information on how degraded the site may be. In addition, the SQTA does not explicitly consider the bioavailability of sediment-associated contaminants. Further, the SQTA mainly considers data from acute toxicity bioassays and, therefore, sub-acute and chronic effects may not be identified.

The SQTA was not initially intended to be a method for developing SQGs. Rather, the procedure was designed to be a practical tool to support site-specific sediment quality assessments. In assessments, the SQTA has been used to identify priority areas for remedial action, to determine the size of areas that require remedial action, to verify the quality of

reference sites, to determine contaminant concentrations that are always associated with effects on aquatic biota, and to describe ecological relationships between the characteristics of bottom sediments and biota that may be at risk (EPA 1989a). The SQTA has been used primarily in Puget Sound, but it has also been used in the Great Lakes, in Vancouver Harbour, in San Francisco Bay, and in the Gulf of Mexico (Chapman 1992).

3.7 Apparent Effects Threshold Approach (AETA)

The AETA for developing SQGs was developed by Tetra Tech Inc. (1986) for use in the Puget Sound area of Washington State. The AETA is based on relationships between measured concentrations of contaminants in sediments and observed biological effects, mainly on benthic organisms. The goal of this procedure is to define the concentration of a contaminant in sediment above which significant ($p \leq 0.05$) biological effects are ***always*** observed. These biological effects include, but are not limited to, toxicity to benthic and/or water column species (as measured using sediment toxicity bioassays), changes in the abundance of benthic invertebrate species, and changes in benthic invertebrate community structure.

The AETA is similar in many ways to the SLCA, since both rely on matching biological effects and sediment chemistry data. However, the AETA is more appropriate for the development of SQGs than the SLCA because it considers more diverse and sensitive measures of biological effects. The AET values are based on dry-weight-normalized contaminant concentrations for metals and either dry-weight or TOC normalized concentrations for organic substances (Barrick *et al*. 1988; Washington Department of Ecology 1990).

One of the principle advantages of the AETA is its capability to employ a wide variety of observations of biological effects from field surveys and the results of laboratory sediment toxicity bioassays. As such, AETs may be derived for each of the areas, species, and biological effects under consideration in an investigation. Like the SLCA, it can be used to develop guidelines for virtually any contaminant for which analytical methods are available. In Puget Sound, AETs were demonstrated to be relevant and precise tools for predicting biological effects associated with elevated levels of sediment-associated contaminants (Puget Sound Estuary Program 1988).

One of the major limitations of the AETA is its requirement for detailed site-specific information that relate concentrations of sediment-associated sediments to specific biological effects. This detailed database is currently available only for Puget Sound, some areas in California, several locations along the Atlantic coast, and the Great Lakes. Implementation of this approach in other areas, where these data are not available, would require an extensive data collection program.

Like the other approaches that rely on the analysis of matched sediment chemistry and biological effects data, the AETA does not provide definitive cause and effects relationships. Evaluation of the data is based on establishing associations between contaminant concentrations and biological effects. This characteristic of the approach results in uncertainty in the resultant SQGs.

Another disadvantage of the AETA is that there is a substantial risk of under-protection of biological resources if the AET is used directly as the SQG. The principle reason for this is that the AET defines the concentration of a contaminant above which biological effects are always observed. Unlike the other approaches to the development of SQGs, AETs can only *increase or remain the same* as new information is added to the database. This limitation may be minimized by defining AETs for each species tested and endpoint measured.

In addition to the potential to be under-protective, AETs may also be overly-protective of aquatic resources (i.e., overly restrictive) under some circumstances. This situation may occur when the substance under consideration consistently co-varies with other substances which are actually responsible for the observed effect. This situation is most likely to occur when AETs are generated using data from a specific geographic area in which the substance under consideration is present at each of the sites tested (e.g., DDT in Puget Sound).

