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Development of an Approach to the Assessment of Sediment Quality in Florida Coastal Waters

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WORKSHOP DRAFT
May 31, 1992

Funds for this project are provided by the Florida Department of Environmental Regulation, Office of Coastal Management, using funds made available through the National Oceanic and Atmospheric Administration under the Coastal Zone Management Act of 1972, as amended.

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Executive Summary

This report was prepared to provide the Florida Department of Environmental Regulation (FDER) with guidance on the development of effects-based sediment quality guidelines (SQGs) for Florida coastal waters. As such, a variety of approaches to the derivation of numerical SQGs were reviewed and evaluated in light of Florida's unique requirements for sediment quality assessment guidelines. The results of this evaluation indicated that the approach recommended by Long and Morgan (1990; National Status and Trends Program Approach) would provide a practical basis for deriving SQGs in the near-term. Using this approach, preliminary SQGs for 25 priority substances in Florida coastal waters were derived and evaluated. These SQGs are designed to provide practical guidance in a number of sediment quality assessment applications, but they are not intended to be used as sediment quality criteria. While the guidelines are considered to be applicable to a variety of sediment types, they should be evaluated to determine their applicability in Florida sediments and refined as more information becomes available. A framework for assessing sediment quality has also been recommended that describes how effects-based SQGs can be used in conjunction with other assessment tools to support decisions on the management of coastal resources.

In Florida, conservation and protection of natural resources has been identified as a high priority environmental management goal. Realization of this goal requires protection of living resources and their habitats in estuarine, nearshore, and marine ecosystems. In the last decade, there has been a significant increase in the level of scientific understanding (and public recognition) of the important role sediments play in the functioning of coastal ecosystems. Sediments are particularly critical in determining the fate and effects of environmental contaminants.

Recent monitoring data indicate that concentrations of various contaminants are present at elevated levels at a number of locations in Florida coastal sediments. While these chemical data provide essential information on the nature and areal extent of contamination, they provide neither a measure of adverse biological effects nor an estimate of the potential for such effects. Therefore, effects-based SQGs are required to evaluate the potential for biological effects associated with sediment-sorbed contaminants and to provide assistance in managing coastal resources.

To identify an appropriate procedure for deriving SQGs, the major approaches used in other jurisdictions to derive SQGs were reviewed and evaluated in the context of Florida's unique requirements for sediment quality assessment values. The results of this analysis indicated that the National Status and Trends Program Approach (NSTPA; Long and Morgan 1990) would respond most directly to Florida's immediate need for reliable and cost-effective SQGs. Therefore, a strategy that relied on a modified version of the NSTPA was recommended to derive numerical SQGs that could be used immediately to assess sediment quality issues and concerns. A critical evaluation of this procedure suggested that, while this approach has limitations that could influence the applicability of the guidelines, it is likely to support the derivation of scientifically defensible preliminary guidelines for Florida coastal waters.

Using the recommended strategy, data derived from a wide variety of methods and approaches were assembled and evaluated to derive preliminary SQGs for 25 priority contaminants in Florida coastal waters. However, insufficient data were available to derive guidelines for another 29 substances that are known or are suspected to contaminate Florida coastal sediments. The numerical SQGs were used to define three ranges of concentrations for each of the 25 contaminants: a probable effects range; a possible effects range; and, a no effects range. These ranges of contaminant concentrations were considered to be more effective assessment tools than single numerical guideline values. A subjective assessment of the credibility of these guidelines indicated that a high level of confidence could be placed on the guidelines derived for 11 substances, and a moderate or low level of confidence could be placed on the guidelines for the remaining 14 substances. The results of this assessment suggest that the preliminary guidelines should be fully evaluated and refined, as necessary using the results of investigations conducted in Florida and elsewhere.

The preliminary SQGs were used to conduct an initial assessment to determine the nature, extent and severity of contamination in Florida coastal sediments. The potential for adverse biological effects associated with measured levels of sediment-sorbed contaminants was used as an index of contamination. This assessment was conducted with the Florida Department of Environmental Regulation (FDER) coastal sediment chemistry database to identify priority areas and priority substances with respect to sediment contamination. The results of this investigation are considered to be preliminary due to the limitations on the available data. Therefore, this database should be re-evaluated when the specific limitations identified in Chapter 7 have been addressed.

A total of 21 areas were considered in the initial assessment of sediment quality in Florida coastal waters. However, insufficient data were available to conduct a thorough assessment of sediment quality conditions in many of these areas, particularly for organic contaminants. In spite of these limitations, the St. Johns River in the vicinity of Jacksonville, the Miami River in Dade County, and Tampa Bay in the vicinity of Tampa/St. Petersburg were identified as the highest priority areas in terms of the extent and severity of sediment contamination. The contaminants of greatest concern in Florida sediment included copper, chromium, lead, mercury, zinc, phenanthrene, pyrene, Aroclor 1254, and total PCBs.

The recommended SQGs were developed specifically to support the identification of contaminated sites and priority chemicals of concern in Florida coastal waters. As such, these guidelines will contribute substantially to the design, implementation, and evaluation of sediment quality monitoring programs in the state. In addition, the recommended guidelines may also be used in a variety of environmental management applications, including identification of the need for further testing to support regulatory decisions and of identifying areas that might be considered for remedial action. Furthermore, SQGs provide a common basis for facilitating multi-jurisdictional agreements on sediment quality.

The preliminary guidelines were established to provide a yardstick for evaluating sediment quality in Florida. As such, these guidelines may be used to screen sediment chemistry data and establish priorities with respect to sediment quality management. However, they should not be used in lieu of water quality criteria, nor should they be used as sediment quality criteria. Ambient environmental conditions may influence the applicability of these guidelines at specific locations and, therefore, they should be applied with care in certain portions of the state.

The preliminary SQGs developed in the present study and the metals interpretive tool provide a consistent basis for evaluating sediment quality conditions in Florida coastal ecosystems. However, no such tools exist for use in freshwater ecosystems. Therefore, effects-based SQGs should be developed to evaluate the biological significance of contaminated sediments in freshwater systems. In addition, a procedure to determining the probable origin of sediment-sorbed metals in freshwater sediments is required.

Currently, there are a relatively large number of independent and loosely-related initiatives that are directed at the evaluation and management of contaminated sediments. While each of these programs are designed to advance our understanding of the nature, extent, and severity of sediment contamination, development of a regional strategy for contaminated sediment identification and management would accelerate this process. Therefore, a cooperative regional strategy should be developed by FDER, Environmental Protection Agency, the Army Corps of Engineers, and other affected agencies to identify priority sediment management and regulatory objectives, and the interagency efforts required to achieve them.

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Acknowledgements

It would be difficult to explicitly acknowledge all of the persons who contributed to the production of this document. However, the author would like to gratefully acknowledge those persons who made very substantial contributions to its preparation. Data and other pertinent information on the biological effects of sediment-sorbed contaminants was supplied by over 140 investigators across North America, including those from research institutes, universities, consulting firms, and state, provincial, and federal agencies. Each of these people deserve a special acknowledgement and the author's most sincere thanks. Preparation of this report would not have been possible without the expert guidance and advice provided throughout the course of this study by the Science Advisory Group on Assessing Sediment Quality. This group was comprised of Ed Long (National Oceanic and Atmospheric Administration), Chris Ingersoll (US Fish and Wildlife Service), Herb Windom (Skidaway Institute of Oceanography), Steve Schropp (Taylor Engineering), Fred Calder, Gail Sloane and Tom Seal (Florida Department of Environmental Regulation). Sherri Smith and Michael Wong (Environment Canada) also provided useful input on the derivation of sediment quality guidelines. In addition, the author would like to thank M.L. Haines, K. Brydges, B. Moore, M. Popadynec, and I.D. Cuthbert (MacDonald Environmental Sciences Limited) for their very significant contributions to the preparation of this document.

Chapter 1

Introduction

Public concerns relative to the quality of coastal waters have been aroused in recent years as a result of the information that has been disseminated on the quality of these systems. For example, Bolton *et al.* (1985) reported that environmental contamination in freshwater, estuarine, and marine ecosystems was widespread throughout North America. More recent data, collected under the National Status and Trends Program [NSTP; which is administered by the National Oceanic and Atmospheric Administration (NOAA)], indicates that while levels of contaminants, in general, have begun to decrease in coastal waters, high and biologically significant concentrations of many contaminants are present in urbanized estuaries throughout the United States (O'Connor 1990).

Traditionally, concerns relative to the management of aquatic resources in coastal waters have focused primarily on water quality. However, the importance of sediments in determining the fate and effects of a wide variety of contaminants has become more apparent in recent years (Long and Morgan 1990). Specifically, sediment quality is important because many toxic contaminants found in only trace amounts in water may accumulate to elevated levels in sediments. As such, sediments serve both as reservoirs and as sources of contaminants to the water column. In addition, sediments tend to integrate contaminant concentrations over time and sediment-sorbed contaminants have the potential to affect benthic and other sediment-associated organisms directly (Chapman 1989). Therefore, sediment quality data provide essential information for evaluating ambient environmental quality conditions in coastal waters.

Over the past 10 years, Florida Department of Environmental Regulation (FDER) and others have collected a substantial quantity of information on the chemical composition of Florida sediments. Preliminary assessment of these data indicates that numerous areas in Florida are contaminated by metals (such as lead, silver, and mercury) and organic substances (such as polycyclic aromatic hydrocarbons and pesticides). However, sediment chemistry data alone do not provide an adequate basis for identifying or managing potential sediment quality problems in the state. Biologically-based sediment quality assessment guidelines (SQAGs) are also required to interpret the significance of sediment chemistry data.

1.1 Purpose of the Report

The purpose of this report is to recommend a scientifically defensible framework for assessing the biological significance of sediment-associated contaminants. Numerical SQAGs represent an integral component of this framework, as they provide a basis for assessing the *potential* effects of sediment-associated contaminants. As such, a variety of approaches to the derivation of sediment quality assessment values were reviewed to identify those that would be applicable to Florida coastal conditions. The results of this review indicate that each of these approaches has a number of deficiencies which limit its direct application in Florida. For this reason, an integrated strategy for the derivation of numerical SQAGs is recommended for the state of Florida. The recommended strategy is designed to provide relevant assessment tools in the near-term and provide a basis for refining these guidelines as the necessary data become available.

Using the recommended approach, numerical SQAGs have been developed for Florida coastal waters. These guidelines were derived using information from numerous investigations of sediment quality conducted throughout North America and, as such, are based on a weight-of-evidence regarding the relationships between contaminant concentrations and adverse biological effects. In this respect, the guidelines represent a cost-effective response to a practical need for assessment tools. However, these guidelines are considered to be preliminary in nature and are likely to be revised or refined depending on the results of field validation and other related studies conducted in Florida and elsewhere in North America.

1.2 Description of the Recommended Approach to The Derivation of Numerical Sediment Quality Assessment Guidelines

The recommended approach to the derivation of numerical SQAGs is described in Chapter 4. This approach to the derivation of sediment quality guidelines (SQGs) is considered to be the most practical for use in Florida because:

- It can be implemented in the near-term;
- It can be implemented using existing data;
- It will provide a weight of evidence from numerous biological effects-based approaches for determining associations between chemical quality and biological effects;
- It will provide assessment tools or guidelines that define ***ranges of contaminant concentrations*** that could be used to evaluate sediment quality data. Specifically these guidelines define ranges of concentrations that have usually or always, frequently, and rarely or never been associated with adverse biological effects. These ranges are considered to be more practical than single values for assessing sediment quality in the diverse conditions found along Florida's extensive coast;

- It will provide summaries of the data that were used to derive the assessment guidelines. These summaries are useful for evaluating the biological significance of contaminant concentrations within these ranges; and,
- It will have long-term applicability in Florida and can be verified and refined with additional data, particularly with data from the southeast.

A detailed discussion of the strengths of this approach is provided in Section 5.3.

Sediment quality guidelines derived using the recommended approach are considered to be preliminary values and should be refined as new information becomes available. Several limitations and considerations in using this approach have been identified, including:

- The approach is designed to determine the potential for sediment-sorbed contaminants to induce biological effects. Direct cause and effect relationships should not be inferred when comparing chemical data to the recommended guidelines;
- The SQGs are applicable to marine and estuarine waters only; they are not applicable to freshwater systems;
- The SQGs are not expressed in terms of the factors that are thought to control the bioavailability of sediment-associated contaminants [i.e., total organic carbon (TOC) for non-polar organics and acid volatile sulfide (AVS) for divalent metals];
- The data that have been used to derive the SQGs consist primarily of the results of acute toxicity studies; few data exist on the chronic responses of aquatic organisms to contaminants that are associated with sediments;
- The recommended guidelines should be used in conjunction with other assessment tools and protocols, such as the metals interpretive tool (Schropp and Windom 1988) and the Green Book (EPA and ACE 1991) to provide comprehensive evaluations of sediment quality; and,
- The recommended guidelines were developed using information from a variety of locations in North America. It is uncertain if these data are representative of the wide range of sediment types that are present in Florida. For this reason, caution should be exercised in utilizing these guidelines, particularly in carbonate-dominated sediments.

A discussion of these limitations and considerations is provided in Section 5.3.

1.3 Applications of the Recommended Sediment Quality Assessment Guidelines

The recommended sediment quality assessment strategy is intended to provide a consistent basis for evaluating sediment quality in Florida. While the SQAGs represent an integral element of this strategy, they should be used in conjunction with other assessment tools to efficiently and cost-effectively evaluate ambient sediment quality conditions. In this context, these SQAGs may be used to:

- Interpret the results of sediment quality monitoring data. In this context, SQAGs may be used to assess the adverse biological effects that could, potentially, be associated with specific concentrations of sediment-sorbed contaminants;
- Support the design of sediment quality monitoring programs. In this context, SQAGs may be used to evaluate existing sediment chemistry data, and rank areas of concern and chemicals of concern in terms of their potential to be associated with adverse biological effects. As such, monitoring priorities may be more clearly and effectively identified;
- Identify the need for site-specific investigations to support regulatory decisions, including source control and other remedial measures. In this context, SQAGs may be used to evaluate existing data and to determine if additional testing (e.g., sediment toxicity bioassays, etc.) is needed to support regulatory decisions;
- Evaluate the hazards associated with increased levels of contaminants at specific sites. In this context, SQAGs may be used as early-warning tools to identify the need for regulatory action before contaminant levels become problematic;
- Support a preliminary assessment of the applicability of the sediment quality criteria currently under development by USEPA. In this context, the SQAGs may be used to assess the level of protection afforded to aquatic organisms by these criteria; and,
- Facilitate multi-jurisdictional agreements on sediment quality issues and concerns. In this context, SQAGs may be used to establish site-specific sediment quality objectives that will help define the responsibilities of various levels of government in preventing and remediating sediment contamination.

These guidelines were established to provide a consistent basis for evaluating sediment quality in Florida. However, these guidelines are preliminary and, as such, have certain limitations on their application. Therefore, SQAGs:

- Should not be used in lieu of water quality criteria. However, these guidelines may be used in regulatory programs to evaluate their effectiveness and identify the need for more stringent regulations;
- Should not be used to define uniform values for sediment quality on a statewide basis (i.e., they should not be used as sediment quality criteria). Ambient environmental conditions may influence the applicability of these guidelines at specific locations.
- Should not be used as criteria for the disposal of dredged materials;
- Should not be used directly as numerical clean-up levels at severely contaminated sites (e.g., Superfund sites); and,
- Should not be used instead of biological tests in evaluating sediment quality.

There are a number of initiatives that are underway or under development in Florida and elsewhere in the United States that will provide relevant data for revising and refining these preliminary guidelines. These initiatives include spiked sediment bioassays, field surveys of sediment toxicity, and the development of sediment quality criteria that explicitly consider the bioavailability of sediment-sorbed contaminants. In the long-term, refinement of the guidelines will provide a means of ensuring broader applicability and utility within the state.

Chapter 2

Florida's Coast: A National Treasure

2.0 Introduction

Of all the states and provinces in continental North America, Florida is the most intimately linked with the sea. The entire state lies within the coastal plain, with a maximum elevation of about 120 meters above sea level, and no part of the state is more than 100 km from the Atlantic Ocean or the Gulf of Mexico (Webb 1990). With the exception of Alaska, Florida has the longest coastline of any state in the United States, with open estuaries and tidal wetlands that cover vast areas (Livingston 1990). These unique characteristics shape Florida's environmental identity and underscore the importance of employing relevant tools in coastal protection decision-making processes.

The State of Florida relies on its coastal waters to provide a variety of economic and social benefits to state residents and visitors, alike. Coastal ecosystems in Florida (including marine, near-shore, and estuarine environments) support a variety of sport and commercial fisheries which contribute significantly to the state economy. Indeed, Florida ranks as one of the leading commercial fishing states in terms of the value of its annual fish catch, with shrimp, lobsters, and scallops being the most important fisheries. Marine environments within the state also provide essential transportation links, support a variety of water-dependent facilities, and offer a diverse array of unique recreational opportunities that attract millions of visitors to the state each year.

2.1 Physical Features of Florida's Coast

Florida has one of the most extensive coastlines in the United States. The marine coastline in the state spans almost 2,200 km, with a tidal shoreline that covers over 13,000 km (NOAA 1975). Florida's coastal systems are unique because this combination of climatological and physiographic features occur nowhere else in the world. Livingston (1990) suggested that essentially all of the inshore marine habitats in the state could be classified as estuarine, primarily due to the prevalent influence of upland runoff in these areas. The Florida coastline is characterized by a variety of major embayments, marsh and mangrove systems that directly front the sea, and by numerous, partially enclosed, brackish water basins (Comp and Seaman 1985). A diversity of natural habitats are found within these areas, including seagrass beds, tidal flats, tidal marshes, soft sediments, hard substrates, shellfish beds, and a variety of transitional zones (Livingston 1990).

The Atlantic coast of Florida, from the St. Mary's River to Biscayne Bay (560 km), is characterized by a high energy shoreline with long stretches of continuous barrier islands

3.2 Spiked-Sediment Bioassay Approach (SSBA)

This approach to generating SQGs relies on empirically generated information on the responses of test organisms to specific contaminant challenges under laboratory conditions. In this procedure, clean sediments are spiked with known concentrations of contaminants to establish definitive cause and effect relationships between chemicals and biological responses (i.e., mortality, reductions in growth or reproduction, physiological changes, etc.). Chemicals can be tested alone or in combination to determine the effects of various concentrations of contaminants in sediment. Numerical SQGs may be derived using this approach by applying a safety factor to the lowest observed effect level (Smith and MacDonald 1992) or by using other appropriate means.

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, and can account for factors that control the bioavailability of these substances, such as total organic carbon and acid volatile sulphide. Application of this procedure facilitates unequivocal determination of causal effects (EPA 1990). As such, guidelines derived using spiked-sediment bioassay data are highly defensible.

The major disadvantage associated with the implementation of this method for deriving SQGs for Florida is that spiked-sediment bioassays have only been conducted on a few species with only a limited number of substances (i.e., cadmium, copper, a few pesticides, and a number of 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 limit the interpretation of the results of spiked-sediment bioassays.

The SBBA 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.* 1986, 1988, 1989). Environment Canada has recently developed a formal protocol for developing SQGs from the results of spiked-sediment bioassays (MacDonald and Smith 1991). This procedure is currently under review and is scheduled for implementation in 1992 (Smith and MacDonald 1992).

In addition to its role in the derivation of numerical SQGs, data developed using this approach are fundamental for evaluating the applicability of guidelines that have been developed using other approaches. For example, EPA (1992) used spiked-sediment bioassay data to evaluate the applicability of the sediment quality criteria that have been developed for fluoranthene. Likewise, Outridge *et al.* (1992) evaluated the applicability of SQGs for cadmium derived using the weight-of-evidence approach (Smith and MacDonald 1992) with data from spiked-sediment bioassays.

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., one standard deviation 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 in the derivation of guidelines. In addition, this approach applies primarily 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. While SQGs may be established at contemporary background levels, it is not clear whether or not these guidelines would be 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 Florida Department of Environmental Regulation (Schropp *et al.* 1990) and others (e.g., Loring 1991) have developed unique applications of the sediment background approach which improve its overall utility. 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 on 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.

3.3 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 and 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 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:

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

where:

SQG = Sediment quality guideline (in μg/kg)
Kp = Partition coefficient for the chemical (in L/kg)
FCV = Final Chronic Value (FCV; in μg/L)

Currently, this procedure is considered to be appropriate for deriving SQC for non-ionic organic substances, such as polycyclic aromatic hydrocarbons; polychlorinated benzenes, biphenyls, dioxins, and furans; and most pesticides (EPA 1991). For these substances, total organic carbon (TOC) normalization appears to provide a reliable basis for predicting toxicity to aquatic organisms (Swartz *et al.* 1990). In addition, the role of acid volatile sulfide (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 this class of chemical 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 control the bioavailability of sediment-sorbed 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 SQC. 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 SQC. Specifically, SQC developed using the EqPA do not explicitly address possible synergistic, antagonistic or additive effects of contaminants. In addition, the technical basis for developing sediment quality criteria for metals is still under development. Further, the interim sediment quality criteria for non-ionic chemicals apply only to sediments that have significant organic carbon contents (≥ 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).

Other disadvantages of the EqPA are related to limitations on the availability of water quality criteria (i.e., FAVs and FCVs) for some substances and of reliable partition coefficients for many priority contaminants. While water quality criteria exist for many contaminants, criteria for several important substances (e.g., dioxins and furans) are currently not available. In addition, application of the interim sediment quality criteria has been restricted by uncertainty in the estimates of partition coefficients for certain substances. 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 SQC is derived using these data. Further, *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 SQC developed using the EqPA (Sediment Criteria Subcommittee 1989).

Nonetheless the EqPA has been selected by the EPA as a primary basis for deriving sediment quality assessment values. As such, 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 EqPA is scheduled for a subsequent review sometime in 1992. It is anticipated that this presentation to the SAB will focus on the aggressive field validation program and the formalized framework for the application of the SQC that have been or are currently being developed (EPA 1991). 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.* 1991), 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 safe sediment concentrations for individual chemicals or classes of chemicals by determining the chemical concentrations in sediments that are predicted 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).

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 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). Therefore, SQGs must 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 (e.g., marine mammals with high daily consumption rates of aquatic organisms) may not be adequately protected. Recently, a protocol for the derivation of numerical tissue residue guidelines for the protection of wildlife has been developed (MacDonald and Walker 1992) and tissue residue guidelines for dioxins and furans are currently being derived (MacDonald *et al.* In preparation).

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). The applicability of this approach to the derivation of 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 EqPA described previously.

