

# **Approach to the Assessment of Sediment Quality in Florida Coastal Waters**

## **Volume 1 - Development and Evaluation of Sediment Quality Assessment Guidelines**

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## Executive Summary

Florida sediment chemical measurements indicate that contaminants are present in elevated levels in a number of coastal areas. However, this information alone is not sufficient to indicate potential biological harm associated with chemical levels. A cost-effective approach for screening chemical levels is needed to estimate potential biological effects. This report was prepared to provide the Florida Department of Environmental Protection with guidance on the development of effects-based sediment quality assessment guidelines (SQAGs) for Florida coastal waters. A variety of approaches for deriving numerical SQAGs were reviewed and evaluated in light of Florida's unique requirements for SQAGs. The results of this evaluation indicated that an approach recommended by Long and Morgan (1990; National Oceanic and Atmospheric Administration) would provide a practical near-term basis for deriving SQAGs. Using this approach, preliminary SQAGs for 34 priority substances in Florida coastal waters were derived and evaluated. These SQAGs are intended to assist sediment quality assessment applications, such as identifying priority areas for non-point source management actions, designing wetland restoration projects, and monitoring trends in environmental contamination. They are not intended to be used as sediment quality criteria. A preliminary evaluation of the SQAGs is included in the report to provide practitioners with further guidance on using these management tools. While this evaluation indicates that the preliminary guidelines are broadly applicable in the southeast, care should be exercised in applying the SQAGs elsewhere in North America. The report also includes recommendations on improving sediment quality assessments. These revised SQAGs replace guidelines initially recommended to the Florida Department of Environmental Protection.

In Florida, natural resource conservation and protection is a high priority environmental management goal. Realizing this goal requires protecting and restoring 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 understanding and public recognition of the important role that sediments play in maintaining coastal ecosystems. In addition to providing important habitats for aquatic organisms, sediments play a critical role in determining the fate and effects of environmental contaminants. Hence, sediment quality issues and concerns are becoming more prominent in managing natural resources.

Recent Florida sediment chemical measurements indicate that various contaminants are present at elevated levels in a number of coastal areas. While these chemical data provide essential information on the nature and areal extent of contamination, they do not provide a direct measurement of adverse biological effects or indicate the potential for such effects. Biological effects-based sediment quality assessment guidelines (SQAGs) are also 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 SQAGs, the major approaches used in other jurisdictions to derive sediment quality guidelines were reviewed and evaluated in the context of Florida's requirements for sediment quality assessment values. The results of this analysis indicated that the National Oceanic and Atmospheric Administration (NOAA)

National Status and Trends Approach (NSTPA; Long and Morgan 1990) would provide a basis for addressing Florida's immediate need for reliable and cost-effective SQAGs. A strategy was recommended to derive preliminary numerical SQAGs which support the near-term requirements for assessing sediment quality. This strategy allows for immediate assessment of sediment quality. A critical evaluation of this procedure suggested that, while this approach has limitations that could influence the applicability of the guidelines, it was a sound near-term strategy for deriving scientifically defensible preliminary assessment 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 SQAGs for 34 priority contaminants in Florida coastal waters. However, insufficient data were available to derive guidelines for another 14 substances that are known or are suspected to contaminate Florida coastal sediments. The numerical SQAGs define three ranges of concentrations for each of the 34 contaminants: a probable effects range; a possible effects range; and, a minimal effects range. These ranges of contaminant concentrations were considered to be more effective assessment tools than single numerical guideline values.

An evaluation was conducted to determine the reliability, comparability, and predictability of the SQAGs. While the results of this evaluation suggest that the guidelines will be broadly applicable, local environmental conditions may influence their applicability. In addition, the information upon which the SQAGs are based is dominated by data collected in the southeast. For these reasons, the guidelines should be applied with care and in concert with other assessment tools to conduct comprehensive sediment quality assessments.

The preliminary guidelines were established to provide yardsticks for evaluating sediment quality in Florida. These guidelines are intended to be used as one tool in a toolbox of companion interpretive approaches (such as the Department of Environmental Protection's metals interpretive tool and various bioassessment techniques) for screening sediment chemistry data and establishing priorities with respect to sediment quality management. They should not be used in lieu of water quality criteria, nor should they be used as sediment quality criteria. The SQAGs do not supersede formal regulatory assessment protocols, such as those implemented under the federal Marine Protection, Research, and Sanctuaries Act.