This approach has been used extensively in Washington State by the Puget Sound Dredged Disposal Analysis Program for evaluating sediments that were to be dredged and disposed of by ocean dumping. In addition, AETs have been used to assess the effects of disposing of contaminated sediments at dump sites in that area (Puget Sound Dredged Disposal Analysis 1989). Recently, the Washington Department of Ecology (1990) established marine sediment management standards using the AETA. These legally-enforceable standards are designed to establish long-term goals for sediment quality, to manage discharges of toxic substances into coastal waters, and to provide a basis for identifying contaminated sites and appropriate cleanup levels.

Following a comprehensive evaluation, the SAB (Sediment Criteria Subcommittee 1989) indicated that the AETA is appropriate for deriving site-specific SQGs, such as the Puget Sound AETs. However, the SAB also recommended that the AETA should not be used to develop general, nationally applicable SQGs.

3.8 Weight of Evidence Approach (WEA)

The WEA for deriving SQGs (Long and Morgan 1990) was originally developed to provide informal tools to evaluate coastal sediment chemistry data collected under the National Status

and Trends Program (NSTP, NOAA). Long and Morgan (1990) compiled a database containing information generated by the three groups of approaches to the establishment of effects-based SQGs: the EqPA, the spiked-sediment toxicity approach, and various approaches that rely on the evaluation of matching sediment chemistry and biological effects data [i.e., co-occurrence approaches (AET, SLC, SQT)]. All of the information in the database was weighted equally, regardless of the method that was used to develop it.

Candidate data sets are screened to evaluate their applicability for incorporation into the database. This screening procedure is designed to evaluate the overall applicability of the data set (i.e., presence of matching sediment chemistry and biological effects data), the methods that were used, the type and magnitude of the end-point measured, and the degree of concordance between the chemical and biological data. Data which shows no concordance between chemical and biological variables is incorporated into the database, but not used in the statistical evaluation of the information (Long 1992).

The data which meets all screening criteria is incorporated into the database. Individual entries consists of the concentration of the contaminant, the type of biological response measured (usually specifying the location of the test as well), and an indication of whether or not there was concordance between the observed effect and the concentrations of a specific chemical (i.e., no effect, no or small gradient, no concordance, or a "hit", which indicated that an effect was measured). Data from non-toxic or unaffected samples are assumed to represent background conditions. Data points are identified for which a biological effect was observed in association with elevated chemical concentrations. These latter data points are sorted in ascending order of concentrations and the lower 10th and 50th percentile concentrations for each compound is determined. The effects range-low (ER-L; 10th percentile value) is considered to represent a lower threshold value, above which adverse effects on sensitive life stages and/or species began. The effects range-median (ER-M; 50th percentile value) is considered to represent a second threshold value, above which adverse effects on most species were frequently or always observed. These two parameters, ER-L and ER-M, are then used as informal SQGs (Long and MacDonald 1992).

One of the most important advantages of this approach is that it provides a weight of evidence from the available information for assessing sediment quality. In addition, it provides a framework for assessing sediment quality by organizing and summarizing data that relate concentrations of sediment-associated contaminants to specific biological effects. The other main advantages of this approach are that it can be applied with existing data (no additional field work or laboratory investigations are required), all of the available data generated in North America using the various approaches described above were compiled, and the database is expandable to encompass data collected in other jurisdictions. Further, the reliability (or degree of confidence) of each value can be determined by evaluating the agreement among the available data (Long *et al*. In press). Lastly, the approach facilitates the identification of ranges of contaminant concentrations which provide a means of determining the probability of observing adverse biological effects at a given contaminant concentration.

The main limitation of this approach concerns the quality and compatibility of the available data. In many cases, the data were generated using different analytical procedures in numerous laboratories and considered many species, endpoints, and locations across North America. For this reason, information on a wide variety of sediment types (i.e., with different particle sizes and concentrations of substances that could influence bioavailability) were combined, and this may have resulted in unknown biases. This amalgamation of the data may have resulted in the interpretation of responses as being attributable to a single contaminant when, in fact, synergistic and/or additive effects were actually driving the response. For substances for which only a moderate amount of data exists, or only acute toxicity data are represented (as is the case for many chemicals), it is possible that inappropriate guidelines could be derived. Furthermore, the compilation and evaluation of the data was very labor-intensive and required sound knowledge of sediment chemistry and biology.