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 utilizes 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 currently available. Furthermore, SLCs are based on specific effects on a variety of 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 heavily on a number of 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 considered explicitly. 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 almost always 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, collected simultaneously. 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 SQC 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 utilized 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 as a tool 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 the development of SQGs (Chapman 1986). The SQTA to the development of SQGs 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 relative priority of the areas that have been surveyed. Also, they can be used to formulate site-specific sediment quality objectives. A procedure has not yet been proposed for developing SQGs that would be applicable on a regional or national basis.

The major advantage of the sediment quality triad approach is that it integrates the data generated from the three separate measurements, and thereby, facilitates the differentiation of the natural variability in biotic characteristics from the variability due to the toxic effects of environmental 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). The triad approach provides a basis for distinguishing these effects; however, it cannot be used to establish cause and effect relationships. The other advantages of this approach are that it may be used for any measured contaminant, it may include both acute and chronic effects, and it does not require information on the specific mechanisms of interaction between organisms and toxic contaminants. The integration of 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 labour-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-sorbed 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 specific assessments of sediment quality. In this context, the SQTA has been used to identify priority areas for remedial action, to determine the size of the 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 sediment quality triad approach has been used primarily in Puget Sound, but it has been also used in the Great Lakes, in Vancouver Harbour, in San Francisco Bay, and in the Gulf of Mexico.

3.7 Apparent Effects Threshold Approach (AETA)

The AETA to the development of 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 a contaminant in sediments and observed biological effects, mainly on benthic organisms. The practical 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 various species, and changes in benthic 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 may be more appropriate for the development of SQGs than the SLCA because it considers 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 total organic carbon normalized concentrations for organic substances (Barrick *et al.* 1988; Washington Department of Ecology 1990a).

One of the principle advantages of the AETA is associated with its capability to utilize a wide variety of observations of biological effects from field surveys and the results of sediment toxicity bioassays conducted in the laboratory. As such, AETs may be derived for each of the areas, species, and biological effects that have been considered in an investigation. Like the SLCA, it can be used to develop guidelines for virtually any contaminant for which analytical methods are currently available. In Puget Sound, AETs have been demonstrated to provide relevant and precise tools for predicting the biological effects that are associated with elevated levels of sediment-sorbed contaminants.

One of the major limitations of the AETA is its requirement for detailed site-specific information with which to relate concentrations of sediment-sorbed sediments to specific biological effects. This type of 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 some 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 because 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 characteristic of the AETA increases the risk of under-protecting aquatic resources. 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 the evaluation of sediments that were to be dredged and disposed of by ocean dumping. In addition, AETs have been used to assess the effects of the disposal of contaminated sediments at dumps site 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 inputs of toxic substances into coastal waters, and to provide a basis for identifying contaminated sites and appropriate cleanup levels.

Following a comprehensive evaluation, the Science Advisory Board (SAB; Sediment Criteria Subcommittee 1989) indicated that the AETA is relevant and appropriate for the derivation of 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 National Status and Trends Program Approach (NSTPA)

The NSTPA to the derivation of SQGs (Long and Morgan 1990) was developed to provide informal tools to assess the potential for biological effects of sediment-sorbed contaminants tested in 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. The objective of this assessment was to identify informal guidelines with which to evaluate coastal sediment chemistry data collected nationwide under the NSTP.

Candidate data sets were screened to evaluate their applicability for incorporation into the database. This screening procedure was 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 showed no concordance between chemical and biological variables were incorporated into the database, but were not used in the statistical evaluation of the information.

The data which passed the screening tests were incorporated into the database. Individual entries consisted 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 were assumed to represent background conditions. Data points were identified for which a biological effect was observed in association with elevated chemical concentrations. These latter data points were sorted in ascending order of concentrations and the lower 10th and 50th percentile concentrations for each compound were determined. The effects range-low (ER-L; 10th percentile value) was 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) was 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, were then used as informal SQGs.

One of the most important advantages of NSTPA is that it provides a weight of evidence approach to the assessment of sediment quality. In addition, it provides a framework for assessing sediment quality by providing summaries of the data that relate concentrations of sediment-sorbed contaminants to specific biological effects. The other main advantages of this approach are that it can be employed with existing data (no additional field work or laboratory investigations are required), all of the available data generated in the United States using the various approaches described above were compiled, and the database is expandable to encompass data that have been collected in other jurisdictions. Further, the accuracy (or degree of confidence) of each value could be identified based on an evaluation of the agreement among the available data. 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 is associated with 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 the United States. For this reason, information on a wide variety of sediment types (i.e., with different particle sizes and concentrations of substances that 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 labour-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.

3.9 Summary

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

Table 1. Summary of the strengths and limitations of the various approaches to the derivation of numerical sediment quality 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 coasts 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

Table 1. Summary of the strengths and limitations of the various approaches to the derivation of numerical sediment quality 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 SQGs. 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 SQGs. Bioavailability is not considered.
NSTPA	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.

Chapter 4

A Recommended Approach for Deriving and Validating Effects-Based Sediment Quality Guidelines in Florida

4.0 Introduction

The results of monitoring activities conducted in estuarine and coastal marine areas (FDER in preparation; Delfino *et al.* 1991; Long and Morgan 1990; Long *et al.* 1991) indicate that concentrations of sediment-sorbed contaminants are elevated at a number of locations throughout Florida. Techniques currently exist to determine the probable origin of many of these substances (i.e., natural vs. anthropogenic; Schropp and Windom 1988; Schropp *et al.* 1989; Schropp *et al.* 1990), however additional information is required to evaluate the potential biological effects of these contaminants. Therefore, effects-based sediment quality guidelines (SQGs) are also required to support the identification of issues and concerns relative to contaminated sediments in Florida.

To date, no effects-based SQGs have been developed which are known to apply directly to conditions in Florida. While effects-based SQGs have been developed specifically for a few regions of the country (i.e., in Puget Sound using apparent effects threshold approach; AETA), the EPA Science Advisory Board (SAB) has cautioned against using these guidelines outside the areas for which they were developed (Sediment Criteria Subcommittee 1989). The SAB has also questioned the validity of the sediment quality criteria that are currently under development by EPA (i.e., using the equilibrium partioning approach; EqPA). These evaluations by the SAB suggest that the SQGs that are under development in other jurisdictions are not likely to address Florida's immediate requirements for sediment assessment tools.

There is a pressing need for sediment quality assessment guidelines (SQAGs) to support environmental management decisions in Florida's coastal areas. In the absence of national or regional guidelines that could be adopted directly or adapted for use in Florida, new effects-based SQAGs must be developed. The following discussion provides an overview of the recommended strategy for deriving and validating numerical SQGs for Florida coastal waters and the rationale behind its selection.

4.1 Considerations for Recommending a Strategy for Deriving Sediment Quality Guidelines for Florida Coastal Waters

A total of eight approaches to the derivation of numerical SQGs were identified and reviewed in Chapter 3. However, selection of an appropriate procedure for deriving guidelines for Florida coastal waters necessitates further evaluation of each of the approaches in light of the state's specific needs. As such, a number of criteria were established to provide an objective basis for evaluating the candidate approaches and selecting a relevant strategy for deriving these guidelines (Table 2). The primary considerations in the selection of the recommended strategy were related to practicality, cost-effectiveness, scientifically defensibility, and broad applicability to the assessment of sediment quality. Each of these factors are discussed below.

Practicality is one of the central considerations with respect to the development of SQGs. Numerical SQGs must be functional (i.e., easy to use) and understandable if they are to be useful for assessing environmental quality. In addition, the immediate need for these assessment tools necessitates selection of an approach that can be implemented quickly.

In Florida, limited resources are available to support the development and implementation of SQGs. Financial and personnel limitations placed on the current initiative make collection of a significant quantity of additional data improbable. Therefore, the approach must be able to develop numerical SQGs with the data that are currently available. In addition, it must be amenable to re-evaluation as new data become available.

For SQGs to be effective in Florida, they must be effects-based (i.e., consider biological effects) and scientifically defensible. Key evaluation criteria for assessing the various approaches include their potential to consider the factors that control the bioavailability of sediment-sorbed contaminants, to establish cause and effect relationships, and to apply to all classes of chemicals and mixtures of contaminants that are expected to occur in Florida. In addition, they must be compatible with other interpretive tools, such as the metals interpretive tool that has already been developed by FDER. Furthermore, it is desirable for candidate approaches to be able to explicitly consider data from Florida and elsewhere in the southeastern United States and provide a means of accounting for site-specific environmental conditions.

Due to the inherent uncertainty associated with each of the candidate approaches, it would be advantageous if the guidelines supported the identification of ranges of contaminant concentrations which are predicted to be associated with specific biological effects. That is, the guidelines should identify ranges of contaminant concentrations that have high, moderate, and low probabilities of being associated with adverse biological effects. The guidelines should also be supported by a weight of evidence provided by the available data.

To be applicable to Florida, SQAGs must address the specific needs of the agencies that are charged with managing environmental quality. For example, SQGs should be relevant to the design, implementation, and evaluation of environmental quality monitoring programs by contributing to the identification of the contaminants and sites that are likely to be

Table 2. Evaluation of the approaches to the derivation of sediment quality guidelines.

Evaluation Criteria	SBA	SSTA	EqPA	TRA	SLCA	SQTA	AETA	NSTPA
Practicality								
Supports development of numerical SQGs?	Y	Y	Y	Y	Y	Y	Y	Y
Feasible to implement in the near term?	Y	N	Y/N	N	Y/N	N	N	Y
Cost Effectiveness								
Expensive to implement?	N	Y	N	Y	Y	Y	Y	N
Requires generation of new data?	N	Y	N	Y	Y	Y	Y	N
Scientific Defensibility								
Considers bioavailability?	N	Y	Y	Y	N	N	Y/N	Y/N
Provides cause and effect relationships?	N	Y	Y	N	Y/N	Y/N	Y/N	Y/N
Based on biological effects data?	N	Y	Y	Y/N	Y	Y	Y	Y
Considers data from South East?	Y	N	N	N	N	N	N	Y/N
Provides weight of evidence?	N	Y/N	Y/N	N	N	Y	Y	Y
Support definition of ranges of concentrations rather than absolute assessment values	N	N	N	N	N	N	N	Y
Considers mixtures of contaminants?	N	N	N	N	Y	Y	Y	Y
Requires field validation?	Y	Y	Y	Y	Y	Y	Y	Y
Considers site-specific conditions?	Y/N	Y/N	N	N	N	Y	Y	N
Applicable to all classes of chemicals?	Y	N	Y	Y	Y	Y	Y	Y
Applicability								
Supports monitoring programs?	Y/N	Y	Y	Y	Y	Y	Y	Y
Supports problem identification?	Y/N	Y	Y	Y/N	Y	Y	Y	Y
Supports regulatory programs?	N	Y	Y/N	N	Y/N	Y/N	Y	Y/N
Overall assessment	*	***	****	**	**	***	***	****

* = poor; ** = fair; *** = good; **** = excellent

associated with adverse biological effects. This would help to identify the need for further investigations at sites with concentrations of specific contaminants that exceed the SQAGs. Guidelines should also support the identification of areas that are most in need of remediation; however, they would not necessarily be used to establish clean-up levels. Furthermore, guidelines should contribute to regulatory programs by helping to evaluate source control measures and/or the need for further biological and chemical testing to support regulatory decisions.

4.2 A Recommended Strategy for Deriving Numerical Sediment Quality Assessment Guidelines for Florida Coastal Waters

Ideally, SQGs should be developed from detailed dose-response data which describe the acute and chronic toxicity of individual contaminants to sensitive life stages of resident species of aquatic organisms. These data should be generated in controlled laboratory studies, in which the influences of important environmental variables (such as TOC, AVS, salinity, and others) are identified and quantified and compared to the values predicted by appropriate models (e.g., EqP models). Finally, the results of these studies should then be validated in field trials to ensure that any guidelines derived from these data are applicable to a broad range of locations. A detailed understanding of the factors that influence toxicity would also support site-specific sediment quality assessments by providing a basis for evaluating the applicability of the preliminary guidelines and, if necessary, modifying the guidelines.

Unfortunately, insufficient data are currently available to support the derivation of numerical SQAGs using the ideal approach. Currently, only a limited number of controlled laboratory studies (i.e., spiked-sediment bioassays) have been conducted to assess the effects of sediment-sorbed contaminants on estuarine and marine organisms (Long and Morgan 1990). However, in spite of this obvious limitation, other types of data are routinely collected which contribute to our understanding of the toxic effects of these contaminants. Specifically, a wide variety of whole sediment toxicity tests have been conducted to assess the biological significance of concentrations of contaminants in sediments from specific geographic locations. These toxicity tests include those performed on benthic organisms (bivalve mollusks, shrimp, amphipods, polychaetes, nematodes, chironomids and other arthropods, etc.) and on pelagic organisms [*Daphnia*, oyster larvae, luminescent bacteria (Microtox), etc.]. Furthermore, numerous field studies have been conducted to assess the diversity and abundance of benthic infaunal species (bivalve mollusks, arthropods, amphipods, etc.) and epibenthic organisms (echinoderms, crustaceans, etc.). For many of these studies, matching data on the concentrations of contaminants in these sediments have also been collected. Studies which report matching sediment chemistry and biological effects data provide information which is highly relevant to the SQGs derivation process.

In recommending a suitable strategy for the derivation of SQAGs for Florida, it is important to explicitly recognize the limitations of the existing database for evaluating the potential biological effects of sediment-sorbed contaminants. In addition, the strategy must address both the immediate requirement for defensible SQAGs and the long-term requirement for increased reliability and applicability of these guidelines (i.e., guidelines that account for the environmental characteristics that influence the bioavailability of sediment-sorbed contaminants).

Evaluation of each of the approaches to the derivation of SQGs in the context of the specific requirements for the Florida coast (as expressed in Section 4.1) indicates that no single approach is likely to satisfy all of the immediate and long-term requirements for SQAGs (Table 2). For this reason, a strategy is recommended that places a priority on the immediate need for defensible SQAGs, while providing a framework for the revision or refinement of these values as the necessary data become available (Figure 1).

The National Status and Trends Program Approach (NSTPA; Long and Morgan 1990; Long 1992) provides a pragmatic means of generating scientifically defensible guidelines using data which are currently available. As such, this approach facilitates the immediate generation of preliminary SQAGs. However, several modifications (which are described in Chapter 5) to this approach are recommended to increase the applicability of the NSTPA to Florida. These modifications are designed to increase the quantity and suitability of data used to evaluate the biological significance of sediment-sorbed contaminants (i.e., to incorporate data from Florida and other southeastern areas and recent data from elsewhere in North America). In addition, the arithmetic procedure for deriving the guidelines has been refined to consider data from relatively uncontaminated areas. A detailed description and evaluation of the modified NSTPA to the derivation of SQGs (hereafter referred to as the Weight-Of-Evidence Approach; WEA) is provided in Chapter 5.

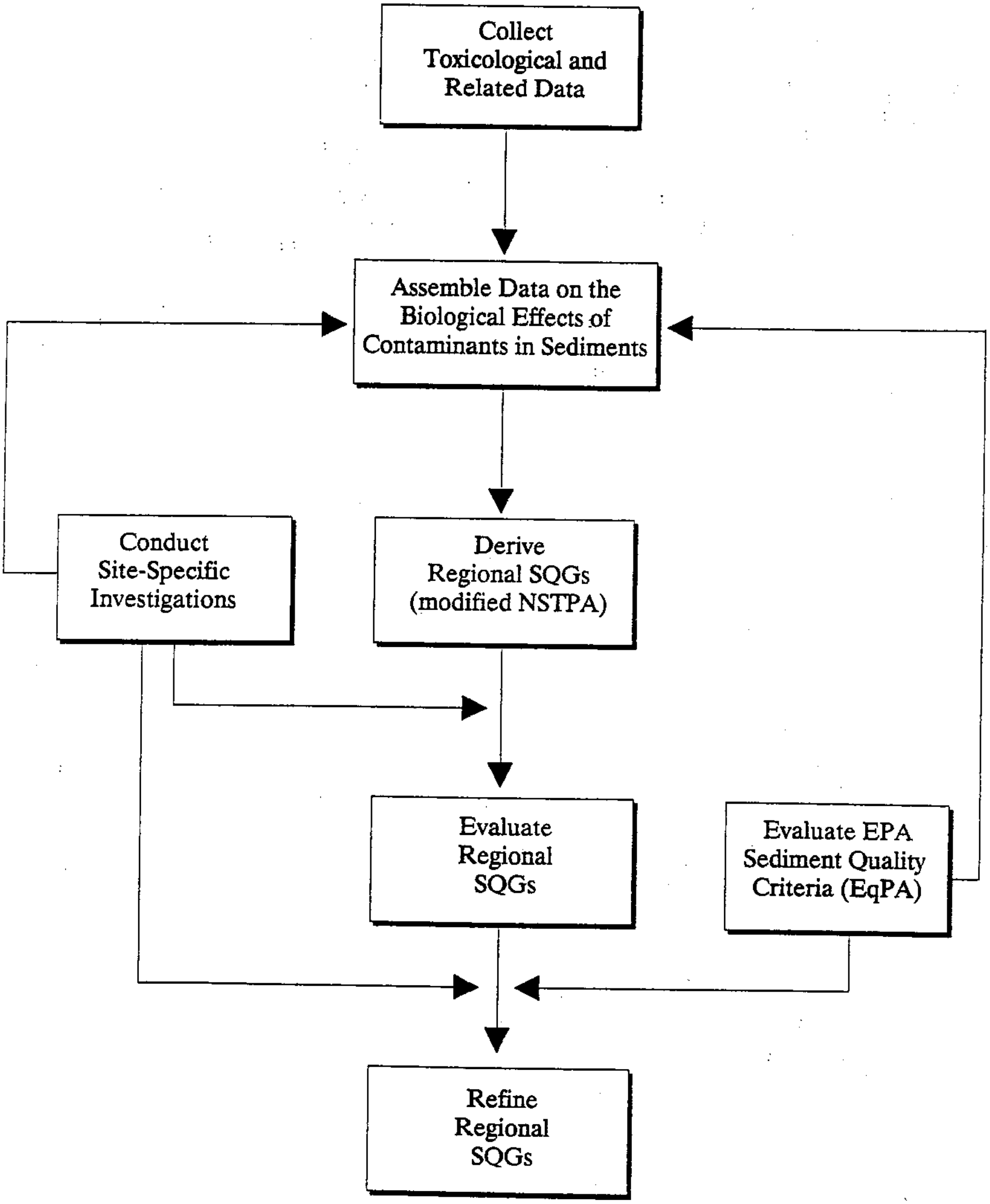
The preliminary SQGs, derived using the WEA, will address Florida's immediate need for effects-based tools for assessing environmental quality. In addition to these guidelines, the sediment quality criteria that are currently under development by EPA may provide further guidance for identifying and managing contaminated sediments. As such, the EPA criteria should be fully evaluated to determine how they could contribute to the assessment and management of coastal sediment quality in Florida. In addition, EPA should be encouraged to conduct field validation studies to determine if the criteria apply directly to the types of sediment that occur in Florida coastal waters.

4.3 Verification and Refinement of Sediment Quality Guidelines

Evaluation of the eight candidate approaches (see Chapter 3) suggests that guidelines derived using the WEA are likely to provide useful tools for assessing the quality of coastal sediments. However, the direct applicability of these guidelines to Florida coastal waters is uncertain. Therefore, additional data will be required to evaluate the applicability of, and if necessary, refine the guidelines for consistent use in Florida.

Field validation of SQGs derived using the WEA will require several types of data, which may be obtained from a variety of sources. First, data from spiked-sediment bioassays are

Figure 1. An overview of the recommended process for deriving numerical sediment quality guidelines in Florida.



required to determine how contaminants behave in different types of sediments. Ideally, these data would be generated in studies that investigate the toxicity of various substances in several types of Florida sediment (ranging from biogenically-derived to terrigenous sediments). Second, data from field studies conducted in locations with strong gradients in the concentrations of individual contaminants or classes of contaminants in sediments are required. These studies would include investigations of the toxicity of bulk sediments to resident species and of the benthic community characteristics at these sites. Both of these latter investigations would benefit from toxicity identification evaluations to identify the contaminant(s) that are responsible for any observed effects (Ankley 1989).

Florida Department of Environmental Regulation (FDER) recognizes the importance of validating the preliminary SQGs and has initiated investigations to obtain the required information. For example, an initial survey of sediment toxicity in Tampa Bay was conducted in 1991, in cooperation with NOAA. A second survey is scheduled for implementation in 1992. The Department has also designed a number of companion investigations (e.g., spiked-sediment bioassays and benthic invertebrate community evaluations) which may be implemented in cooperation with NOAA and EPA.

In addition to FDER initiatives, there are several other potential sources of data for validating the preliminary SQGs. For example, EPA is currently developing national sediment quality criteria for priority contaminants using the EqPA. Interim sediment quality criteria for numerous substances have been developed using this approach (Bolton *et al.* 1985; Lyman *et al.* 1987; Pavlou 1987; Pavlou *et al.* 1987). However, due to the uncertainty associated with the estimates of partitioning coefficients (K_{ow} and K_{oc}) and the applicability of interim criteria, EPA is planning to conduct an extensive research program to validate these criteria in field and laboratory trials. Similar research is being conducted by various researchers throughout the country. Together, these studies will provide much of the data necessary for evaluating the applicability of the preliminary SQGs, and for modifying the SQGs if necessary.

A variety of refinements to the preliminary SQGs are possible, depending on the results of field validation studies. One of the most likely refinements will involve expression in the guidelines in terms of factor(s) that are *demonstrated* to influence the toxicity (i.e., bioavailability) of the substance under consideration. For example, guidelines for non-ionic organic chemicals are likely to be expressed in terms of sediment TOC content, while guidelines for some metals may be expressed in terms of AVS content or some other normalizing factor. Verification and refinement of the preliminary SQGs will significantly increase confidence in their applicability and enhance their role in the sediment quality assessment process.

Chapter 5

Derivation of Numerical Sediment Quality Guidelines for Florida Coastal Waters Using the Weight-Of-Evidence Approach

5.0 Introduction

The National Status and Trends Program Approach (NSTPA; Long and Morgan 1990) has been identified as a central component of the immediate and long-term strategies for the development of sediment quality assessment guidelines (SQAGs) for Florida coastal waters. This approach relies on the collection, evaluation, collation and analysis of data from a wide variety of sources in the United States to establish relationships between concentrations of sediment-sorbed contaminants and the potential for adverse biological effects. A modified version of the NSTPA (termed the weight-of-evidence approach; WEA) is recommended for deriving numerical sediment quality guidelines (SQGs) in the near-term. In the longer-term, the applicability of the preliminary guidelines to Florida coastal sediments should be evaluated through the implementation of a well-designed field validation program.

5.1 Modification of the National Status and Trends Program Approach for Use in the Derivation of Sediment Quality Guidelines for Florida

The WEA was selected to derive preliminary SQGs due to its practicality for developing guidelines quickly, its limited requirement for additional resources, its overall scientific defensibility, and its applicability to all aspects of sediment quality assessment. This approach is closely related to the NSTPA, however, a number of modifications were implemented to increase the relevance of the resultant guidelines to Florida coastal sediments. Specifically, the modifications to the NSTPA are designed to increase the level of internal consistency in the database (by establishing additional screening criteria), to verify and expand the information contained in the original NSTP database, and to utilize all of the information in the database to derive SQGs (in contrast, only data which had concordance between sediment chemistry and biological effects were used to derive the informal NSTP guidelines). In addition, user access to the information from individual studies has been improved by providing expanded data tables.

5.1.1 Procedures and Criteria for Screening Candidate Data Sets

The WEA is designed to integrate a diverse assortment of data to support the derivation of numerical SQGs. As such, data from spiked-sediment bioassays, sediment toxicity bioassays, and assessments of benthic invertebrate community characteristics were merged, along with the sediment quality assessment values developed in other jurisdictions (e.g., Puget Sound AETs, SQC derived using the EqPA, etc.) into a single database. These data were fully evaluated prior to inclusion to assure internal consistency in the database.

The screening procedures used to support the development of this database were designed to ensure that only high quality data is used to derive SQGs for Florida. The screening criteria used to evaluate spiked-sediment bioassay data and other matching sediment chemistry and biological effects data (i.e., co-occurrence data) are described in Appendix 1. These screening criteria were established to evaluate the acceptability of the experiment design, test protocols, analytical methods, and statistical procedures used in each study. To ensure internal consistency in the database, only those studies that met these screening criteria were considered appropriate for inclusion in the database. The sediment quality assessment values that have been derived by other jurisdictions were either incorporated directly into the database (if the concentrations of contaminants were originally expressed on a dry weight basis) or converted to concentrations expressed on a dry weight basis at 1% total organic carbon (TOC; if the assessment values were originally expressed on a TOC basis). Conversion of contaminant levels to dry weight concentrations at 1% TOC was considered to provide relatively conservative assessment values for entry into the database.

5.1.2 Expansion of the National Status and Trends Program Database

One of the principal limitations of the original NSTP database on the biological effects of sediment-sorbed contaminants, with respect to the derivation of SQGs for Florida, is its bias toward data derived from studies in the northeastern and western coastal areas of the United States. At the time the original database was assembled, few data were included on the biological effects of sediment-sorbed contaminants from sites located in the southeastern United States. Therefore, collection of acceptable data from Florida and other areas in the southeast was considered to be a priority in the present study.

To address the need for additional information on the biological effects of sediment-sorbed contaminants in general, and from sites in the southeastern United States in particular, a major initiative was undertaken to expand the original NSTP database. The first stage of the database expansion process involved identification and retrieval of candidate data sets from sites located in the southeastern United States. To this end, investigators in the field of sediment quality assessment located in the Gulf coast and southern Atlantic coast states were contacted and asked to identify studies they had conducted or participated in which contained matching sediment chemistry and biological effects data. Data sets were requested if the descriptions of these studies indicated that the data were likely to be acceptable. In addition, these investigators were asked to provide additional contacts who might be able to supply additional data relevant to the expansion of the database. Contacts in the southeast included representatives from U.S. Environmental Protection Agency, U.S. Army Corps of Engineering, Florida Department of Environmental Regulation (FDER), U.S. Fish and Wildlife Service, National Marine Fisheries Service, various academic institutions, and regionally-based consulting firms.

Significant effort was also expended to obtain additional data from other locations in the United States and Canada. In addition to the agencies identified above, contacts were made at Washington Department of Ecology, Oregon Department of Environmental Quality, California State Water Resources Control Board, Maryland Department of Environment, Port Authority of New York and New Jersey, Environment Canada, Public Works Canada, and the National Oceanic and Atmospheric Administration (NOAA).

Over the course of this study, more than 300 publications were retrieved and evaluated to determine their suitability for use in the derivation of SQGs. Nearly 90 of these publications were used to verify and expand the original NSTP database. Roughly 25% of the publications that were used in the present study were from studies conducted in the southeastern portion of the United States (i.e., North and South Carolina, Georgia, Florida, Alabama, Mississippi, Louisiana, and Texas).

Each of the data sets obtained during the course of the study were thoroughly reviewed and evaluated using the screening procedures outlined in Appendix 1. Acceptable data sets were subsequently analyzed and information pertaining to the potential biological effects of sediment-sorbed contaminants was integrated into the database. Following input into the database, every data entry (including each of the original NSTP database entries) was examined and verified against the original data source. This quality assurance procedure was designed to ensure that the database would meet Florida's requirements for consistently high quality data. This comprehensive, high quality database provides a basis for the derivation of preliminary SQGs for priority substances in Florida.

5.2 *Derivation of Numerical Sediment Quality Guidelines*

The expanded NSTP database is a comprehensive source of information on the potential effects of sediment-sorbed contaminants. Each record in the database contains detailed information on the location of the study, the species affected, the endpoint measured, the particle size distribution, the factors that could affect bioavailability of the contaminants (such as TOC and acid volatile solids; AVS), and the concentrations of the contaminants, if these data were available. This database was serially searched on-line to obtain information relevant to individual contaminants (e.g., cadmium, copper, etc.). Subsequently, the data obtained for each substance was sorted to create two separate data sets which incorporated the entries associated with biological effects and the entries associated with no observed biological effects, respectively.

The 'biological effects data set' (BEDS) was comprised primarily of information from co-occurrence analyses (COA) in which specific adverse biological effects (as indicated from the results of sediment toxicity bioassays or benthic invertebrate community assessments) were observed at some of the sites sampled. However, results of the COA were only included in the BEDS if concordance between the concentration of the chemical analyte and the observed biological response was apparent. In this respect, a contaminant was considered to be associated with the observed toxic response if the mean concentration at the sites at which significant effects were observed was a factor of two or more greater than the mean concentration at the sites at which effects were not observed (consistent with Long and Morgan 1990). Data obtained from other types of studies (i.e., spiked-sediment bioassays) and sediment quality assessment values (i.e., from the SLCA, EqPA, SQTA, etc.) were also included in the BEDS. Each of these entries was designated as a 'hit' [as indicated by an asterisk (*) in the supporting documentation; MacDonald *et al.* 1992]

A separate data set, the 'no biological effects data set' (NBEDS), was also established to include the balance of the data assembled over the course of the study. Several types of information were included in this data set. In general, these entries consisted of data from bioassays in which exposures of aquatic organisms to test sediments did not result in significant biological effects (i.e., no effect; NE). In addition, entries were included in the NBEDS when little or no concordance between the concentration of a contaminant in sediment and the observed biological effect was apparent (i.e., no concordance; NC or small gradient; SG). Data from field surveys of benthic invertebrate community indices were designated in a similar manner. Indeterminate AET values were reported in the data tables (MacDonald *et al.* 1992) but were not included in data evaluation. Each of these data sets were sorted by contaminant concentration to produce data sets in which concentrations occurred in ascending order. These two data sets were then used as the scientific basis for the development of SQGs.

The biological effects and NBEDS were used to derive numerical SQGs for Florida coastal waters (Figure 2). The arithmetic procedures used in the guidelines derivation process were designed to define three distinct ranges of contaminant concentrations; a no effects range, a possible effects range, and a probable effects range (Figure 3).

The range of sediment contaminant concentrations that are not likely to be associated with adverse biological effects on aquatic organisms (i.e., the no effects range) was defined using a two step process. First, a threshold effects level (TEL) was calculated. The TEL is considered to represent the upper limit of the range of sediment contaminant concentrations that is dominated by no effects data entries. The TEL was calculated as follows:

$$\mathbf{TEL} = \sqrt{\mathbf{BEDS\text{-}L} \cdot \mathbf{NBEDS\text{-}M}}$$

where:

TEL = Threshold Effect Level
BEDS-L = 15th percentile concentration in the biological effects data set;
NBEDS-M = 50th percentile concentration in the no biological effects data set.

Application of a safety factor to lowest observed effect levels is commonly recommended to account for the extended exposures to toxic substances, contaminant mixtures, and other factors that could affect the toxicity of a substance to aquatic organisms in the field (e.g.,

Figure 2. An overview of the modified NSTPA to the derivation of numerical sediment quality guidelines in Florida.

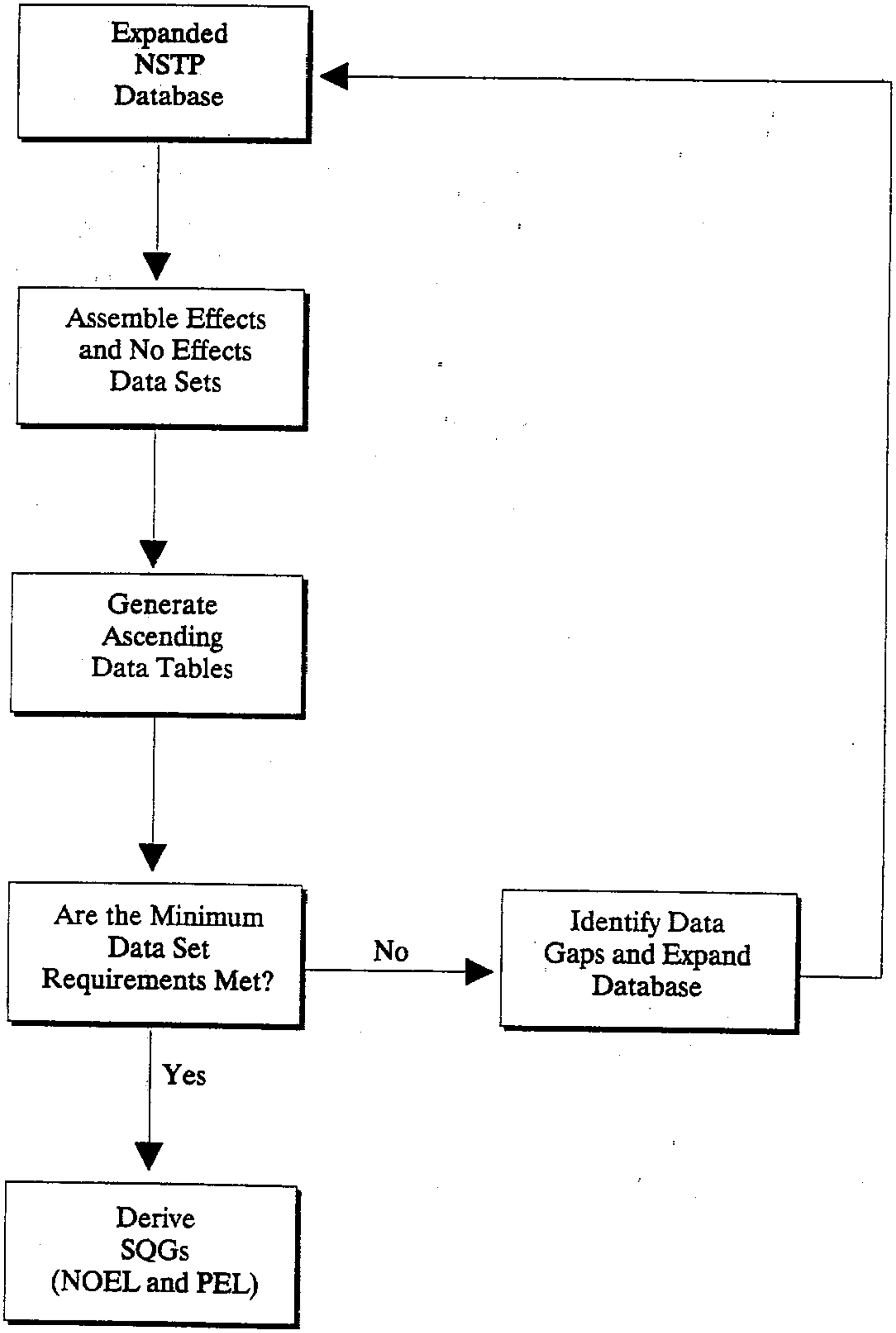
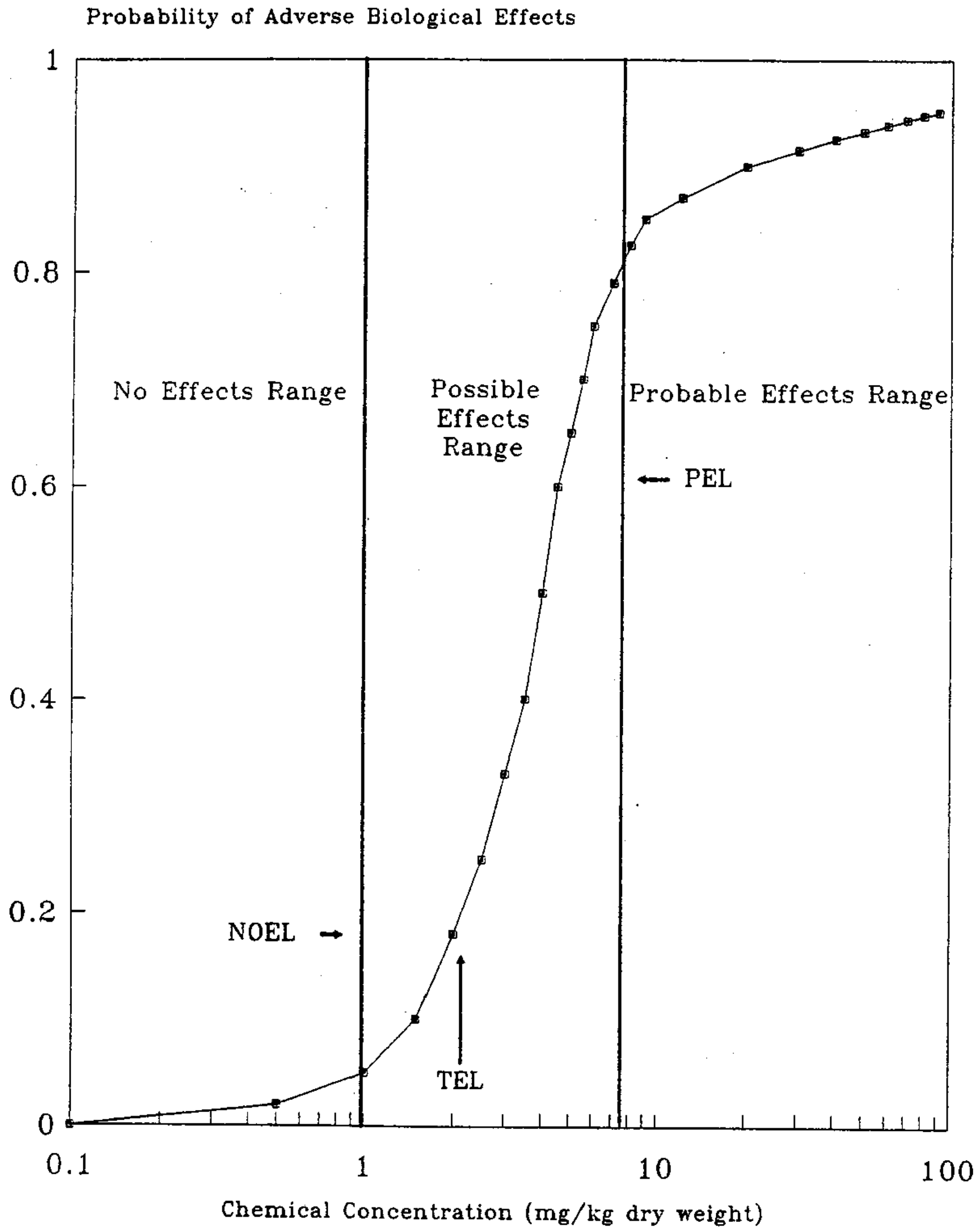


Figure 3. Conceptual example of sediment quality guidelines for cadmium.



EPA 1972; Kenaga 1982; CCME 1991). Therefore, a safety factor was applied to the TEL to estimate a no observed effect level (NOEL) for each contaminant as follows:

$$\mathbf{NOEL} = \mathbf{TEL \div SF}$$

where:

NOEL = No Observed Effect Level
SF = Safety Factor = 2

A safety factor of two was selected to convert the TELs to NOELs based on a previous analysis of the ratios of ER-L to ER-M values for various substances (Long and Morgan 1990). Application of this safety factor was considered to provide a pragmatic means of compensating for the limitations on the database with respect to the dearth of chronic toxicity data. As such, the NOEL is considered to represent the upper limit of the no effects range of contaminant concentrations. Within this range, concentrations of sediment-sorbed contaminants are not considered to represent significant hazards to aquatic organisms.

A probable effects level (PEL) was also calculated to define the lower limit of the range of contaminant concentrations that are usually or always associated with adverse biological effects (i.e., the lower limit of the probable effects range). The procedure utilized to calculate the PEL is designed to define a range of concentrations that is dominated by entries from the BEDS. Within the probable effects range, concentrations of sediment-sorbed contaminants are considered to represent significant and immediate hazards to aquatic organisms. The PEL was calculated as follows:

$$\text{PEL} = \sqrt{\textbf{BEDS-M} \cdot \textbf{NBEDS-H}}$$

where:

PEL = Probable Effects Level
BEDS-M = 50th percentile concentration in the biological effects data set;
NBEDS-H = 85th percentile concentration in the no biological effects data set.

The range of concentrations that could, potentially, be associated with biological effects (i.e., the possible effects range) is delineated by the NOEL (lower limit) and the PEL (upper limit). Within this range of concentrations, adverse biological effects are possible, however, it is difficult to reliably predict the occurrence, nature, and/or severity of these effects on an *a priori* basis. Site-specific conditions at sites with contaminant concentrations within this range are likely to control the expression of toxic effects. When contaminant concentrations fall within this range, further investigation is recommended to determine if sediment-sorbed contaminants represent significant hazards to aquatic organisms. ***It should be noted that guidelines, developed using the recommended procedures, do not address the potential for bioaccumulation of persistent toxic chemicals and potential adverse effects on higher trophic levels of the food chain.***

5.3 Rationale for the Recommended Guidelines Derivation Procedure

There are a wide variety of procedures that could be used to derive numerical SQGs from the expanded NSTP database. For example, Long and Morgan (1990) utilized the 10th (ER-L) and 50th (ER-M) percentile values in the BEDS to establish informal guidelines for evaluating sediment chemistry data collected under the NSTP. This method was similar to the procedure used by Klapow and Lewis (1979) to establish marine water quality standards in California. A major advantage of the procedure used by Long and Morgan (1990) is that it supports the establishment of three distinct ranges of chemical concentrations. However, only data from the BEDS were used in the calculation. As such, a large quantity of relevant information was not utilized in the guidelines derivation process.

The recommended procedure for deriving numerical SQGs described above is generally based on the approach used by Long and Morgan (1990). However, this procedure was modified to incorporate the information contained in both BEDS and NBEDS. The recommended procedure is designed to provide a consistent basis for estimating the concentrations of specific contaminants in sediment that are rarely or never, occasionally, and usually or always associated with adverse biological effects. As such, three ranges of contaminant concentrations may be defined; a no effects range, a possible effects range, and a probable effects range.

The arithmetic procedures for deriving the guidelines were designed to define ranges of concentrations with specific ratios of effects to no effects data entries. For example, the PEL is designed to delineate the lower limit of the range of concentrations which is dominated by data entries that are associated with adverse biological effects (i.e., a 'hit rate' of approximately 75% was considered to fulfil this narrative objective). If there were a total of 100 entries in each of the data sets, then the PEL would define the lower limit of a range of concentrations within which there would be, on average 50 entries from the BEDS and 15 entries from the NBEDS. This is predicted to be the case because the PEL is calculated as the geometric mean of the 50th percentile of the effects data set and 85th percentile of the NBEDS. The geometric mean is used in this calculation to account for uncertainty in the distributions of the data sets (Sokal and Rohlf 1981). The 'hit rate' within this range of concentrations would be, on average, 50/65 or 77%. This predicted 'hit rate' was considered to fulfil the narrative description of the probable effects range.

The no effects range of contaminant concentrations was defined in a similar manner. However, a safety factor was applied to the TEL to provide an extra margin of safety since the database used to calculate the guidelines was biased towards acute toxicity data.

There is a great deal of variability in the quantity of information available for each chemical, ranging from less than five data entries for 2,3,7,8-T_4CDD to several hundred data entries for cadmium. Due to the uncertainty associated with the evaluation of matching sediment chemistry and BEDS, a minimum quantity of data is required to support the derivation of SQGs. Minimum data requirements were established to ensure that any guidelines developed are supported by the weight of evidence that links contaminant concentrations to biological effects. To adhere to this principle, SQGs were derived only equalling or exceeding the PEL. This feature of the approach provides environmental managers with an additional tool for ranking the relative priority of contaminated sediments.

Lastly, the NSTPA has been extensively reviewed by experts from across North America. Over 1000 copies of the original publication (Long and Morgan 1990) have been distributed to date. In addition, it has recently been peer reviewed and published in a primary journal (Long 1992). Further, it has been selected for incorporation into the *Sediment Classification Methods Compendium* (EPA 1989a). Since its development in 1990, this approach has received positive evaluations from a wide variety of user groups and has been adopted directly and/or modified for implementation by both California (Lorenzato and Wilson 1991) and Canada (Smith and MacDonald 1992) as part of their guidelines derivation process. These favourable assessments emphasize the importance and utility of this procedure for deriving numerical SQGs.

In spite of the obvious benefits associated with the WEA for deriving SQGs, a number of limitations are also evident which could restrict application of these guidelines in Florida. The most serious of these shortcomings is associated with the limitations on the data that describe the bioavailability of sediment-sorbed contaminants. As such, it is not currently possible to express the guidelines in terms of the factors that influence the bioavailability of these contaminants. The importance of addressing bioavailability is emphasized by the results of several spiked-sediment bioassays. For example, Swartz *et al.* (1987) demonstrated that there was a three-fold increase in the toxicity of fluoranthene to the amphipod, *Rhepoxynius abronius*, when sediment TOC levels decreased from 0.5% to 0.2%. While reliance on ranges of concentrations instead of absolute values and consideration of the no effect data set serves to minimize this limitation, a potential for significant under- or over-protection of aquatic resources exists if guidelines are implemented that do not consider the bioavailability of sediment-sorbed contaminants.

Florida Department of Environmental Regulation recognizes the importance of accounting for the bioavailability of sediment-sorbed contaminants and has designed a field validation program to address this issue. In addition, FDER has developed a companion tool for use with the biological effects-based guidelines. The metals interpretive tool is based on normalizing metal levels to concentrations of aluminum in sediment, and provides a means of assessing the probable origin of sediment-sorbed metals (Schropp and Windom 1988; Schropp *et al.* 1989; Schropp *et al.* 1990; see Chapter 8 for a description of this tool). This tool emphasizes the importance of 'normalizers' in the interpretation of sediment chemistry data and provides a practical tool that compliments the effects-based guidelines. A detailed discussion on how these tools may be used together to assess sediment quality is provided in Chapter 8.