The database evaluated in Long and Morgan (1990) consists of information generated at numerous locations around the United States. The authors felt that the degree of confidence in the ER-L and ER-M values should be considered moderate for metals and PCBs, and low for pesticides and PAHs. They felt that, although the compiled database was fairly extensive, much more data was needed to support or refute this approach for all groups of chemicals, for individual analytes within the groups, and for all types of sediments. Nonetheless, the informal guidelines have been used in numerous applications, ranging from contaminated site assessment to litigation.

3.9 Summary

A total of eight approaches for deriving numerical SQGs were investigated to identify an appropriate procedure for providing immediate guidance in Florida. The strengths and limitations of each of these approaches are summarized in Table 1. This summary indicates that no single approach is likely to support deriving SQGs under all circumstances. Therefore, each of these approaches are further evaluated to assess the degree to which they responded to Florida's requirements for sediment quality assessment guidelines (SQAGs). The results of this evaluation were used to develop a strategy for deriving numerical SQAGs for Florida coastal waters (Chapter 4).

Table 1. Summary of the strengths and limitations of approaches for deriving numerical sediment quality assessment guidelines.

Approach	Strengths	Limitations
SBA	▪ Sufficent data are generally available.	▪ Not based on biological effects.
SSBA	▪ Based on biological effects. ▪ Suitable for all classes of chemicals and most types of sediments. ▪ Supports cause and effect evaluations.	▪ Sufficient data are not generally available. ▪ Implementation costs are high. ▪ Spiking procedures are not yet standardized.
EqPA	▪ Based on biological effects. ▪ Suitable for all classes of chemicals and most types of sediments. ▪ Bioavailability is considered. ▪ EPA will support research to validate this approach. ▪ Supports cause and effect evaluations.	▪ Few sediment quality criteria are currently available. ▪ Water quality criteria are not available for some substances. ▪ In situ sediments are rarely at equilibrium.
TRA	▪ Simple to apply. ▪ Bioaccumulation is considered. ▪ A protocol for the derivation of TRGs is available.	▪ Tissue residue guidelines for wildlife are not yet available. ▪ In situ sediments are rarely at equilibrium.
SLCA	▪ Based on biological effects. ▪ Sufficient data are generally available. ▪ Suitable for all classes of chemicals and most types of sediments.	▪ Not possible to establish cause and effect relationships. ▪ Large database is required. ▪ End point used is insensitive. ▪ Bioavailability is not considered.

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Table 1. Summary of the strengths and limitations of approaches for deriving numerical sediment quality assessment guidelines. (continued).

Approach	Strengths	Limitations
SQTA	▪ Based on biological effects. ▪ Chemistry, bioassay and in situ biological effects are integrated. ▪ Provides a weight of evidence.	▪ Difficult to derive numerical SQAGs. ▪ Labour intensive and expensive. ▪ Statistical criteria for evaluating TRIAD have not been established. ▪ Extensive site-specific database is required. ▪ Not possible to establish cause and effect relationships. ▪ Bioavailability is not considered.
AETA	▪ Based on biological effects. ▪ All types of biological data are considered. ▪ Suitable for all classes of chemicals and most types of sediments.	▪ Extensive site-specific database is required. ▪ Not possible to establish cause and effect relationships. ▪ Risk of under- or over- protection of resource. ▪ Not applicable to the derivation of broadly applicable SQAGs. ▪ Bioavailability is not considered.
WEA	▪ Based on biological effects. ▪ All types of biological data are considered. ▪ Suitable for all classes of chemicals and most types of sediments. ▪ Provides a weight of evidence. ▪ Provides data summaries for evaluating sediment quality. ▪ May be implemented with existing data.	▪ Large database is required. ▪ Not possible to establish cause and effect relationships. ▪ Amalgamation of data from multiple sources could result in unknown biases in the database. ▪ Bioavailability is not considered.