It is anticipated that the bioavailability of sediment-sorbed contaminants will be one of the principal issues addressed during the refinement of the preliminary guidelines. Currently, there is little comprehensive information with which to reliably predict the bioavailability of sediment-sorbed contaminants. When data were reported for TOC, AVS, grain size, and other potential normalizers, they were included in the expanded NSTP database. Unfortunately, only a small proportion of the investigations reported data for these variables. However, there are a number of initiatives that are likely to be relevant to the refinement of the preliminary SQGs in Florida. Specifically, EPA is currently in the process of developing sediment quality criteria that explicitly consider the factors that are likely to affect the bioavailability of contaminants. For this reason, it is recommended that the EqP values (currently under development by EPA; Di Toro *et al.* 1991) be fully evaluated and considered for use in Florida. Data from other studies conducted in Florida and elsewhere may also contribute to the identification of factors that influence bioavailability.

Another limitation of the WEA is that it does not fully support the quantitative evaluation of *cause and effect* relationships between contaminant concentrations and biological responses. Although information from spiked-sediment bioassays and EqP models is included in the expanded NSTP database, the recommended approach is considered to predict *associations* between contaminant concentrations and biological responses only. A wide variety of factors other than concentrations of the contaminant under consideration could have influenced the actual response observed in any given investigation. While the assembly of extensive information from numerous estuarine and marine sites across North America into a single database serves to minimize this limitation, there is still an undefined level of uncertainty associated with the resultant SQGs.

Application of the recommended approach may also be restricted by other limitations on the available information. Currently, only limited data exist on the chronic responses of marine and estuarine organisms to sediment-based contaminant challenges. In addition, the data from Florida and other areas in the southeast which link levels of contaminants to adverse biological effects are not overly abundant. Furthermore, only limited data are available on some potentially important sediment contaminants in Florida (including a variety of pesticides, dioxins and furans, etc.). This information shortfall impairs our ability to evaluate the overall applicability of the information to Florida.

The results of this evaluation indicate that SQGs developed using the recommended procedure are likely to be appropriate tools for conducting assessments of sediment quality in Florida. However, care should be exercised in applying these guidelines under some circumstances. In particular, these guidelines may not be directly applicable to sediments with atypical levels of the factors that influence the bioavailability of contaminants (e.g., very high or very low levels of TOC). Detailed discussions on the application of SQGs in regional and site-specific sediment quality assessments are provided in Chapters 7 and 8, respectively.

Chapter 6

Numerical Sediment Quality Guidelines for Florida Coastal Waters

6.0 Introduction

In Florida the maintenance and enhancement of designated uses of coastal ecosystems is identified as a high priority environmental management goal. Realization of this management goal is dependent on the maintenance of acceptable environmental conditions for the living resources in estuarine, nearshore, and marine ecosystems. While state water quality criteria provide effective tools for managing water quality, they provide little guidance on the management of sediment quality. Sediment quality guidelines (SQGs) are required to effectively address concerns relative to contamination of coastal ecosystems with substances that tend to associate with sediments. In particular, there is a need for guidelines that apply to the substances that are known or suspected to be contained in Florida coastal sediments.

6.1 A Preliminary Evaluation of Priority Contaminants in Florida Coastal Waters

Florida is not a highly industrialized state and, therefore, persistent and highly toxic contaminants are not likely to be distributed widely in its coastal zone. Nonetheless, various anthropogenic activities in the state do contribute significant quantities of environmental contaminants into coastal waters. Concerns relative to the contamination of coastal ecosystems fall into four general categories (Hand *et al.* 1990); urban stormwater runoff, agricultural runoff, domestic wastewater, and industrial wastewater. Consideration of each of these potential sources of environmental contaminants provides a basis for developing a preliminary list of chemical concerns in the Florida coastal zone (Table 3).

It would be virtually impossible to develop SQGs for every substance that may be released in Florida coastal waters. For this reason, the evaluation of chemical concerns in Florida coastal systems (Chapter 7) has been focused on the development of a list of priority substances (Table 3) known to be released in significant quantities into receiving water systems and to form associations with coastal sediments. These substances are considered to be of highest priority with respect to the development of numerical SQGs applicable to Florida's coast.

Stormwater runoff and associated contaminants are of particular concern in Florida. While nutrients and sediments are the most prevalent pollutants in urban stormwater, metals, PAHs, and other toxic substances may also be transported into receiving water systems by runoff from urban areas. Due to the substantial population growth in recent years and the

Table 3. Preliminary identification of chemical concerns in Florida coastal waters.

Substance	Reference/Rationale
Metals	
Arsenic	Long *et al.* (1991); FDER (In preparation).
Cadmium	Long *et al.* (1991); FDER (In preparation).
Chromium	Long and Morgan (1990); Long *et al.* (1991); FDER (In preparation).
Copper	Used in aquatic herbicides/found in fish; Long *et al.* (1991); Tefrey *et al.* (1983); Leslie (1990); FDER (In preparation).
Lead	Long and Morgan (1990); Long *et al.* (1991); FDER (In preparation).
Mercury	Long and Morgan (1990); Long *et al.* (1991); FDER (In preparation).
Nickel	Long *et al.* (1991); FDER (In preparation).
Silver	Long and Morgan (1990); FDER (In preparation).
Tributyltin	Used as anantifoulant on ships.
Zinc	Long and Morgan (1990); Long *et al.* (1991); FDER (In preparation).
Polycyclic Aromatic Hydrocarbons (PAHs)	
Acenaphthene	Delfino *et al.* (1991); FDER (In preparation).
Acenaphthylene	Delfino *et al.* (1991); FDER (In preparation).
Anthracene	Long and Morgan (1990); Delfino *et al.* (1991); FDER (In preparation).
Benz(a)anthracene	Long and Morgan (1990); Delfino *et al.* (1991); FDER (In preparation).
Benzo(a)pyrene	Long and Morgan (1990); Delfino *et al.* (1991); FDER (In preparation).
Chrysene	Long and Morgan (1990); Delfino *et al.* (1991); FDER (In preparation).
Dibenzo(a,h)anthracene	Long and Morgan (1990); Delfino *et al.* (1991); FDER (In preparation).
Fluorene	Long and Morgan (1990); Delfino *et al.* (1991); FDER (In preparation).
Fluoranthene	FDER (In preparation).
Napthalene	Long and Morgan (1990); Delfino *et al.* (1991); FDER (In preparation).
2-methylnapthalene	Long and Morgan (1990).
Phenanthrene	Long and Morgan (1990); Delfino *et al.* (1991); FDER (In preparation).
Pyrene	Long and Morgan (1990); Delfino *et al.* (1991); FDER (In preparation).
Total PAHs	Long and Morgan (1990); Long *et al.* (1991); FDER (In preparation).
Polychlorinated Biphenyls (PCBs)	
Total PCBs	Long and Morgan (1990); Long *et al.* (1991); Delfino *et al.* (1991); FDER (In preparation).

Table 3. Preliminary identification of chemical concerns in Florida coastal waters (continued).

Substance	Reference/Rationale
Pesticides	
Aldrin/Dieldrin	Long and Morgan (1990); Long *et al.* (1991); FDER (In preparation).
Azinophosmethyl (guthion)	Organophosphorous insecticide (K_{ow} > 10,000?)
Chlordane	Long and Morgan (1990); Long *et al.* (1991); FDER (In preparation).
Chlorothalonil	Chlorophenyl fungicide (K_{ow} = 20,000)
Chlorpyrifos	Organophosphorous insecticide (K_{ow} > 50,000)
DDT and metabolites	Long and Morgan (1990); Long *et al.* (1991); FDER (In preparation). Delfino *et al.* (1991).
Disulfoton	Organophosphorous insecticide (K_{ow} > 10,000)
Endosulfan	Delfino *et al.* (1991); FDER (In preparation).
Endrin	Organochlorine insecticide (K_{ow} > 10,000?); FDER (In preparation).
Heptachlor	Organochlorine insecticide (K_{ow} > 10,000?); FDER (In preparation).
Heptachlor epoxide	Organochlorine insecticide (K_{ow} > 10,000?); FDER (In preparation).
Lindane (gamma-BHC)	Organochlorine insecticide (K_{ow} > 10,000?); FDER (In preparation).
Mirex	Organochlorine insecticide (K_{ow} > 10,000?); FDER (In preparation).
Phorate	Organophosphorous insecticide (K_{ow} > 10,000?).
Quintozene (PCNB)	Chlorophenyl fungicide (K_{ow} = 10,000).
Toxaphene (alpha-BHC)	Organochlorine insecticide; FDER (In preparation).
Trifluralin	Dinitroanaline herbicide (K_{ow} > 200,000); FDER (In preparation).

* Criteria for selection of pesticides: K_{ow} > 5,000, and significant historic or current use (i.e., > 100,000 pounds/year in Florida; Pait *et al.* (1989; Worthing and Hance (1991).

Substance	Reference/Rationale
Chlorinated Organic Compounds	
2,3,7,8-T_4CDD	Pulp and paper industry.
2,3,7,8-T_4CDF	Pulp and paper industry
Pentachlorophenol	Delfino *et al.* (1991); FDER (In preparation).
Phthalates	
Bis(2-ethylhexyl)phthalate	Delfino *et al.* (1991).
Dimethyl phthalate	Delfino *et al.* (1991).
Di-n-butylphthalate	Delfino *et al.* (1991).

proximity of urban developments to the coast, urban stormwater represents a major source of contaminants into coastal ecosystems in Florida. Florida Department of Environmental Regulation (FDER; in preparation), Long and Morgan (1990), Delfino *et al.* (1991), and Long *et al.* (1991) provided lists of metals, PAHs, and other substances that have been detected in Florida coastal sediments at elevated levels (i.e., at levels that exceed the effects range low; ER-Ls reported by Long and Morgan 1990). These substances are reflected in the preliminary evaluation of chemical concerns in the Florida coastal zone.

High yields of agricultural products in Florida require the use of substantial quantities of fertilizers and pesticides. However, poorly managed runoff from agricultural areas has the potential to severely affect receiving water systems. The principal contaminants associated with agricultural runoff include nutrients, sediments, herbicides, insecticides, and other pesticides. While agricultural runoff is known to have significant impacts on lakes, rivers, and canals in the immediate vicinity of agricultural operations, contaminants may also be transported into coastal waters. The high-use pesticides (present or historic use) with significant potential to contaminate sediments in Florida's coastal areas are listed in Table 3. This list was assembled by considering pesticide use patterns (Pait *et al.* 1989), in conjunction with the physical/chemical properties of the substance (Worthing and Hance 1991). In addition, pesticides which have been detected in coastal sediments (Long and Morgan 1990; Long *et al.* 1991; Delfino *et al.* 1991) or in aquatic biota (Trefrey *et al.* 1983; Leslie 1990) in Florida were included in this list.

As might be expected in a state characterized by rapid urban development, inputs of domestic wastewater represent significant sources of environmental contaminants. While upgrades to wastewater treatment plants (WWTP) in recent years have resulted in improved water quality in many areas, progress towards the effective management of domestic wastewater treatment plant effluents is hampered by rapid population growth and severe limitations on financial resources in some portions of the state (Hand *et al.* 1990). Environmental contaminants that are commonly associated with WWTP effluents include nutrients, metals, halogenated methanes, and various chlorinated organic substances (MacDonald 1989).

While Florida is not characterized by widespread industry, substantial quantities of industrial wastewater are discharged into Florida waters (Farrow 1990). The major sources of these effluents are pesticides, organic chemicals and plastics, petroleum refining, and pulp and paper industries (Farrow 1989; 1990). In addition to pesticides, metals, and PAHs (Long and Morgan 1990; Long *et al.* 1991; Delfino *et al.* 1991), industrial activities have resulted in the release of substantial quantities of PCBs, polychlorinated dibenzo-p-dioxins (and related substances), and a wide variety of other organic contaminants into coastal waters (see MacDonald 1989 for a discussion on the nature and extent of contaminants that are often associated with industrial wastewaters).

6.2 Numerical Sediment Quality Guidelines

For each substance in Table 3, the available aquatic toxicity data was collected, evaluated, and included in the biological effects database for sediments, as appropriate. Using the procedure described in Chapter 5, SQGs (no observed effect levels and probable effect levels) were calculated for each substance for which adequate data were available. These guidelines are listed in Table 4. In addition, a brief discussion on the sources, fate, and effects of each substance (or group of substances) is provided. A preliminary evaluation of the guidelines and the degree of confidence associated with the results for each substance is provided in Table 5.

6.2.1 Metals

Numerical SQGs have been derived for a total of eight metals that occur in Florida coastal sediments. As is the case for the other substances, the SQGs are reported on a dry weight basis. While it is likely that further research will support the derivation of effects-based guidelines that are expressed in terms of the factors that influence bioavailability (e.g., AVS), such data are not yet available. Therefore, the preliminary guidelines should be used in conjunction with other assessment tools (such as the metals interpretive tool) to evaluate sediment quality conditions in Florida.

Arsenic

Arsenic is released naturally into the environment due to the weathering of arsenic-rich rocks and volcanic activity. However, in addition to the natural sources of this substance, arsenic is released into the environment as a result of human activities. For example, arsenic is used in pigments, for medical purposes, in glassmaking, and in alloys with lead and copper. In addition, arsenic is also used in some pesticides (including herbicides), in plant defoliants, and in various preservatives. Any of these activities may result in contamination of aquatic resources with arsenic (CCREM 1987).

The majority of arsenic in surface water occurs in a soluble form which can be coprecipitated with hydrated iron and aluminum oxides, or adsorbed/chelated by suspended organic matter in sediments or humic substances in bottom sediments. Arsenic has a strong affinity for sulphur, and it readily adsorbs on and coprecipitates with other metal sulfides (Demayo *et al.* 1979).

The availability of arsenic in sediments to aquatic biota appears to be minimal under oxidizing conditions. Bioaccumulation of arsenic has been observed in numerous aquatic organisms, though there is no evidence that arsenic is biomagnified to a significant degree through the food chain (Jaagumagi 1990a).

Table 4. A summary of sediment quality guidelines applicable to Florida coastal waters.

Substance	Total Number of Records	Number of Entries in the BEDS	Number of Entries in the NBEDS	Sediment Quality Guidelines NOEL	PEL
Metals (SQGs in mg/kg)					
Arsenic	140	27	113	8	64
Cadmium	261	83	178	1	7.5
Chromium	191	37	154	33	240
Copper	218	74	144	28	170
Lead	210	73	137	21	160
Mercury	169	42	127	0.1	1.4
Nickel	161	19	142	ID	ID
Silver	87	25	62	0.5	2.5
Tributyl Tin	21	4	17	ID	ID
Zinc	219	74	145	68	300
Polychlorinated Biphenyls (PCBs; SQGs in ug/kg)					
Total PCBs	125	50	75	24	260
Polycyclic Aromatic Hydrocarbons (PAHs; SQGs in ug/kg)					
Acenaphthene	69	30	39	22	450
Acenaphthylene	45	15	30	ID	ID
Anthracene	87	46	41	85	740
Fluorene	94	48	48	18	460
2-methyl naphthalene	46	28	18	ID	ID
Naphthalene	91	44	47	130	1100
Phenanthrene	98	51	47	140	1200
Sum LMW-PAHs	66	32	34	250	2400

Table 4. A summary of sediment quality guidelines applicable to Florida coastal waters (continued).

Substance	Total Number of Records	Number of Entries in the BEDS	Number of Entries in the NBEDS	Sediment Quality Guidelines	
				NOEL	PEL
Polycyclic Aromatic Hydrocarbons (PAHs; SQGs in ug/kg)					
Benz(a)anthracene	79	43	36	160	1300
Benzo(a)pyrene	87	44	43	230	1700
Chrysene	87	45	42	220	1700
Dibenzo(a,h)anthracene	73	31	42	31	320
Fluoranthene	116	71	45	380	3200
Pyrene	93	50	43	290	1900
Sum HMW-PAHs	60	25	35	870	8500
Total PAHs	77	33	44	2900	28000
Pesticides (SQGs in ug/kg)					
Aldrin	40	5	35	ID	ID
Azinophosmethyl (Guthion)	0	0	0	ID	ID
Chlordane	42	10	32	ID	ID
Chlorthalonil	0	0	0	ID	ID
Chlorpyrifos	1	1	0	ID	ID
p,p'-DDD	46	13	33	ID	ID
p,p'-DDE	64	23	41	1.7	130
p,p'-DDT	45	15	30	ID	ID
Total DDT	54	29	25	4.5	270
Dieldrin	47	12	35	ID	ID
Disulfoton	0	0	0	ID	ID
Endosulfan	9	3	6	ID	ID
Endrin	19	5	14	ID	ID

Table 4. A summary of sediment quality guidelines applicable to Florida coastal waters (continued).

Substance	Total Number of Records	Number of Entries in the BEDS	Number of Entries in the NBEDS	Sediment Quality Guidelines	
				NOEL	PEL
Pesticides (SQGs in ug/kg)					
Heptachlor	27	3	24	ID	ID
Heptachlor epoxide	7	0	7	ID	ID
Lindane (gamma-BHC)	42	10	32	ID	ID
Mirex	7	0	7	ID	ID
Phorate	0	0	0	ID	ID
Quintozene (PCNB)	0	0	0	ID	ID
Toxaphene (alpha-BHC)	25	3	22	ID	ID
Trifluralin	0	0	0	ID	ID
Chlorinated Organic Substances (SQGs in ug/kg)					
2,3,7,8-Tetrachlorodibenzo-p-dioxin	0	0	0	ID	ID
2,3,7,8-Tetrachlorodibenzofuran	0	0	0	ID	ID
Pentachlorophenol	18	3	15	ID	ID
Phthalates (SQGs in ug/kg)					
Bis(2-ethylhexyl)phthalate	31	16	15	ID	ID
Dimethyl phthalate	15	8	7	ID	ID
Di-n-butyl phthalate	16	7	9	ID	ID

Total Number of Records = Number of data records in the expanded NSTP database.
BEDS = Biological effects data set.
NBEDS = No biological effects data set.
Sediment Quality Guidelines were rounded to two significant figures.
All of the sediment quality guidelines are expressed on a dry weight basis, as potential normalizers (e.g., Al, TOC, AVS) were rarely reported.
ID = insufficient data to derive sediment quality guidelines.

Table 5. A preliminary evaluation of the relative degree of sediment quality guidelines applicable to the Florida coast.

Substance	% ’Hits’ in the No Effects Range (< =NOEL)	% ’Hits’ in Possible Effects Range (>NOEL to <PEL)	% ’Hits’ in Probable Effects Range (>= PEL)	Subjective Degree of Confidence in: NOEL	PEL
Metals					
Arsenic	2.6	12.5	56.7	H	M
Cadmium	5.8	26	68.2	H	H
Chromium	1.6	10.2	66.7	H	H
Copper	9.5	29.2	67.8	H	H
Lead	0	29.9	75	H	H
Mercury	7	30.1	33.3	H	L
Silver	0	18.8	76.2	H	H
Zinc	2.5	24.8	68.2	H	H
Polychlorinated Biphenyls (PCBs)					
Total PCBs	21.4	41.4	51.3	M	M
Polycyclic Aromatic Hydrocarbons (PAHs)					
Acenaphthene	33	26.7	76.2	L	H
Anthracene	25	36.8	84.8	M	H
Fluorene	30	33.3	84.8	L	H
Naphthalene	16	28.1	91.2	M	H
Phenanthrene	18.2	39.2	80.6	M	H
Sum LMW-PAHs	0	35	100	H	H

Table 5. A preliminary evaluation of the relative degree of sediment quality guidelines applicable to the Florida coast (continued).

Substance	% 'Hits' in the No Effects Range (< NOEL)	% 'Hits' in Possible Effects Range (NOEL to PEL)	% 'Hits' in Probable Effects Range (> PEL)	NOEL	PEL
Polycyclic Aromatic Hydrocarbons (PAHs)					
Benz(a)anthracene	26.7	36.4	87.1	L	H
Benzo(a)pyrene	0	51.1	74.1	H	H
Chrysene	20	37.5	84.4	M	H
Dibenzo(a,h)anthracene	0	56.4	50	H	M
Fluoranthene	7.7	44.4	93.9	H	H
Pyrene	0	34.1	89.7	H	H
Sum HMW-PAHs	15.4	26.9	76.2	M	H
Total PAHs	6.7	27	88	H	H
Pesticides					
p,p'-DDE	0	48.1	52.6	H	M
Total DDT	52.6	52.6	56.3	L	M

% 'Hits' = Number of data entries from biological effects data set /number of data entries from no biological effects data set.
NOEL = No Observed Effect Level
PEL = Probable Effects Level
H = High; M = Moderate; L = Low
Confidence in the NOEL was considered to be H, M, and L when % 'hits' was <10%, 10-25%, and >25%, respectively.
Confidence in the PEL was considered to be H, M, and L when % 'hits' was >65%, 50-65%, and <50%, respectively.

Exposure of aquatic organisms to arsenic-contaminated sediments may result in a variety of effects. While arsenic is known to be acutely toxic to aquatic biota, a variety of sublethal effects (including effects on the growth, reproduction, locomotion, behavior, and respiration) have also been observed in organisms exposed to arsenic (Eisler 1988). In mammals, exposure to arsenic has also been linked with a number of carcinogenic, mutagenic, and teratogenic effects.

Evaluation of the available information on the toxicity of sediment-sorbed arsenic to aquatic biota results in a recommended **NOEL of 8 mg/kg**. Adverse biological effects were only rarely observed (2.6% of the data entries) when concentrations of arsenic were within the no effects range (i.e., 0 - 8 mg/kg). The recommended NOEL is similar to the chronic marine threshold concentration (8.25 @ 1% TOC) calculated using the EqPA (Bolton *et al.* 1985) and somewhat lower than the ER-L (33 mg/kg) calculated using the NSTPA (Long and Morgan 1990).

The recommended **PEL of arsenic is 64 mg/kg**. Adverse biological effects were frequently observed (56.7% of the data entries) when concentrations of arsenic were within the probable effects range (i.e., ≥ 64 mg/kg). This level is the same as the PSSDA screening level in Puget Sound (ACE 1988), the San Francisco Bay AET for *R. abronius* (Long and Morgan 1990), and the AET for benthic species in California (Becker *et al.* 1990). The ER-M, calculated using the NSTPA, is 85 mg/kg (Long and Morgan 1990).

Cadmium

Cadmium is a trace element used in a wide variety of applications, including electroplating, the manufacture of pigments, storage batteries, telephone wires, photographic supplies, glass, ceramics, some biocides, and as a stabilizer in plastics. In addition, cadmium may be present in phosphate rock used for fertilizers. The main anthropogenic sources of cadmium appear to be mining, metals smelting, industries involved in the manufacture of alloys, paints, batteries, and plastics, agricultural uses of sludge, fertilizers and pesticides that contain cadmium, and the burning of fossil fuels (CCREM 1987).

In surface waters, cadmium generally occurs in the Cd(II) form as a constituent of inorganic (halides, sulfides, and oxides) and organic compounds. Transport of cadmium to the sediments occurs mainly through sorption to organic matter (and subsequent deposition) and through coprecipitation with iron, aluminum, and manganese oxides (Jaagumagi 1990a).

The availability of cadmium to aquatic biota is dependent on such factors as pH, redox potential, water hardness, and the presence of other complexing agents. Recently, Di Toro *et al.* (1991) revealed the importance of AVS in controlling the availability of cadmium. In general, cadmium is considered to have an extensive residence time and accumulates to significant levels in biological tissues (Jaagumagi 1990a).

Exposure of aquatic organisms to cadmium can result in a variety of adverse effects, including acute mortality, reduced growth, and inhibited reproduction (Eisler 1985a). In sediment, cadmium is toxic to marine amphipods at concentrations as low as 6.9 mg/kg (Swartz *et al.* 1985). Effects on the emergence, reburial, and avoidance behaviour of marine amphipods have also been observed in spiked-sediment bioassays with cadmium (Long and Morgan 1990).

Evaluation of available information on the toxicity of sediment-sorbed cadmium to aquatic biota results in a recommended **NOEL of 1.0 mg/kg**. Adverse biological effects were only rarely observed (5.8% of the data entries) when concentrations of cadmium were within the no effects range (i.e., 0 - 1.0 mg/kg). The recommended NOEL is significantly lower than the ER-L (5 mg/kg) calculated using the NSTPA (Long and Morgan 1990).

The recommended **PEL of cadmium is 7.5 mg/kg**. Adverse biological effects were frequently observed (68.2% of the data entries) when concentrations of cadmium were within the probable effects range (i.e., ≥ 7.5 mg/kg). This level is similar to the 1988 Puget Sound AET (9.6; PTI 1986) and the AET for benthic species in California (Becker *et al.* 1990). The ER-M, calculated using the NSTPA, was 9.0 mg/kg (Long and Morgan 1990).

Chromium

Like cadmium, chromium is a trace metallic element widely used in industrial processes. Hexavalent chromium compounds are used in the metallurgical industry in the production of chrome alloy and chromium metal. In addition, these compounds are used in the chemical industry in chrome plating and in the production of paints, dyes, explosives, ceramics, and paper. Trivalent chromium salts are used in textile dyeing, in the ceramics and glass industry, and in photography (CCREM 1987). The main sources of chromium to the environment are emissions from the ferrochromium and metal plating industries, with coal and oil burning, refractory production, cement manufacturing, and the production of chromium steels representing relatively less important sources (Taylor *et al.* 1979).

In aquatic systems, chromium is present mainly in the Cr(III) and Cr(VI) forms. The Cr(VI) form is relatively soluble and does not tend to sorb onto particulate matter to any significant extent. Under anaerobic conditions, Cr(VI) may be reduced to Cr(III). In contrast to Cr(VI), Cr(III) readily sorbs onto organic particulates and coprecipitates with hydrous iron and manganese oxides. Under anoxic conditions in the sediments, Cr may also form insoluble sulfides (Jaagumagi 1990a).

Adverse biological effects associated with exposure to chromium include mortality and decreased growth, with plants being more sensitive than fish (CCREM 1987). While chromium is not accumulated to a significant degree by fish (BCF < 3), algal communities may concentrate this substance (BCF = 8500; CCREM 1987). Chromium(VI) is more readily accumulated than Cr(III) and is considered to be the more toxic form (Jaagumagi 1990a).

Evaluation of the available information on the toxicity of sediment-sorbed chromium to aquatic biota results in a recommended **NOEL of 33 mg/kg**. Adverse biological effects were rarely observed (1.6% of the data entries) when concentrations of chromium were within the no effects range (i.e., 0 - 33 mg/kg). The ER-L, calculated using the NSTPA, was 80 mg/kg (Long and Morgan 1990).

The recommended **PEL of chromium is 240 mg/kg**. Adverse biological effects were frequently observed (66.7% of the data entries) when concentrations of chromium were within the probable effects range (i.e., ≥ 240 mg/kg). This level is similar to the 1988 Puget Sound AET for benthic organisms (260 mg/kg; PTI 1998) and the AET for amphipods, bivalves, and benthic species in California (> 240 mg/kg; Becker *et al.* 1990). The ER-M, calculated using the NSTPA, was 145 mg/kg (Long and Morgan 1990).

Copper

Copper is a common metallic element in crustal rocks and minerals. It's natural sources in aquatic environments include the weathering or the solution of copper-bearing minerals, copper sulfides, and native copper. Potential anthropogenic sources of copper include corrosion of brass and copper pipe by acidic waters, the use of copper compounds as aquatic algicides, sewage treatment plant effluents, runoff and groundwater contamination from agricultural uses of copper as fungicides and pesticides in the treatment of soils, and effluents and atmospheric fallout from industrial sources. Major industrial sources include mining, smelting and refining industries, copper wire mills, coal burning industries, and iron and steel producing industries (CCREM 1987).

Copper may exist in four oxidation states in aquatic systems, with Cu(I) and Cu(II) being the most common. In water, copper may form associations with organic matter and precipitates of hydroxides, phosphates, and sulfides. Formation of these complexes tends to facilitate transport to sediments. Under normal pH and redox conditions, copper tends to be present in sediments in the form of organic complexes, cupric carbonate complexes, and coprecipitates with iron and manganese oxides (Jaagumagi 1990a).

Copper is an essential micronutrient, and, therefore, it is readily accumulated by aquatic organisms (particularly in plants). However, no evidence exists to suggest that this substance is biomagnified in aquatic ecosystems (Jaagumagi 1990a). Copper is a broad spectrum biocide, which may be associated with acute and chronic toxicity, reduction in growth, interference with smoltification (the physiological changes that occur in preparation for the transition from freshwater to saltwater) in salmonids, and a wide variety of sublethal effects (Spear and Pierce 1979). There appears to be little difference in the sensitivity of aquatic organisms across taxonomic groups (CCREM 1987).

Evaluation of available information on the toxicity of sediment-sorbed copper to aquatic biota results in a recommended **NOEL of 28 mg/kg**. Adverse biological effects were infrequently observed (9.5% of the data entries) when concentrations of copper were within the no effects range (i.e., 0 - 28 mg/kg). The ER-L, calculated using the NSTPA, was 70 mg/kg (Long and Morgan 1990).

The recommended **PEL of copper is 170 mg/kg**. Adverse biological effects were frequently observed (67.8% of the data entries) when concentrations of copper were within the probable effects range (i.e., ≥ 170 mg/kg). This level may be compared to 1986 AET for benthic organisms in Puget Sound (310 mg/kg; Bellar *et al.* 1986) and the AET for benthic species in California (310 mg/kg; Becker *et al.* 1990). The ER-M, calculated using the NSTPA, was 390 mg/kg (Long and Morgan 1990).

Lead

Lead occurs as a constituent in a variety of minerals. The single largest use of lead is in the production of lead-zinc batteries. The second largest use of lead is in the manufacture of chemical compounds, particularly alkyllead additives for gasolines. Lead and its compounds are also used in electroplating, metallurgy, construction materials, coatings and dyes, electronic equipment, plastics, veterinary medicines, fuels and radiation shielding. Other uses of lead are for ammunition, corrosive-liquid containers, paints, glassware, fabricating storage tank linings, transporting radioactive materials, solder, piping, cable sheathing, roofing and sound attenuators (CCREM 1987).

While lead may be present in three oxidation states in aquatic environments, Pb(II) is the most stable ionic species. In sediments, lead is primarily found in association with iron and manganese hydroxides, however, it may also form associations with clays and organic matter. Lead tends to remain tightly bound to sediments under oxidizing conditions, however, it may be released into the water column under reducing conditions (Jaagumagi 1990a).

Aquatic organisms exhibit a wide range of sensitivities to lead, with gastropods being particularly vulnerable to exposures to lead. Aquatic plants appear to be relatively insensitive to the toxic effects of lead. Lead may be accumulated to relatively high levels by aquatic biota. Bioconcentration factors (BCFs) in algae (i.e., the ratio of tissue concentrations to concentrations in water) may be as high as 20,000, however, BCFs on fish and invertebrates tend to be much lower (500 to 1700; CCREM 1987).

Evaluation of the available information on the toxicity of sediment-sorbed lead to aquatic biota results in a recommended **NOEL of 21 mg/kg**. Adverse biological effects were never observed when concentrations of lead were within the no effects range (i.e., 0 - 21 mg/kg). The NOEL is similar to the chronic marine EqP threshold of 33 mg/kg (Bolton *et al.* 1985) and the ER-L of 35 mg/kg (Long and Morgan 1990).

The recommended **PEL of lead is 160 mg/kg**. Adverse biological effects were usually observed (75% of the data entries) when concentrations of lead were within the probable effects range (i.e., ≥ 160 mg/kg). This level is similar to the California AET for benthic species (150 mg/kg; Becker *et al.* 1990). The ER-M, calculated using the NSTPA, was 110 mg/kg (Long and Morgan 1990).

Mercury

Mercury is a trace element that occurs most commonly in the sulfide mineral cinnabar. Mercury is used in the production of chlorine, caustic soda and hydrogen, in the paint industry, in the pulp and paper industry, for electrical equipment, in medicinal compounds, and in thermometers. Mercury-based pesticides were once used in agriculture, however, the use of such pesticides has now been restricted (CCREM 1987). Significant anthropogenic sources to aquatic ecosystems can include mining and smelting, coal combustion, paints, waste incineration, and the chlor-alkali industry (Jaagumagi 1990a).

In aquatic systems, mercury is generally sorbed to particulate matter. In natural systems, mercury can exist in three oxidation states, including elemental Hg, Hg(I), and Hg(II). Both Hg(I) and Hg(II) can be methylated by microorganisms under anaerobic and aerobic conditions. In sediments, mercury tends to form associations with organic matter. Under anaerobic conditions, mercury may combine with sulphur to form insoluble sulfides (Jaagumagi 1990a).

Mercury is highly toxic to aquatic biota, with methylmercury being the most toxic form of the substance. Aquatic plants, invertebrates, and fish exhibit similar sensitivities to mercury, however, a great deal of variability exists within each of these groups. Mercury has the potential to accumulate to high levels in aquatic organisms, with BCFs as high as 85,000 observed in some fish species (CCREM 1987). Due to its high mammalian toxicity, bioaccumulation of mercury in fish and other aquatic species has significant implications with respect to human health.

Evaluation of available information on the toxicity of sediment-sorbed mercury to aquatic biota results in a recommended **NOEL of 0.1 mg/kg**. Adverse biological effects were only infrequently observed (7% of the data entries) when concentrations of mercury were within the no effects range (i.e., 0 - 0.1 mg/kg). The ER-L, calculated using the NSTPA, was also 0.15 mg/kg (Long and Morgan 1990).

Using the modified NSTPA, a **PEL of 1.4 mg/kg** was calculated. Evaluation of this SQG indicates that adverse biological effects were at similar frequencies within the possible (0.15 - 1.4 mg/kg; 30.1% hits) and probable (≥ 1.4 mg/kg; 33.3% hits) effects ranges. Therefore, only a moderate amount of confidence should be placed on the recommended PEL. However, this value is similar to the San Francisco Bay AETs for amphipods (1.3 mg/kg) and bivalves (1.5 mg/kg; Long and Morgan 1990). The ER-M, calculated using the NSTPA, was also similar at 1.3 mg/kg (Long and Morgan 1990).

Nickel

Nickel ranks as the 23rd element in order of abundance in the earth's crust and occurs naturally, mainly, in combination with sulphur, arsenic, and antimony. In ore deposits, it commonly occurs with iron and copper. Nickel is used, primarily, in the manufacturing of stainless steel, nickel plating, and other nickel alloys. Nickel is also used as a catalyst in generating plants, gas turbine engines, cryogenic containers, and pollution abatement equipment. The most important anthropogenic sources of nickel include fossil fuel combustion, nickel ore mining, smelting and refining activities, and the electroplating industries (CCREM 1987).

In aquatic systems, nickel occurs primarily in the Ni(II) form. Nickel is deposited in sediments as a result of coprecipitation with iron and manganese oxides and sorption to organic matter. In sediments, nickel tends to form complexes with iron and manganese oxides, however, it may form insoluble complexes with sulfides under anaerobic conditions (Jaagumagi 1990a).

Exposure of aquatic organisms to nickel-contaminated sediments may result in a variety of adverse effects, including mortality, reduction in growth, and avoidance reactions. The toxicity of nickel increases in the presence of copper, therefore, synergism may be a factor that modifies the toxicity of this substance. While bioconcentration of nickel has been observed in a variety of organisms (particularly in annelids), biomagnification is not a significant concern in aquatic environments (CCREM 1987).

While insufficient data were available to derive a numerical SQG for nickel, the chronic marine EqP threshold for nickel was 5 mg/kg (Bolton *et al.* 1985) and the ER-L was 30 mg/kg (Long and Morgan 1990).

By comparison, the Puget Sound AETs for nickel range from 28 mg/kg (1986 Mictrotox AET; Bellar *et al.* 1986) to > 140 mg/kg (1988 amphipod and benthic community AETs; PTI 1988). The ER-M reported by Long and Morgan (1990) was similar at 50 mg/kg.

Silver

Silver is among the least common but most widely distributed elements in crustal rocks. Photographic materials represent the single largest use of silver. Other uses of this element include the manufacture of sterling and plated ware, jewellery, coins and medallions, electrical and electronic products, brazing alloys and solders, catalysts, mirrors, fungicides, and dental and medical supplies. Potential sources of silver to the aquatic environment include leachates from landfills, waste incineration, coal combustion, and effluents from the iron, steel and cement industries. In addition, wastewater treatment plants may also contribute significant loadings of silver to aquatic ecosystems (CCREM 1987).

In aqueous systems, silver may occur as elemental Ag, Ag(I), or Ag(II), however, ionic silver is primarily found in the univalent state. In water, silver may occur in colloidal form, sorbed to humic substances, and in various complexes with sulphur, arsenic, antimony, tellurium, and selenium. In sediments, silver tends to be found in association with manganese dioxide, sulphur, and various halides. Silver may also be adsorbed to organic material in sediments (CCREM 1987).

Silver is one of the most toxic metals to aquatic life. In general, plants are somewhat less sensitive than fish and aquatic invertebrates, with toxicity dependent primarily on metal speciation. Silver nitrate and silver iodide have been identified as highly toxic species. Silver has a fairly low potential to accumulate in aquatic organisms, with BCFs ranging from less than 1 to 240 (CCREM 1987).

Evaluation of the available information on the toxicity of sediment-sorbed silver to aquatic biota results in a recommended **NOEL of 0.5 mg/kg.** Adverse biological effects were never observed when concentrations of silver were within the no effects range (i.e., 0 - 0.5 mg/kg). The ER-L, calculated using the NSTPA, was 1.0 mg/kg (Long and Morgan 1990).

The recommended **PEL of silver is 2.5 mg/kg.** Adverse biological effects were frequently observed (76.2% of the data entries) when concentrations of silver were within the probable effects range (i.e., ≥ 2.5 mg/kg). In California, the AETs for bivalves and benthic species were 2.3 and 3.7 mg/kg, respectively (Becker *et al.* 1990). The recommended PEL is similar to the ER-M of 2.2 mg/kg reported by Long and Morgan (1990).

Tributyltin

Tributyltin is a member of a family of organotin compounds that are used in the production of plastics and as biocidal wood preservatives. Tributyltin oxide (TBTO) and tributyltin fluoride (TBTF) are the most important of the tributyltin compounds. Tributyltin oxide is used as a slimicide in cooling water towers, as a wood preservative, and as an antifouling additive in marine paint. The major use of TBTF is also as an antifouling agent in marine paint, and the use of both substances in marine paints represents potentially significant sources of tributyltin into aquatic ecosystems (CCREM 1987).

Tributyltin compounds are highly toxic to aquatic organisms (both plants and animals), as would be expected given their use as general biocides. Eisler (1985b) reported that tributyltins were capable of causing adverse biological effects at extremely low levels, and that these substances have been implicated as a major cause of reproductive failure in European flat oysters at several locations in recent years. Its high toxicity and significant potential for release into the aquatic environment make tributyltins a serious concern in marine sediments. While insufficient data are available to develop SQGs (NOEL and PEL) for tributyltin, extreme mortality (100%) has been observed in grass shrimp exposed (96 hour static test) to concentrations as low as 10 mg/kg (Clark *et al.* 1987). Since grass shrimp are relatively insensitive test species, adverse effects on other organisms could be expected at concentrations well below this level.

Zinc

Zinc ranks as the 24th most abundant crustal element, occurring primarily as sulfide, carbonate, and silicate ores. Zinc is used in coatings to protect iron and steel, in alloys for die casting, in brass, in dry batteries, in roofing and exterior fittings for buildings, and in some printing processes. The principal sources of zinc to aquatic systems include municipal wastewater effluents, zinc mining, smelting , and refining activities, wood combustion, waste incineration, iron and steel production, and other atmospheric emissions (CCREM 1987).

In aquatic systems, zinc occurs primarily as Zn(II), but can also form organozinc compounds. At neutral pH, zinc may be deposited in sediments by sorption to hydrous iron and manganese oxides, clay minerals, and organic matter. However, adsorption is very low at pHs below 6. Iron and manganese oxides/hydroxides appear to be the most important scavengers of zinc in coarse sediments that are low in organic matter. However, sorption to organic matter appears to be the most important environmental fate process in fine grained sediments. Under reducing conditions, organically-bound zinc generally forms insoluble sulfides (Jaagumagi 1990a).

Zinc is an essential micronutrient and uptake in most aquatic organisms appears to be independent of environmental concentrations. It has been found to bioaccumulate in some organisms, though there is no evidence of biomagnification (Jaagumagi 1990a). Aquatic organisms exhibit a wide range of sensitivities to zinc, however, there do not appear to be systematic differences in the toxicity of this substance between three major taxonomic groups (fish, invertebrates, and aquatic plants; CCREM 1987).

Evaluation of the available information on the toxicity of sediment-sorbed zinc to aquatic biota results in a recommended **NOEL of 68 mg/kg**. Adverse biological effects were rarely observed (2.5% of the data entries) when concentrations of zinc were within the no effects range (i.e., 0 - 68 mg/kg). The ER-L, calculated using the NSTPA, was 120 mg/kg (Long and Morgan 1990).

The recommended **PEL of zinc is 300 mg/kg**. Adverse biological effects were frequently observed (68.2% of the data entries) when concentrations of zinc were within the probable effects range (i.e., ≥ 300 mg/kg). In California, AET values ranged from 150 mg/kg for bivalves to 340 mg/kg for benthic species (Becker *et al.* 1990). The Puget Sound AETs were considerably higher (410 to 1600 mg/kg; PTI 1988; Bellar *et al.* 1986). The recommended PEL is similar to ER-M of 270 mg/kg reported by Long and Morgan (1990).

6.2.2 Polycyclic Aromatic Hydrocarbons

Polycyclic aromatic hydrocarbons (PAHs) is the general term applied to a group of compounds comprised of several hundred organic substances with two or more benzene rings. They occur in the environment mainly as a result of incomplete combustion of organic matter (forest fires, internal combustion engines, wood stoves, coal, coke, etc.). They are also major constituents of petroleum and its derivatives, with oil spills and refinery effluents being major sources of PAH contamination to estuarine and marine environments (MacDonald *et al.* 1991). In addition, WWTP effluents and runoff from urban areas, particularly from roads, are known to contain significant quantities of PAHs. Further, inputs of PAHs in aquatic ecosystems may occur as a result of oil spills, forest fires and agricultural burning, leaching from waste disposal sites, and coal gasification (Eisler 1987; Neff 1979; Campbell *et al.* 1979). PAHs are also produced by natural processes at very low rates (Blumer 1976).

In marine and estuarine environments, PAHs tend to form associations with suspended and deposited particulate matter (Eisler 1987). This sorption of PAHs to sediments is strongly correlated with the total organic carbon (TOC) content of sediments (Gillam 1991). Sediments contaminated with PAHs have been identified in a number of locations in the Florida coastal zone (Long and Morgan 1990). Substances detected most frequently in coastal sediments include acenaphthylene, anthracene, benz(a)anthracene, benzo(a)pyrene, chrysene, fluoranthene, phenathrene, and pyrene (Delfino *et al.* 1991). In general, elevated levels of sediment-sorbed PAHs in Florida are found in the vicinity of urban areas.

Exposure to PAHs may result in a wide range of effects on biological organisms. While some PAHs are known to be carcinogenic, others display little or no carcinogenic, mutagenic, or teratogenic activity (Neff 1979; EPA 1980, 1982a, b, c; NRCC 1983; Sims and Overcash 1983). Many carcinogenic PAHs also exhibit teratogenic and mutagenic effects. Several PAHs exhibit low levels of toxicity to terrestrial life forms, yet are highly toxic to aquatic organisms (Eisler 1987). The bioavailability (and hence, toxicity) of PAHs is known to depend on the concentration of TOC in the sediment (Bolton *et al.* 1985; Lyman *et al.* 1987).

Acenaphthene

Evaluation of the available information on the toxicity of sediment-sorbed acenaphthene to aquatic biota results in a recommended **NOEL of 22 μg/kg**. However, a significant number of adverse biological effects were observed (33% of the data entries) when concentrations of acenaphthene were within the no effects range (i.e., 0 - 22 μg/kg). A more conservative estimate of the NOEL would be in the order 10 μg/kg. Therefore, some potential for adverse biological effects exists when concentrations of acenaphthene fall between 10 and 22 μg/kg. Adverse biological effects are most likely to be observed within this range of concentrations when low levels of TOC (i.e., < 1%) are present in sediments. Long and Morgan (1990) reported an ER-L of 150 μg/kg for this substance.

The recommended **PEL of acenaphthene is 450 μg/kg**. Adverse biological effects were frequently observed (76.2% of the data entries) when concentrations of acenaphthene were within the probable effects range (i.e., $\geq$ 450 μg/kg). In California, AET values ranged from 9 μg/kg for bivalves (in San Francisco Bay; Long and Morgan 1990) to 56 μg/kg for amphipods (Becker *et al.* 1990). The Puget Sound AETs were considerably higher (500 to 2000 μg/kg; PTI 1988; Bellar *et al.* 1986). The recommended PEL is somewhat lower than the ER-M of 650 μg/kg reported by Long and Morgan (1990).

Acenaphthylene

Insufficient data were available to develop SQGs for acenaphthylene. However, adverse biological effects were never observed at concentrations of acenaphthylene below 35 μg/kg in sediments. This concentration could be used as an interim NOEL until additional data become available. In California, the AET for benthic species was 44 μg/kg (Becker *et al.* 1990)

Adverse biological effects were frequently observed (83.3% of the data entries) at concentrations of acenaphthylene at or above 500 μg/kg. In the absence of other numerical SQGs, 500 μg/kg could be used as an interim probable effects level. The 1986 Puget Sound benthic community AET was 640 μg/kg (Bellar *et al.* 1986). No NSTPA values were calculated for this substance (Long and Morgan 1990).

Anthracene

Evaluation of available information on the toxicity of sediment-sorbed anthracene to aquatic biota results in a recommended **NOEL of 85 μg/kg**. Adverse biological effects were occasionally observed (25% of data entries) when concentrations of anthracene fell within the no effects range (i.e., 0 - 85 mg/kg). Therefore, only a moderate level of confidence should be placed in this guideline. Several AET values fell within the no effects range, including the AETS for bivalves in San Francisco Bay (24 μg/kg), for benthic species in northern California (60 μg/kg), and for mussels statewide (60 μg/kg; Becker *et al.* 1990). The ER-L of 85 μg/kg reported by Long and Morgan (1990) was the same as the NOEL calculated in this study.

The recommended **PEL of anthracene is 740 μg/kg**. Adverse biological effects were frequently observed (84.8% of the data entries) when concentrations of anthracene were within the probable effects range (i.e., ≥ 740 μg/kg). This level was lower than the Puget Sound AET values, which ranged from 960 (bivalve) to 13,000 μg/kg (amphipod; PTI 1988). The recommended PEL is similar to ER-M of 960 μg/kg reported by Long and Morgan (1990).

Fluorene

Evaluation of available information on the toxicity of sediment-sorbed fluorene to aquatic biota results in a recommended **NOEL of 18 μg/kg**. However, a significant number of adverse biological effects were observed (30% of the data entries) when concentrations of fluorene were within the no effects range (i.e., 0 - 18 μg/kg). A more conservative estimate of the NOEL would be in the order 10 μg/kg. Therefore, some potential for adverse biological effects exists when concentrations of fluorene fall between 10 and 18 μg/kg. Adverse biological effects are most likely to be observed within this range of concentrations when low levels of TOC (i.e., < 1%) are present in sediments. Long and Morgan (1990) reported an ER-L of 35 μg/kg for fluorene.

The recommended **PEL of fluorene is 460 μg/kg.** Adverse biological effects were frequently observed (84.8% of the data entries) when concentrations of fluorene were within the probable effects range (i.e., ≥ 450 μg/kg). This level is lower than the Puget Sound AET values, which ranged from 540 (bivalve) to 3,600 μg/kg (amphipod; PTI 1988). The recommended PEL is also somewhat lower than the ER-M of 640 μg/kg reported by Long and Morgan (1990).

2-methylnaphthalene

Insufficient data were available to calculate a numerical sediment quality guideline for 2-methylnaphthalene. However, Long and Morgan (1990) reported an AET of 27 μg/kg for bivalves in San Francisco Bay. The ER-L, calculated using the NSTPA, was 65 μg/kg (Long and Morgan 1990). In California, AET values ranged from 70 μg/kg for bivalves and benthic species to > 130 μg/kg for amphipods (Becker *et al.* 1990). The Puget Sound AETs were considerably higher (670 to 1900 μg/kg; PTI 1988; Bellar *et al.* 1986). The recommended PEL is roughly half of the ER-M of 670 μg/kg reported by Long and Morgan (1990).

Naphthalene

Evaluation of available information on the toxicity of sediment-sorbed naphthalene to aquatic biota results in a recommended **NOEL of 130 μg/kg.** Adverse biological effects were occasionally observed (28.1% of the data entries) when concentrations of naphthalene were within the no effects range (i.e., 0 - 130 μg/kg). The recommended NOEL was similar to the San Francisco Bay AET of 160 μg/kg for bivalves and amphipods (Long and Morgan 1990). The recommended NOEL is significantly lower than the ER-L of 340 μg/kg reported by Long and Morgan (1990).

The recommended **PEL of naphthalene is 1100 μg/kg.** Adverse biological effects were usually observed (91.2% of the data entries) when concentrations of naphthalene were within the probable effects range (i.e., ≥ 1100 μg/kg). The Puget Sound AETs were considerably higher (2100 to 2700 μg/kg; PTI 1988; Bellar *et al.* 1986). Likewise, the ER-M of 2100 μg/kg, reported by Long and Morgan (1990), was significantly higher than the recommended PEL.

Phenanthrene

Evaluation of available information on the toxicity of sediment-sorbed phenanthrene to aquatic biota results in a recommended **NOEL of 140 μg/kg.** Adverse biological effects were sometimes observed (18.2% of the data entries) when concentrations of phenanthrene were within the no effects range (i.e., 0 - 140 μg/kg). The recommended NOEL was similar to the Northern California AET of 170 μg/kg for benthic species (Becker *et al.* 1990). The ER-L, calculated using the NSTPA, was 225 μg/kg (Long and Morgan 1990).

The recommended **PEL of phenanthrene is 1200 μg/kg**. Adverse biological effects were usually observed (80.6% of the data entries) when concentrations of phenanthrene were within the probable effects range (i.e., ≥ 1200 μg/kg). The Puget Sound AETs were similar to the recommended PEL at 1500 μg/kg for Microtox and bivalves, however the AET for amphipods was considerable higher (5400 μg/kg; PTI 1988). The recommended PEL is similar to the ER-M of 1380 μg/kg reported by Long and Morgan (1990).

Sum Low Molecular Weight PAHs

The group of low molecular weight (LMW) PAHs considered in the present study includes acenaphthene, acenaphthylene, anthracene, fluorene, 2-methylnaphthalene, naphthalene, and phenanthrene. Due to their similar mode of toxic action, these substances are frequently considered together in toxicity assessments (e.g., Gillam 1991). Evaluation of available information on the effects of LMW PAHs on aquatic biota results in a recommended **NOEL of 250 μg/kg** and a **PEL of 2400 μg/kg**. Within the no effects range and probable effects range, the frequency of adverse biological effects data entries were 0% and 100%, respectively. By comparison, AETs for LMW PAHs in California ranged from 320 μg/kg for bivalves to 2100 μg/kg for amphipods (Becker *et al.* 1990). In Puget Sound, AETs ranged from 5100 to 6100 μg/kg (Beller *et al.* 1986; PTI 1988).

Benz(a)anthracene

Evaluation of available information on the toxicity of sediment-sorbed benz(a)anthracene to aquatic biota results in a recommended **NOEL of 160 μg/kg**. Adverse biological effects were periodically reported (26.7% of data entries) when concentrations of benz(a)anthracene fell within the no effects range of concentrations (i.e., 0 - 160 μg/kg). In California, AETs for bivalves (statewide) and benthic species (northern California; 150 μg/kg) fell slightly below the recommended NOEL. An ER-L of 230 μg/kg was reported by Long and Morgan (1990).

The recommended **PEL of benz(a)anthracene is 1300 μg/kg**. Adverse biological effects were usually observed (87.1% of the data entries) when concentrations of benz(a)anthracene were within the probable effects range (i.e., ≥ 1300 μg/kg). This level was similar to several of the Puget Sound AET values, which ranged from 1300 (Microtox; Bellar *et al.* 1986) to 5100 μg/kg (amphipod; PTI 1988). Pavlou *et al.* (1987) reported a chronic marine sediment quality criterion of 1600 μg/kg for this substance at 1% TOC. The ER-M, reported by Long and Morgan (1990), was also 1600 μg/kg.

Benzo(a)pyrene

Evaluation of available information on the toxicity of sediment-sorbed benzo(a)pyrene to aquatic biota results in a recommended **NOEL of 230 μg/kg**. Adverse biological effects were never observed when concentrations of benzo(a)pyrene were within the no effects range (i.e., 0 - 230 μg/kg). The ER-L, calculated using the NSTPA, was 400 μg/kg (Long and Morgan 1990).

The recommended **PEL of benzo(a)pyrene is 1700 μg/kg**. Adverse biological effects were frequently observed (74.1% of the data entries) when concentrations of benzo(a)pyrene were within the probable effects range (i.e., ≥ 1700 μg/kg). The recommended PEL is similar to the San Francisco Bay AET for bivalves (1800 μg/kg; Long and Morgan 1990). Puget Sound AETs for Microtox and oysters (1600 μg/kg) were also similar to the recommended PEL (PTI 1988). The recommended PEL is somewhat lower than the ER-M of 2500 μg/kg reported by Long and Morgan (1990).

Chrysene

Evaluation of available information on the toxicity of sediment-sorbed chrysene to aquatic biota results in a recommended **NOEL of 220 μg/kg**. Adverse biological effects were infrequently observed (20% of the data entries) when concentrations of chrysene were within the no effects range (i.e., 0 - 220 μg/kg). The recommended NOEL was similar to the Northern California AET of 190 μg/kg for benthic species (Becker *et al.* 1990). The ER-L of 400 μg/kg, reported by Long and Morgan (1990), was somewhat higher than the recommended NOEL.

The recommended **PEL of chrysene is 1700 μg/kg**. Adverse biological effects were usually observed (84.4% of the data entries) when concentrations of chrysene were within the probable effects range (i.e., ≥ 1700 μg/kg). The Puget Sound AET for Microtox, at 1400 μg/kg was similar to the recommended PEL (Bellar *et al.* 1986; PTI 1988). However, the AETs for bivalves and amphipods (2800 - 9200 μg/kg) were considerable higher (PTI 1988). The ER-M for chrysene was 2800 μg/kg (Long and Morgan 1990).

Dibenzo(a,h)anthracene

Evaluation of the available information on the toxicity of sediment-sorbed dibenzo-(a,h)anthracene to aquatic biota results in a recommended **NOEL of 31 μg/kg**. Adverse biological effects were never observed when concentrations of dibenzo(a,h)anthracene were within the no effects range (i.e., 0 - 31 μg/kg). The recommended NOEL was similar to the Northern California AET of 63 μg/kg for benthic species and the California AET for bivalves of 63 μg/kg (Becker *et al.* 1990). The ER-L reported by Long and Morgan (1990) was 60 μg/kg.

The recommended **PEL of dibenzo(a,h)anthracene is 320 μg/kg**. Adverse biological effects were commonly observed (50% of the data entries) when concentrations of dibenzo(a,h)anthracene were within the probable effects range (i.e., ≥ 320 μg/kg). The frequency of adverse biological effects within the probable effects range was greater (65.6% of the data entries) when the PEL was estimated at 200 μg/kg. In Puget Sound, AETs ranged from 230 μg/kg (for bivalves and Microtox) to 1200 μg/kg (for benthic species;

Bellar *et al.* 1986; PTI 1988). The recommended PEL is similar to the ER-M of 260 μg/kg reported by Long and Morgan (1990).

Fluoranthene

Evaluation of the available information on the toxicity of sediment-sorbed fluoranthene to aquatic biota results in a recommended **NOEL of 380 μg/kg**. Adverse biological effects were rarely observed (7.7% of data entries) when concentrations of fluoranthene were within the no effects range (i.e., 0 - 380 μg/kg). The recommended NOEL was similar to the Northern California AET of 390 μg/kg for benthic species (Becker *et al.* 1990). The ER-L (600 μg/kg) was somewhat higher than the recommended NOEL (Long and Morgan 1990).

The recommended **PEL of fluoranthene is 3200 μg/kg**. Adverse biological effects were usually observed (93.9% of the data entries) when concentrations of fluoranthene were within the probable effects range (i.e., ≥ 3200 μg/kg). The recommended PEL was similar to 1986 Puget Sound AET for amphipods (3900 μg/kg; Bellar *et al.* 1986) and similar to the California AET for amphipods (> 3700 μg/kg; Becker *et al.* 1990). The recommended PEL is also similar to the ER-M of 3600 μg/kg reported by Long and Morgan (1990).

Pyrene

Evaluation of the available information on the toxicity of sediment-sorbed pyrene to aquatic biota results in a recommended **NOEL of 290 μg/kg**. Adverse biological effects were never observed when concentrations of pyrene were within the no effects range (i.e., 0 - 290 μg/kg). The recommended NOEL was similar to the national screening level concentration of 434 μg/kg (at 1% TOC) reported by Neff *et al.* (1986). The ER-L, calculated using the NSTPA, was 350 μg/kg (Long and Morgan 1990).

The recommended **PEL of pyrene is 1900 μg/kg**. Adverse biological effects were usually observed (89.7% of the data entries) when concentrations of pyrene were within the probable effects range (i.e., ≥ 1900 μg/kg). The Puget Sound AETs were consistently above the recommended PEL, ranging from 2600 μg/kg (for Microtox) to 16000 (for benthic community and amphipods; Bellar *et al.* 1986; PTI 1988). The recommended PEL is similar to the ER-M of 2200 μg/kg reported by Long and Morgan (1990).

Sum High Molecular Weight PAHs

The group of high molecular weight (HMW) PAHs considered in the present study consists of benz(a)anthracene, benzo(a)pyrene, chrysene, dibenzo(a,h)anthracene, fluoranthene, and pyrene. Due to similarities in their mode of action and toxic effect levels, these substances are frequently considered together in sediment quality assessments (Gillam 1991). Evaluation of available information on the effects of HMW PAHs on aquatic biota results in a recommended **NOEL of 870 μg/kg** and a **PEL of 8500 μg/kg**. Adverse biological effects were occasionally (15.4% of data entries) and usually (76.2% of data entries) with the no effects and probable effects ranges, respectively. By comparison, the California AETs ranged from 1700 to > 11,000 μg/kg (Becker *et al.* 1990). The Puget Sound AETs were much higher, ranging from 17,000 to 69,000 μg/kg (Beller *et al.* 1986; PTI 1988).

Total PAHs

Total PAHs refers to the sum of the concentrations of each of the 13 low and high molecular weight PAHs listed in the previous sections. While the mode of action of LMW and HMW PAHs is thought to differ (MacDonald *et al.* 1992), these substances are sometimes grouped in assessments of sediment quality (Gillam 1991). Evaluation of available information on the effects of sediment-sorbed PAHs (total) on aquatic biota results in a recommended **NOEL of 2900 mg/kg** and a **PEL of 28,000 mg/kg**. Within the no effects range, adverse biological effects were infrequently observed (6.7% of data entries). However, adverse biological effects were usually observed (88% of data entries) when contaminant concentrations were within the probable effects range. By comparison, the northern California AET for amphipods was > 15,000 μg/kg.

6.2.3 Polychlorinated Biphenyls (PCBs)

Polychlorinated biphenyls (PCBs) is the generic term for a group of 209 congeners that contain a varying number of substituted chlorine atoms in a biphenyl ring. Commercially, PCBs are used in complex mixtures, based primarily on the percentage of chlorine in the mixture. Mixtures containing 21 - 54% chlorine by weight have been used extensively in closed electric systems as dielectric fluids. Other PCBs have been used as plasticizers, heat transfer fluids, hydraulic fluids, fluids in vacuum pumps and compressors, lubricants, wax extenders, special adhesives, and surface coatings for carbonless copy paper (Moore and Walker 1991). However, all of these uses were curtailed in the United States in 1971.

Contamination of aquatic ecosystems by PCBs has arisen exclusively from human activities. While PCBs may enter the environment from a variety of sources, the major inputs to aquatic systems include leachates from landfills, municipal wastewater effluents, industrial effluents, atmospheric deposition (due to incomplete incineration of PCB contaminated wastes), and disposal of industrial and municipal wastewater treatment sludges (Moore and Walker 1991).

PCBs are highly persistent, stable compounds, which have high octanol/water partition coefficients. As such, sorption to sediments is a predominant environmental fate process in aquatic systems (Jaagumagi 1990a). PCBs tend to be associated with fine grained particles (< 0.15 μm) and organic matter in sediments. As is the case with many non-polar organic contaminants, the bioavailability of PCBs is dependent on the TOC content of the sediments (Bolton *et al.* 1985; Lyman *et al.* 1987).

Exposure to PCBs may result in a wide variety of effects on aquatic organisms, including acute and chronic lethality, reproductive toxicity, developmental abnormalities, and growth retardation (Moore and Walker 1991). While PCBs are not highly toxic to aquatic organisms, these substances have considerable potential to accumulate in the tissues of aquatic species and, therefore, may represent significant hazards to consumers of aquatic species. Bioaccumulation factors for PCBs have ranged as high as 4.4 x 10^7 in laboratory studies and biomagnification in higher trophic levels has been demonstrated (Moore and Walker 1991).

Evaluation of the available information on the toxicity of sediment-sorbed total PCBs to aquatic biota results in a recommended **NOEL of 24 µg/kg.** Adverse biological effects were occasionally observed (21.4% of data entries) when concentrations of total PCBs were within the no effects range (i.e., 0 - 24 µg/kg). The recommended NOEL was similar to the national screen level concentration of 36.6 µg/kg (at 1% TOC) reported by Neff *et al.* (1986) and the Burrard Inlet sediment quality objective (Swain and Nijman 1991). The ER-L, calculated using the NSTPA, was 50 µg/kg (Long and Morgan 1990).

The recommended **PEL of total PCBs is 260 µg/kg.** Adverse biological effects were frequently observed (54.3% of the data entries) when concentrations of total PCBs were within the probable effects range (i.e., ≥ 260 µg/kg). The frequency of adverse biological effects with the probable effects range was greater (73.7% of the data entries) when the PEL was estimated at 500 µg/kg. The recommended PEL (270 µg/kg) was similar to the northern California AET for amphipods (260 µg/kg) and the California AET for benthic species (360 µg/kg; Becker *et al.* 1990). The Puget Sound AETs were generally well above the recommended PEL, ranging from 130 µg/kg (Microtox) to 3100 (for amphipods; Bellar *et al.* 1986; PTI 1988). The recommended PEL is somewhat lower than the ER-M of 400 µg/kg reported by Long and Morgan (1990).

6.2.4 Pesticides

A wide variety of pesticides are used in agricultural and other applications throughout Florida. A list of the substances of greatest concern with respect to contamination of coastal zone sediments is provided in Table 2. These substances were identified based on historic and current use patterns (i.e., > 100,000 pounds applied in Florida annually), physical/chemical properties (i.e., log K_{ow}), and existing sediment quality monitoring data (Long and Morgan 1990; Long *et al.* 1991; Delfino *et al.* 1991).

Sufficient toxicological data exist to develop SQGs for only a subset of the priority pesticides used in Florida. Additional information will be required to support the derivation of guidelines for the other priority pesticides in Florida coastal waters.

Aldrin/Dieldrin

Aldrin is an organochlorine pesticide that has been used as a pest control agent in a variety of domestic and agricultural applications (Jaagumagi 1990b). Originally, aldrin was used to control a broad spectrum of soil, fruit, and vegetable pests, as well as for specific control of grasshoppers, locusts, and termites (CCREM 1987). However, the current uses of aldrin are restricted to those situations where there is no effluent discharge (i.e., ground injection for termite control; CCREM 1987). In aquatic systems, aldrin is rapidly biotransformed (through epoxidation) to dieldrin, which is highly stable in aquatic environments.

Like aldrin, dieldrin is an organochlorine pesticide. Dieldrin that has been one of the most widely used domestic pesticides in the United States (CCREM 1987), primarily to control soil, fruit, and vegetable pests. As is the case with aldrin, dieldrin use is currently restricted to situations where there is no effluent discharge (CCREM 1987). Sorption to sediments is an important environmental fate process for dieldrin. In sediments, this substance may persist for extended periods. Dieldrin has been detected in coastal sediments at a number of locations throughout Florida (Long and Morgan 1990).

Insufficient data were available to develop SQGs for either aldrin or dieldrin. The San Francisco Bay AETs for bivalves and amphipods was 1.9 μg/kg of aldrin (Long and Morgan 1990). In California, the AET for benthic species was 6.2 μg/kg of dieldrin (Becker *et al.* 1990).

Azinophosmethyl

Insufficient data were available to develop SQGs for azinophosmethyl, which is also known as guthion.

Total Chlordane

Chlordane is a broad spectrum chlorinated hydrocarbon pesticide that occurs as a mixture of isomers, the most common of which are alpha-chlordane and gamma-chlordane (Jaagumagi 1990b). Chlordane has been used in a wide variety of agricultural and domestic applications in Florida. Specifically, it has been used as a wood preservative, as an insecticide in home and garden applications, and to control pests on livestock (Worthing and Hance 1991). While the use of this compound has been discontinued in recent years, its persistence and tendency to accumulate in sediments makes chlordane an ongoing concern in Florida sediments. This substance has been detected in coastal sediments in various locations in the state (Long and Morgan 1990).

Insufficient data were available to develop SQGs for chlordane. Long and Morgan (1990) reported an ER-L and ER-M of 0.5 μg/kg and 6 μg/kg, respectively. The San Francisco Bay AET for bivalves and amphipods was 2 μg/kg (Long and Morgan 1990).

Chlorthalonil

Insufficient data were available to develop SQGs for chlorthalonil.

Chlorpyrifos

Insufficient data were available to develop SQGs for chlorpyrifos.

DDT and metabolites

DDT or 1,1,1-trichloro-2,2-bis(4-chlorophenyl)ethane is a broad spectrum organochlorine insecticide that has been used worldwide since the early 1940s (Jaagumagi 1990b). DDT has been used extensively in agricultural applications, primarily as a non-systemic ingested and contact insecticide to control a wide variety of pest species (Worthing and Hance 1991). While this substance is no longer registered for use in North America, it is highly toxic and persistent in the environment. Therefore, residues of DDT and its metabolites (DDE and DDD) may represent significant sediment quality concerns in Florida. DDT, DDE, and DDD have all been detected recently in Florida coastal sediments (DelFino *et al.* 1991; Long and Morgan 1990).

p,p'-DDD

Insufficient data were available to develop SQGs for p,p'-DDD. An ER-L of 2 μg/kg was reported by Long and Morgan (1990). The ER-M was 20 μg/kg (Long and Morgan 1990).

p,p'-DDE

Evaluation of the available information on the toxicity of sediment-sorbed total p,p'-DDE to aquatic biota results in a recommended **NOEL of 1.7 μg/kg.** Adverse biological effects were never observed when concentrations of total p,p'-DDE were within the no effects range (i.e., 0 - 1.7 μg/kg). The recommended NOEL was similar to the San Francisco Bay AET for bivalves and mussels (2.2 μg/kg; Long and Morgan 1990). The ER-L, calculated using the NSTPA, was 2 μg/kg (Long and Morgan 1990).

The recommended **PEL of total p,p'-DDE is 130 μg/kg.** Adverse biological effects were commonly observed (52.6% of the data entries) when concentrations of total p,p'-DDE were within the probable effects range (i.e., ≥ 130 μg/kg). The recommended PEL was much higher than the Puget Sound AETs for benthic species (9 μg/kg) and amphipods (15 μg/kg; PTI 1988) and much lower than the California AET for benthic species (2800 μg/kg; Becker *et al.* 1990). The recommended PEL is much higher than the ER-M of 15 μg/kg reported by Long and Morgan (1990).

p,p'-DDT

Insufficient data were available to develop SQGs for p,p'-DDT. The chronic marine sediment quality criterion, derived using the EqPA, was 1.5 (at 1% TOC; Bolton *et al.* 1985). An ER-L of 1 μg/kg was reported by Long and Morgan (1990).

In California, AETs for p,p'-DDT ranged from 6.3 μg/kg for benthic species to > 620 μg/kg for amphipods (Becker *et al.* 1990). The Puget Sound AET for amphipod was > 270 μg/kg (PTI 1988). Long and Morgan (1990) reported an ER-M of 7 μg/kg for this substance.

Total DDT

Evaluation of the available information on the toxicity of sediment-sorbed DDT (total) to aquatic biota results in a recommended **NOEL of 4.5 μg/kg.** Adverse biological effects were frequently observed (52.6% of data entries) when concentrations of total DDT were within the no effects range (i.e., 0 - 4.5 μg/kg). The recommended NOEL was similar to the chronic marine sediment quality criterion of 3.29 μg/kg recommended by JRB Associates (1984). The ER-L, calculated using the NSTPA, was 3 μg/kg (Long and Morgan 1990).

The recommended **PEL of total DDT is 270 μg/kg.** Adverse biological effects were commonly observed (52.6% of the data entries) when concentrations of total DDT were within the probable effects range (i.e., ≥ 270 μg/kg). The California AETs for benthic species (3000 μg/kg) and for amphipods (> 9300 μg/kg) were much higher than the PEL (Becker *et al.* 1990). The recommended PEL was slightly lower than the ER-M of 350 μg/kg reported by Long and Morgan (1990).

Disulfoton

Insufficient data were available to develop SQGs for disulfoton.

Endosulfan

Insufficient data were available to develop SQGs for endosulfan. McLeese *et al.* (1982) reported a 12 day LC_{50} of 340 μg/kg for the sandworm, *Nereis virens.* Chandler *et al.* (1991) reported effects on colonization of polychaetes in Southern California at 50 μg/kg and mortality to copepods at 200 μg/kg.

Endrin

Insufficient data were available to develop SQGs for endrin. Chronic marine sediment quality criteria, calculated using the EqPA, ranged from 0.53 to 3.21 μg/kg (EPA 1988; JRB Associates 1984).

Heptachlor

Insufficient data were available to develop SQGs for heptachlor. The chronic marine sediment quality criterion, calculated using the EqPA, was 5 μg/kg (Bolton *et al.* 1985).

Heptachlor Epoxide

Insufficient data were available to develop SQGs for heptachlor epoxide.

Lindane (gamma-BHC)

Insufficient data were available to develop SQGs for lindane. In California, AET values for lindane ranged from 0.7 (for amphipods) to > 1.3 (for benthic species; Becker *et al.* 1990).

Mirex

Insufficient data were available to develop SQGs for mirex.

Phorate

Insufficient data were available to develop SQGs for phorate.

Toxaphene

Insufficient data were available to develop SQGs for toxaphene. Bolton *et al.* (1985) reported a chronic marine sediment quality criterion of 5 μg/kg for this substance.

Trifluralin

Insufficient data were available to develop SQGs for trifluralin.

6.2.5 Chlorinated Organic Substances

Dioxins and Furans

Polychlorinated dibenzo-p-dioxins (PCDDs) are composed of a triple-ring structure consisting of two benzene rings connected to each other by two oxygen atoms. Depending on the number and position of chlorine substitution on the benzene rings, 75 chlorinated dioxin congeners are possible. The polychlorinated dibenzofuran (PCDF) molecule is also a triple-ring structure with the two benzene rings connected to themselves by a single oxygen atom (Figure 1). One hundred and thirty-five (135) chlorinated dibenzofuran congeners are possible.

Sources and releases to the environment have been well documented in the literature (OMOE 1985; Hutzinger *et al.* 1985; EPS 1985; EPA 1985; NRCC 1981; NRCC 1984). PCDDs and PCDFs are not produced intentionally but are unavoidable by-products of chemical manufacturing or the result of incomplete combustion of materials containing chlorine atoms and organic compounds (OMOE 1985). PCDDs and PCDFs may also be formed during the disinfection of complex effluents (e.g. pulp and paper effluents) containing many organic constituents.

Dibenzodioxins and dibenzofurans have the potential to enter the aquatic environment due to direct effluent discharges, runoff from areas in which dioxin/furan contaminated products are used and stored, and deposition of materials that are transported atmospherically. The most significant sources of dioxins include the wood preservative pentachlorophenol, municipal incinerators, and pulp and paper mills that utilize chlorine in the bleaching process. Polychlorinated biphenyls (PCBs) are the most significant source of furans (Boddington *et al.* 1990).

PCDDs and PCDFs may be distributed throughout the environment via air, water, soil, and sediments. PCDDs and PCDFs tend to be very insoluble in water, adsorb strongly onto soils, sediments, and airborne particulates, and bioaccumulate in biological tissues (Hutzinger *et al.* 1985). These substances have been associated with a wide variety of toxic effects in animals, including acute toxicity, enzyme activation, tissue damage, developmental abnormalities, and cancer.

Insufficient toxicological data are available to derive SQGs for any of the 75 dioxin or furan congeners that could be present in Florida coastal sediments.

Pentachlorophenol

Insufficient data were available to develop SQGs for pentachlorophenol. In Puget Sound, AETs of 360 and 690 µg/kg have been reported for amphipods and benthic species, respectively (PTI 1988).

6.2.6 Phthalate Esters

Phthalate esters represent a large group of chemicals that are used widely as plasticizers in polyvinyl chloride (PVC) resins, adhesives, and cellulose film coatings. They are also found in cosmetics, rubbing alcohol, insect repellents, insecticides, and solid rocket propellants

(CCREM 1987). Due to their wide use, phthalate esters have a significant potential to be released into coastal ecosystems. For this reason, numerical SQGs for these substances are required to assess the hazards posed to aquatic organisms.

Bis(2-ethylhexyl)phthalate

Insufficient data were available to develop SQGs for bis(2-ethylhexyl)phthalate. Puget Sound AETs ranged from 1300 μg/kg (benthic community) to > 3100 μg/kg (amphipods) for this substance (PTI 1988; Bellar *et al.* 1986). Becker *et al.* (1990) reported that the California AET for benthic species was 5100 μg/kg.

Dimethyl phthalate

Insufficient data were available to develop SQGs for dimethyl phthalate. Puget Sound AETs ranged from 71 μg/kg (Microtox) to > 160 μg/kg (amphipods and bivalves) for this substance (PTI 1988; Bellar *et al.* 1986). Bolton *et al.* (1985) reported a chronic marine sediment quality criterion of 490 μg/kg, using the EqPA.

Di-n-butyl phthalate

Insufficient data were available to develop SQGs for di-n-butyl phthalate. Puget Sound AETs ranged from 1400 μg/kg (Microtox and oysters) to > 5100 μg/kg (benthic species) for this substance (PTI 1988; Bellar *et al.* 1986).

Chapter 7

An Initial Assessment of the Potential for Biological Effects of Sediment-Sorbed Contaminants in Florida Coastal Waters

7.0 Introduction

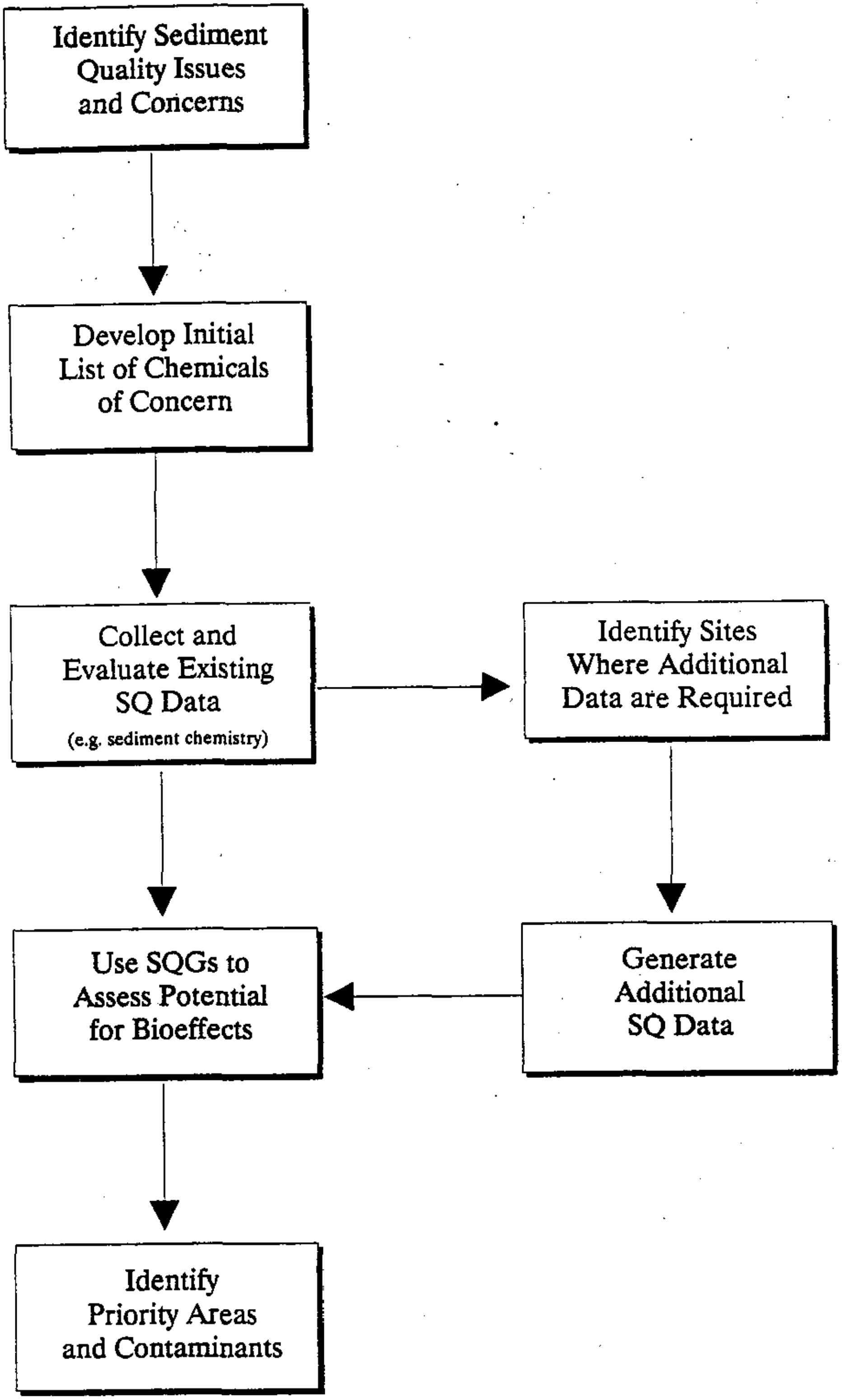
This Chapter describes an initial assessment of the potential for biological effects of sediment-sorbed contaminants, using Florida Department of Environmental Regulation (FDER) coastal sediment chemistry data and the sediment quality guidelines (SQGs) identified in Chapter 6. This initial assessment will help focus sediment management efforts by identifying priority contaminants and priority sites with respect to sediment contamination. Effective identification of sediment quality concerns in Florida will help direct limited resources to yield the greatest environmental benefits.

This initial regional assessment of sediment quality consisted of four steps. First, regional sediment quality issues and concerns were identified by reviewing potential sources of contaminants in the state. Priority substances with respect to sediment contamination were subsequently identified by integrating relevant data from a number of sources. Next numerical SQGs were then derived preferentially for those substances that were likely to be of greatest concern in Florida sediments. The third step was to compile a database containing sediment chemistry data for Florida coastal waters. Finally, a comparison of sediment chemistry data with the SQGs was done to provide a preliminary means of identifying priority sites and priority contaminants with respect to the potential for adverse biological effects (Figure 4).

7.1 Identification of Regional Sediment Quality Issues and Concerns

In Florida, sediment quality issues and concerns are primarily associated with direct and non-point (diffuse) source inputs of contaminants from urban and suburban areas into coastal waters. These inputs of contaminants include effluent discharges from wastewater treatment plants, stormwater runoff, and a variety of related sources. In addition, industrial facilities have the potential to release significant quantities of contaminants into estuarine and marine systems with the most significant of these being the pesticides, organic chemicals, plastics and pulp and paper industries. Further, intensive agricultural operations in the state have the potential to contribute pesticides and fertilizers to aquatic ecosystems. Other possible sources of contaminants into Florida coastal waters include leachates from landfills, dredge and fill activities, and the operation of ships and pleasure craft. Each of these potential sources of contaminants was considered in identifying substances for this preliminary evaluation. (A discussion of sediment quality issues and concerns and

Figure 4. Framework for conducting preliminary regional sediment quality assessment of Florida coastal waters.



uncertain and is likely to be dependent on such factors as bioavailability, which may influence the toxicity of the substance (NOEL < possible effects range < PEL). Sediment-sorbed contaminants are considered to represent *potential* hazards to exposed organisms when concentrations fall within this range. Sediments with concentrations of contaminants within this range may require further assessment to determine the biological significance of the contamination. In general, further assessment would be supported by biological tests designed to evaluate the biological significance of sediment-sorbed contaminants to key species of aquatic biota.

The no effects range is defined as the range of concentrations of a sediment contaminant within which biological effects are rarely or never observed (no effects range ≤ NOEL). Sediments with concentrations of contaminants within the no effects range are considered to be of *acceptable quality* for those contaminants. In general, further investigations of sediment quality conditions would be considered to be of relatively lower priority for sediments in which contaminant concentrations fall within the no effects range. However, biological testing may be required to validate the results of the initial assessment of the potential for adverse biological effects.

7.4 Assessment of the Potential for Biological Effects of Sediment-Sorbed Contaminants

Sediment chemistry data were used in conjunction with the recommended SQGs to conduct an initial assessment of the potential for adverse effects in the Florida coastal zone. This assessment was conducted through a search of the FDER coastal sediment chemistry database, using the SQGs as search criteria. In this way, data entries that exceeded the probable effects level and the no effects level, respectively, could be identified. The highest priority substances with respect to sediment contamination were identified as those that frequently occurred at concentrations within the probable effects ranges. The highest priority area with respect to sediment contamination were identified as those with the greatest frequency of contaminant concentrations within the probable effects ranges. Pooled data for a number of sampling stations and sampling dates were used to assess sediment quality within each geographic area.

7.4.1 Areas of Concern in Florida Coastal Waters

A total of 21 areas were considered in this initial assessment of sediment quality in Florida. Evaluation of FDER coastal sediment chemistry database provides a great deal of insight into sediment quality conditions within each of these areas. However, this initial assessment is constrained by limitations on the available data for some areas. For example, data on levels of metals were available on less than ten sites in the Jupiter, Ft. Lauderdale and Florida Keys areas. Even more severe limitations on the data were apparent when PAHs, PCBs, pesticides and other organic contaminants were considered (see Tables 6-9). In spite of these limitations, it is apparent that sediment quality represents a significant environmental concern in a number of locations within the state.

Table 6. Number of samples that fall within the probable effects range (i.e., > PEL) of contaminant concentrations for each Atlantic coast sampling area (continued).

	Number of Observations Within the Probable Effects Range								
Substance	JAX	STA	DAY	IRS	JPT	WPB	FTL	MIA	KEY
Chlorinated Organic Compounds									
2,3,7,8-Tetrachlorodibenzo-p-dioxin	NG	NG	NG	NG	NG	NG	NG	NG	NG
2,3,7,8-Tetrachlorodibenzofuran	NG	NG	NG	NG	NG	NG	NG	NG	NG
Pentachlorophenol	NG	NG	NG	NG	NG	NG	NG	NG	NG
Pesticides									
Aldrin	NG	NG	NG	NG	NG	NG	NG	NG	NG
Azinophosmethyl (Guthion)	NG	NG	NG	NG	NG	NG	NG	NG	NG
Chlordane	NG	NG	NG	NG	NG	NG	NG	NG	NG
Chlorthalonil	NG	NG	NG	NG	NG	NG	NG	NG	NG
Chlorpyrifos	NG	NG	NG	NG	NG	NG	NG	NG	NG
p,p'-DDD	NG	NG	NG	NG	NG	NG	NG	NG	NG
p,p'-DDE	0	0	0	0	0	0	0	0	0
p,p'-DDT	NG	NG	NG	NG	NG	NG	NG	NG	NG
Total DDT	0	0	0	0	0	0	0	0	0
Dieldrin	NG	NG	NG	NG	NG	NG	NG	NG	NG
Disulfoton	NG	NG	NG	NG	NG	NG	NG	NG	NG
Endosulfan	NG	NG	NG	NG	NG	NG	NG	NG	NG
Heptachlor	NG	NG	NG	NG	NG	NG	NG	NG	NG
Heptachlor epoxide	NG	NG	NG	NG	NG	NG	NG	NG	NG
Lindane (gamma-BHC)	NG	NG	NG	NG	NG	NG	NG	NG	NG
Phorate	NG	NG	NG	NG	NG	NG	NG	NG	NG
Quintozene (PCNB)	NG	NG	NG	NG	NG	NG	NG	NG	NG
Toxaphene (alpha-BHC)	NG	NG	NG	NG	NG	NG	NG	NG	NG
Trifluralin	NG	NG	NG	NG	NG	NG	NG	NG	NG
Number of Samples	47	3	6	21	0	11	5	78	0

Table 7. Number of samples that fall within the possible effects range (i.e., > NOEL and < PEL) of contaminant concentrations for each Atlantic coast sampling area.

Substance	Number of Observations Within the Possible Effects Range								
	JAX	STA	DAY	IRS	JPT	WPB	FTL	MIA	KEY
Metals									
Arsenic	16	15	8	7	0	2	0	14	0
Cadmium	3	0	1	6	0	0	0	24	0
Copper	12	6	6	15	0	1	1	16	0
Chromium	28	12	3	24	0	1	1	38	0
Lead	30	2	5	21	0	5	2	27	0
Mercury	37	1	11	28	0	9	2	53	4
Nickel	NG	NG	NG	NG	NG	NG	NG	NG	NG
Silver	4	0	0	4	0	0	0	13	0
Tributyl Tin	NG	NG	NG	NG	NG	NG	NG	NG	NG
Zinc	27	4	1	17	0	2	0	20	0
Number of Samples	68	37	31	86	7	27	5	110	4
Polycyclic Aromatic Hydrocarbons (PAHs)									
Acenaphthene	5	0	0	0	0	0	0	0	0
Acenaphthylene	NG	NG	NG	NG	NG	NG	NG	NG	NG
Anthracene	5	0	0	0	0	0	2	5	0
Benz(a)anthracene	0	0	0	0	0	0	0	5	0
Benzo(a)pyrene	7	0	0	0	0	3	0	10	0
Chrysene	6	0	0	0	0	0	0	12	0
Dibenzo(a,h)anthracene	0	0	0	0	0	1	0	0	0
Fluoranthene	6	2	5	0	0	3	1	20	0
Fluorene	5	0	0	0	0	0	1	0	0
2-methyl naphthalene	NG	NG	NG	NG	NG	NG	NG	NG	NG
Naphthalene	0	0	0	0	0	1	0	2	0
Phenanthrene	1	2	4	0	0	0	0	5	0
Pyrene	7	0	1	0	0	3	0	16	0
Number of Samples	34	2	6	7	0	6	4	66	0

Table 7. Number of samples that fall within the possible effects range (i.e., > NOEL and < PEL) of contaminant concentrations for each Atlantic coast sampling area (continued).

	Number of Observations Within the Possible Effects Range								
Substance	JAX	STA	DAY	IRS	JPT	WPB	FTL	MIA	KEY
Polychlorinated Biphenyls (PCBs)									
Aroclor 1016	0	0	0	0	0	0	0	1	0
Aroclor 1242	0	0	0	0	0	0	0	0	0
Aroclor 1248	0	0	0	0	0	0	1	0	0
Aroclor 1254	2	0	0	0	0	0	2	5	0
Aroclor 1260	0	0	0	0	0	0	0	2	0
Total PCBs*	2	0	0	0	0	0	1	5	0
Number of Samples	47	3	6	21	0	11	5	78	0
Phthalates									
Bis(2-ethylhexyl)phthalate	NG	NG	NG	NG	NG	NG	NG	NG	NG
Dimethyl phthalate	NG	NG	NG	NG	NG	NG	NG	NG	NG
Di-n-butyl phthalate	NG	NG	NG	NG	NG	NG	NG	NG	NG

*Sum of Aroclor

NG = no guideline; insufficient data to derive sediment quality guidelines.

JAX = Jacksonville; STA = St. Augustine; DAY = Daytona Beach; IRS = Indian River; JPT = Jupiter; WPB = West Palm Beach; FTL = Ft. Lauderdale; MIA = Miami; KEY = Florida Keys

Table 8. Number of samples fall within the probable effects range (i.e., > PEL) of contaminant concentrations for each Gulf coast sampling area (continued).

Substance	Number of Observations Within the Probable Effects Range										
	EVG	FTM	TPA	WCF	APL	APA	SJB	PCY	DES	PEN	PER
Chlorinated Organic Compounds											
2,3,7,8-Tetrachlorodibenzo-p-dioxin	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
2,3,7,8-Tetrachlorodibenzofuran	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Pentachlorophenol	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Pesticides											
Aldrin	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Azinophosmethyl (Guthion)	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Chlordane	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Chlorthalonil	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Chlorpyrifos	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
p,p'-DDD	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
p,p'-DDE	0	0	0	0	0	0	0	0	0	0	0
p,p'-DDT	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Total DDT	0	0	0	0	0	0	0	0	0	0	0
Dieldrin	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Disulfoton	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Endosulfan	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Heptachlor	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Heptachlor epoxide	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Lindane (gamma-BHC)	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Phorate	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Quintozene (PCNB)	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Toxaphene (alpha-BHC)	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Trifluralin	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Number of Samples	3	8	24	0	0	0	5	3	3	29	7

Table 9. Number of samples that fall within the possible effects range (i.e., > NOEL and < PEL) of contaminant concentrations for each Gulf coast sampling area.

Substance	EVG	FTM	TPA	WCF	APL	APA	SJB	PCY	DES	PEN	PER
	Number of Observations Within the Possible Effects Range										
Metals											
Arsenic	6	1	17	4	6	7	5	10	4	20	7
Cadmium	0	1	42	0	1	0	0	1	0	5	0
Copper	1	1	22	8	0	3	1	3	1	5	1
Chromium	4	3	52	13	3	6	7	13	5	28	2
Lead	0	2	42	6	1	6	8	10	4	18	2
Mercury	2	8	59	10	4	2	9	9	2	20	3
Nickel	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Silver	2	0	21	0	2	0	0	10	2	0	0
Tributyl Tin	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Zinc	0	1	39	1	0	6	8	7	2	21	4
Total Number of Samples	96	67	141	30	56	30	22	39	20	79	17
Polycyclic Aromatic Hydrocarbons (PAHs)											
Acenaphthene	0	0	0	0	0	0	0	0	0	1	0
Acenaphthylene	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Anthracene	0	0	0	0	0	0	0	0	0	5	0
Benz(a)anthracene	0	0	0	0	0	0	0	0	0	1	0
Benzo(a)pyrene	0	0	0	0	0	0	0	0	0	9	2
Chrysene	0	0	0	0	0	0	0	0	0	1	4
Dibenzo(a,h)anthracene	0	0	0	0	0	0	0	0	0	0	0
Fluoranthene	0	0	0	0	0	0	0	0	0	5	0
Fluorene	0	0	0	0	0	0	0	0	0	0	0
2-methyl naphthalene	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Naphthalene	0	0	0	0	0	0	0	0	0	0	1
Phenanthrene	0	0	0	0	0	0	0	0	0	9	2
Pyrene	0	0	0	0	0	0	0	0	0	8	4
Total PAHs	0	0	0	0	0	0	0	0	0	0	0
Total Number of Samples	3	12	11	0	0	0	0	0	3	29	9

Table 9. Number of samples that fall within the possible effects range (i.e., > NOEL and < PEL) of contaminant concentrations for each Gulf coast sampling area (continued).

Substance	Number of Observations Within the Possible Effects Range EVG	FTM	TPA	WCF	APL	APA	SJB	PCY	DES	PEN	PER
Polychlorinated Biphenyls (PCBs)											
Aroclor 1016	0	0	0	0	0	0	0	0	0	0	0
Aroclor 1242	0	0	0	0	0	0	0	0	0	1	0
Aroclor 1248	0	0	0	0	0	0	0	0	0	0	0
Aroclor 1254	0	0	0	0	0	0	0	0	0	2	1
Aroclor 1260	0	0	0	0	0	0	0	0	0	0	0
Total PCBs*	0	0	0	0	0	0	0	0	0	3	1
Total Number of Samples	3	8	24	0	0	0	5	3	3	29	7
Phthalates											
Bis(2-ethylhexyl)phthalate	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Dimethyl phthalate	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG
Di-n-butyl phthalate	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG	NG

*Sum of Aroclor

NG = no guideline; insufficient data to derive sediment quality guidelines.

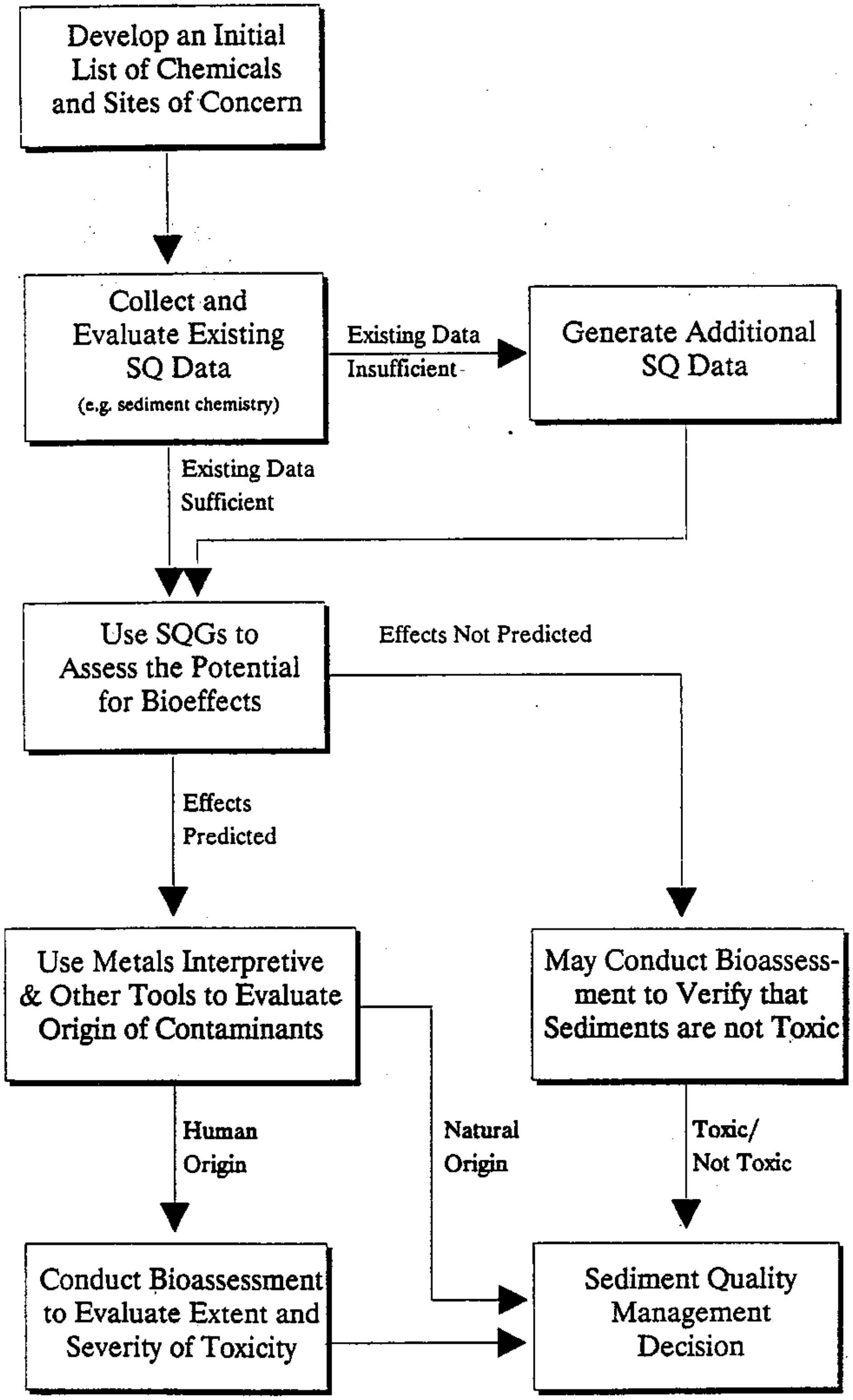
EVG = Everglades; FTM = Ft. Mayers; TPA = Tampa Bay; WCF = West Central Florida; APL = Apalachee Bay; APA = Apalachichola Bay; STJ = St. Josephs Bay; PCY = Panama City; DES = Destin; PEN = Pensacola Bay; PER = Perido Bay.

fluoranthene, fluorene, and phenanthrene. The concentrations of these contaminants fell within the probable effects range on more than one occasion in Florida coastal sediments. Insufficient data were available to assess the potential for biological effects associated with levels of nickel, tributyltin, acenaphthylene, 2-methylnaphthalene, dioxins and furans, pentachlorophenol, 11 individual pesticides, and three individual phthalates.

7.4.3 Limitations of the Initial Assessment of Sediment Quality in Florida

While this initial assessment of sediment quality provides an initial indication of the potential for biological effects of sediment-sorbed contaminants in Florida, these results should *not* be used, by themselves, to make management decisions regarding sediment quality. Several limitations of this assessment are identified to emphasize this point. The sediment chemistry database used in this assessment has broad coverage, however, the data on many analytes are limited. Much of the data on levels of organic contaminants is relatively old (greater than 5 years old) and therefore of questionable value with respect to reflecting present conditions. In addition, the data collected by Delfino *et al.* (1991) and by NOAA (NSTP) should be evaluated to provide a more comprehensive assessment of sediment quality.

Figure 5. Framework for conducting site-specific assessments of sediment quality conditions in Florida.



to determining the applicability of the data. For example, natural degradative processes in the environment can lead to reductions in the concentrations of sediment-sorbed organic contaminants over time (Mosello and Calderoni 1990). In addition, major events (such as storms) may result in the transport of sediments between sites. Further, industrial developments and/or regulatory activities may alter the sources and composition of contaminants released into the environment. Therefore, it is important that assessments of sediment quality be undertaken with the most recent data available.

In addition to temporal variability, the chemistry of bed sediments is known to vary significantly on a spatial basis (Florida Department of Environmental Regulation; FDER. In preparation; Mah *et al.* 1989). Therefore, any single sample is likely to represent only a small proportion of the geographic area in which it was collected. For this reason, data from a number of stations are required to provide a representative picture of sediment quality conditions at the site, with the actual number of stations required dependent on the size of the area under consideration, the concentrations of sediment-sorbed contaminants, and the variability of contaminant concentrations.

Another important factor to consider in evaluating the applicability of existing sediment quality data is the list of variables that were analyzed. It is important that the list of analytes reflect potential contaminant sources from land and water use activities in the area. For example, in harbors, variables such as pentachlorophenol (which is used as a preservative for pilings), tributyltin (which is used in antifouling paints for ships), and copper (which is used in antifouling paints for pleasure craft) should be measured. Similarly, polycyclic aromatic hydrocarbons and lead should be measured in the vicinity of oil exploration, extraction, transport, or refining operations and storm sewers that collect urban runoff (especially from roads). In agricultural areas, persistent pesticides and nutrients should be considered in sediment quality assessments.

If the results of the data evaluation process indicate that the sediment chemistry data are acceptable, it is possible to proceed with the preliminary assessment of the potential for biological effects of sediment-sorbed contaminants. However, if the sediment chemistry data are considered to be of unacceptable quality or are not considered to adequately represent the site, additional sediment chemistry data may be required to complete the sediment quality assessment.

8.3 Collect Supplemental Sediment Chemistry Data

The third stage in the sediment quality assessment process involves the generation of supplemental sediment chemistry data. Additional testing of subject sediments may be required when existing data are of insufficient quality or quantity to support the assessment of sediment quality at a specific site. The initial list of chemical concerns for the site under consideration provides a defensible means of identifying a list of potential analytes for inclusion in the sediment quality monitoring program.

conditions would be considered to be of relatively low priority for sediments in which contaminant concentrations fall within the no effects range. However, biological testing may be required to validate the results of the preliminary assessment of the potential for adverse biological effects (particularly in sediments with low levels of TOC, AVS, and/or other variables that could influence the bioavailability of sediment-sorbed contaminants).

8.5 Evaluate Natural vs. Anthropogenic Sources of Sediment-Sorbed Contaminants

In the past, determination of whether estuarine and coastal sediments were anthropogenically enriched with metals had been a difficult process that required comprehensive, site-specific assessments. However, the FDER (Schropp and Windom 1988; Shropp *et al.* 1990) has developed a practical approach for assessing metals contamination in coastal sediments. This procedure relies on normalization of metal concentrations to a reference element. In the case of Florida, normalization of metal concentrations to concentrations of aluminum in estuarine sediments provided the most promising method of comparing metal levels on a regional basis.

Briefly, data on sediment metal concentrations were collected from roughly 100 sites which were thought to be representative of natural estuarine areas throughout Florida. Simple linear regressions of each of seven metals on aluminum were performed on log-transformed data and 95% prediction limits were calculated. Significant correlations were obtained for arsenic, cadmium, chromium, copper, lead, nickel, and zinc. The regression lines and prediction limits were plotted. These plots then formed the basis for interpreting data on the concentrations of metals in sediments, such that anthropogenic enrichment of metal levels would be suspected at sites with metals concentrations exceeding the upper 95% prediction limit (for one or more substances). An evaluation of this procedure using data from Tampa Bay (Schropp *et al.* 1989) confirmed the effectiveness and utility of this interpretive tool.

The metals interpretive tool provides an effective means of identifying sites that are anthropogenically enriched with metals. As such, this tool provides a basis for further refining the list of priority substances and priority sites in Florida. While no equivalent tool exists for evaluating the origin of many organic substances, a considerable number of organic contaminants are released in the environment only as a result of human activities. Therefore, the development of a comparable interpretive tool may not be as critical as for metals. Substances that fall into this category include chlorophenols (and related compounds), PCBs, pesticides, dioxins and furans, phthalates, and a host of other compounds. There are several methods that can be used to fingerprint the origin of PAHs. The ratios of the concentrations of some hydrocarbons or groups of hydrocarbons can be examined to distinguish between storm runoff, oil spills, and other sources.

Grandidierella japonica). The marine document is being revised to include an additional east coast amphipod, *Leptocheirus plumulosus*. These bioassays may be modified to assess toxicity to other benthic invertebrate species that occur in estuarine and marine environments, including other amphipods, other crustaceans, polychaetes, and bivalves (ASTM 1990a). In addition, procedures for conducting sediment toxicity tests with polychaetes and echinoderms are currently under consideration by the ASTM (Ingersoll 1991).

In addition to whole sediment toxicity tests, various procedures are available for assessing the potential for adverse effects on aquatic organisms due to the resuspension of sediments or partitioning of contaminants into the water column. Perhaps the most sensitive and frequently used of these is the bacterial luminescence test (Microtox; Burton and Stemmer 1988; Schiewe *et al.* 1985). Tests using algae, invertebrates, and fish have also been employed to assess the toxicity of the suspended and/or aqueous phases. While no standard methods have yet been approved by the ASTM, a document on the use of oyster and echinoderm embryos and larvae in sediment toxicity testing of marine sediments is currently in preparation (Ingersoll 1991). In addition, procedures for conducting water column bioassays and bioaccumulation tests have been recommended by the EPA and ACE (1991) and Lee *et al.* (1989) and document on sediment resuspension testing is under consideration by ASTM.

While requirements for biological tests differ between applications, sediment toxicity tests should follow the general protocols established and approved by the ASTM. These protocols may be modified to assess toxicity to resident species, for longer time periods (i.e., to address chronic toxicity), or for different endpoints, however, the basic principles of these protocols should be followed. When ASTM methods do not exist or do not apply, care should be taken and documented to ensure that the experimental design of these tests is defensible.

Other types of biological information may also be used in the sediment quality assessment process. For example, comparison of biological indicators (such as the diversity and abundance of benthic invertebrate communities) at test sites and appropriate reference sites (i.e., sites with similar particle size distributions, TOC, etc.) provides a means of assessing the relative toxicity of test sediments. Various statistical procedures may be used to help identify contaminants associated with observed biological effects when adequate sediment chemistry data are available. In addition, spiked-sediment bioassays may be used to establish cause and effect relationships for specific substances or mixtures of contaminants. Further, tests to evaluate the toxicity of pore water provide information which may be used to identify the toxic elements of contaminated sediments. Information on levels of contaminants in aquatic biota and on bioaccumulation may help determine the significance of contaminant levels in sediments relative to the protection of human health and the health of wildlife that consume aquatic organisms.

Chapter 9

Summary and Recommendations

9.1 Summary

This report describes and evaluates preliminary chemical sediment quality assessment guidelines (SQAGs) for Florida coastal waters. It also provides an initial evaluation of contaminated sediments and a framework for applying the guidelines.

In Florida, conservation and protection of natural resources has been identified as a high priority environmental management goal. Realization of this goal requires protection of living resources and their habitats in estuarine, nearshore, and marine ecosystems. In the last decade, there has been a significant increase in the level of scientific understanding (and public recognition) of the important role sediments play in coastal ecosystem functions. Sediments are particularly critical in determining the fate and effects of environmental contaminants.

Recent monitoring data indicate that concentrations of various contaminants are present at elevated levels at a number of locations in Florida coastal sediments. These data emphasize the need for sediment quality guidelines (SQGs) to evaluate the potential for biological effects associated with sediment-sorbed contaminants and to provide assistance in managing coastal resources.

To identify an appropriate procedure for deriving SQGs, the major approaches used in other jurisdictions to derive numerical SQGs were evaluated in the context of Florida's requirements for sediment quality assessment values. The results of this analysis indicated that the National Status and Trends Program Approach (NSTPA; Long and Morgan 1990) would respond most directly to Florida's requirements. Therefore, a strategy that relied on a modified version of the NSTPA was recommended to derive numerical SQGs that could be used immediately to assess sediment quality issues and concerns. A critical evaluation of this procedure suggested that, while this approach has limitations that could influence the applicability of the guidelines, it is likely to support the derivation of scientifically defensible preliminary guidelines for Florida coastal waters.

Preliminary SQGs have been developed for 25 priority contaminants in Florida coastal waters. However, insufficient data were available to derive guidelines for another 29 substances that are known or are suspected to contaminate Florida coastal sediments. The numerical SQGs were used to define three ranges of concentrations for each of the 25 contaminants: a probable effects range; a possible effects range; and, a no effects range. A subjective assessment of the credibility of these guidelines indicated that a high level of confidence could be placed on the guidelines derived for 11 substances, and a moderate or low level of confidence could be placed on the guidelines for the remaining 14 substances.

for additional priority substances (for which insufficient data are currently available) identified in Florida coastal sediments.

Additional biological testing is recommended to support the sediment quality assessment process in Florida. In particular, data from toxicological studies conducted with Florida sediments are required to evaluate the applicability of the preliminary SQGs to Florida coastal ecosystems. In this respect, additional biological testing is required to determine if there are systematic differences between the sensitivities of species represented in the existing database compared to the sensitivities of resident species of Florida coastal waters. These data may also be used to assess the bioavailability of contaminants in Florida coastal sediments.

The relative sensitivity of species that occur in Florida is a central consideration in the evaluation of the applicability of the preliminary SQGs. The SQGs recommended for assessing the potential for biological effects of sediment-sorbed contaminants in Florida were developed using data on a wide variety of species that occur in North America. However, biological effects data on aquatic organisms from the southeastern portion of the United States were limited. Therefore, it is difficult to determine if the recommended SQGs would adequately protect aquatic organisms that occur in Florida coastal waters. For this reason, additional biological testing should be undertaken to determine if aquatic organisms that occur in Florida have sensitivity ranges similar to those of organisms occurring in other parts of North America.

Bioavailability is a central issue in the evaluation of the preliminary SQGs. Many types of sediments occur in Florida coastal ecosystems, ranging from terrigenous sediments in the northern portion of the Gulf coast to carbonate sediments in some areas of south Florida. There is significant potential for differences in the bioavailability (and hence the toxicity) of contaminants in these different sediment types. Although the information used to derive the preliminary SQGs includes data from a wide variety of sites in North America, it is possible that these data do not adequately represent the full range of conditions that occur in Florida. Therefore, further biological testing should be conducted at a variety of locations in Florida to determine if the recommended SQGs are appropriate for Florida coastal waters. These locations should be selected to encompass a wide range of sediment types, and should include contaminated and uncontaminated reference sites.

The preliminary guidelines are based on dry weight-normalized contaminant concentrations. However, there is an increasing body of information which suggests that toxicity can be predicted more accurately when concentrations of various 'normalizers' (such as total organic carbon and acid volatile sulfide) are considered. Therefore, there is a need to generate additional data to define bioavailability relationships for individual contaminants, and refine the guidelines appropriately when these relationships become more clearly established.

Sediment quality criteria are currently under development by EPA (using the EqPA). These criteria are likely to be expressed in terms of the variables that influence the bioavailability of sediment-sorbed contaminants. These criteria should be fully evaluated and used, as

9.2.4 *Site-Specific Assessment of Sediment Quality*

The recommended approach for assessing sediment quality in Florida relies on the identification of three ranges of contaminant concentrations: the no effects range; the possible effects range; and, the probable effects range. This approach was selected to explicitly account for the uncertainties associated with the evaluation of the available data which link contaminant concentrations with adverse biological effects. When contaminant concentrations fall within the probable effects range at a particular site, there is a high probability that adverse biological effects will be observed. These sites should be given highest priority for further investigations.

Effects-based SQGs should not be used alone to make contaminated sediment management decisions. Ancillary tools, such as the metals interpretive tool, should be used to determine the probable origin of sediment-sorbed contaminants. In addition, uncertainty regarding the potential for biological effects of sediment-sorbed contaminants at specific locations may be addressed by implementing appropriate biological investigations. These tools, when used together, will provide an efficient and effective basis for making contaminated sediment management decisions.

9.2.5 *Coordination with Federal Agencies*

Currently, there are a relatively large number of independent and loosely-related initiatives that are directed at the evaluation and management of contaminated sediments. While each of these programs are designed to advance our understanding of the nature, extent, and severity of sediment contamination, development of a regional strategy for contaminated sediment identification and management would accelerate this process. Therefore, a cooperative regional strategy should be developed by FDER, EPA, the Army Corps of Engineers, and other affected agencies to identify priority sediment management and regulatory objectives, and the interagency efforts required to achieve them.

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Appendix 1. Screening Criteria for Evaluating Candidate Data Sets for the Sediment Toxicity (SEDTOX) Database

B. *Matching Sediment Chemistry and Biological Effects Data*

1. The data set must contain matching sediment chemistry and biological effects data. That is, biological and chemical data must be collected from the same locations and at the same time.

2. The procedures used for collection, handling, and storage of saltwater and freshwater sediments should be consistent with the protocols recommended by the ASTM (E 1391-90). For example:
   - (a) Sediments that have been frozen must not be used for biological tests (except for Microtox tests).
   - (b) Sediments should not be stored for greater than two weeks prior to use in toxicity tests.

3. The concentrations of one or more analyte(s) must vary by at least a factor of ten at different sampling sites.

4. The chemical analytical procedures **must** have been appropriate for determining the total concentrations of the analytes in bulk sediment samples. For example, strong acid digestions are required to determine total concentrations of metals.

5. Test sediments **should** be characterized so that any factors which may affect toxicity can be included in the evaluation process. In the overlying water, variables such as temperature, pH, dissolved oxygen, residual chlorine, suspended solids, and water hardness (and/or alkalinity) or salinity **should** be measured. In the sediment, variables such as moisture content, organic carbon, acid volatile sulfides, and particle size distribution **should** be reported. However, studies that do not report these variables may still be included in the database.

6. The procedures used to assess the toxicity of sediment-sorbed contaminants in whole sediments (and other appropriate media) should be consistent with the protocols recommended by the ASTM (E 1367-90, E 1383-90, etc.). Other tests which employ more novel protocols should be evaluated on a case by case basis (e.g., Green Book tests are acceptable).

7. Responses and survival of controls must be reported and within acceptable limits.

8. Appropriate statistical procedures should be used and reported in detail.