## **Recommendations**

In addition to the guidelines themselves, there are several recommendations for follow-up actions contained in this report. The major recommendations are as follows:

- ▶ Recent results of Florida sediment toxicity studies should be used to increase the number of substances covered by the SQAGs and strengthen their applicability and defensibility.
- ▶ The preliminary SQAGs developed in the present study and Florida Department of Environmental Protection's previously developed guidelines for interpreting sediment metal concentrations provide a basis for evaluating sediment quality conditions in Florida coastal ecosystems. No such tools exist for use in freshwater ecosystems, and effects-based SQAGs should be developed to evaluate the biological significance of contaminated sediments in freshwater systems. In addition, the Florida Department of Environmental Protection's approach for assessing sediment metal contamination in coastal waterbodies should be validated and modified as necessary for use in freshwater ecosystems.
- ▶ In the southeast, various independent and loosely-related initiatives are directed at evaluating and managing contaminated sediments. Development of a regional intergovernmental strategy for contaminated sediment assessment and management would improve the effectiveness of these programs and encourage greater local support in preventing sediment contamination and restoring coastal resources. Therefore, a cooperative regional strategy should be developed by the U.S. Environmental Protection Agency, U.S. Army Corps of Engineers, U.S. Geological Survey, Florida Department of Environmental Protection, and other affected agencies to identify priority sediment management, monitoring, and regulatory objectives, and the cooperative efforts required to achieve them.

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**NOTE:** This report has been reviewed by the Florida Department of Environmental Protection and approved for publication. Such approval does not signify that the methods and guidelines necessarily represent the policies of Florida Department of Environmental Protection, nor does mention of trade names or commercial products constitute endorsement or recommendation for their use.

# Chapter 1

## Introduction

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Public awareness of the quality of coastal waters has been raised in recent years as a result of the information that has been disseminated on 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 by the National Oceanic and Atmospheric Administration (NOAA) National Status and Trends Program (NSTP), indicates that levels of contaminants, in general, have begun to decrease in coastal waters (O'Connor 1990). Nonetheless, many coastal waters continue to receive discharges of contaminants from non-point and point sources, resulting in elevated and biologically significant concentrations of many contaminants in urbanized estuaries throughout the United States.

Traditionally, the management of aquatic resources in coastal waters 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). In addition to providing a habitat for many organisms, sediments are important because many toxic substances found only in trace amounts in water may accumulate to elevated levels in them. As such, sediments serve both as reservoirs and as potential sources of contaminants to the water column. As well as their potential to degrade surface water quality, sediment-associated 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 aquatic ecosystems.

Over the past 10 years, Florida Department of Environmental Protection (FDEP) and others have collected a substantial quantity of information on the chemical composition of Florida coastal 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) also are 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 are integral to this framework, providing the basis for assessing *potential* effects of sediment-associated contaminants. A variety of approaches for deriving 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 approach has deficiencies which limit its direct application in Florida. For this reason, an integrated strategy for deriving numerical SQAGs is recommended for the state of Florida. The recommended strategy is designed to provide relevant assessment tools in the near-term and allows for refining these guidelines as additional data become available.

Using the recommended approach, numerical SQAGs were developed for assessing sediment quality in Florida coastal waters. These guidelines were derived using information from numerous investigations of coastal sediment quality conducted in North America and are based on a weight of evidence that links contaminant concentrations with adverse biological effects. In this respect, the guidelines are a cost-effective response to a practical need for assessment tools. However, these guidelines are should be revised or refined depending on the results of field validation and other related studies conducted in Florida and elsewhere in North America. These guidelines should be used in conjunction with other interpretive tools to conduct comprehensive and reliable assessments.

## 1.2 Description of the Recommended Approach for Deriving Numerical Sediment Quality Assessment Guidelines

The recommended approach for deriving numerical SQAGs is described in Chapter 4. This approach 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 provides a *weight of evidence* from numerous biological effects-based approaches for determining associations between chemical quality and biological effects;
- It provides assessment tools or guidelines that define *ranges of contaminant concentrations* that can be used to evaluate sediment quality data. Specifically, these guidelines define ranges of concentrations that are usually or always, frequently, and rarely or

never 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 provides 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 has long-term applicability in Florida since it 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 5.4.

Sediment quality assessment guidelines derived using the recommended approach should be refined as new information becomes available. Several limitations and precautions in using this approach are identified as follows:

- ▶ The approach is designed to determine the potential for sediment-associated contaminants to induce biological effects. Direct cause and effect relationships should not be inferred when comparing chemical data to the recommended guidelines;
- ▶ The SQAGs are applicable to marine and estuarine waters only; they are not applicable to freshwater systems;
- ▶ The SQAGs 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 SQAGs consist primarily of the results of acute toxicity studies. This reflects the general lack of data 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 FDEP 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 North American coastal locations. 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 5.4.

### **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. The 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 levels of sediment-associated 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 EPA. 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 affected agencies and other parties in preventing and remediating sediment contamination.

These guidelines were established to provide a consistent basis for evaluating sediment quality in Florida. However, these guidelines have certain limitations on their application. The SQAGs:

- Should not be used in lieu of water quality criteria. However, these guidelines may be used to evaluate the effectiveness of regulatory programs and to 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 material and should not be used to replace formal assessment protocols established for the disposal of dredged material (EPA and ACE 1991); and,
- Should not be used directly as numerical clean-up levels at severely contaminated sites (e.g., Superfund sites).

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 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-associated contaminants. In the long-term, refinement of the guidelines should be pursued to ensure their broader applicability and utility within the state.

## **Chapter 2**

### **Florida's Coast: A National Treasure**

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#### **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's 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 provide essential transportation links, support a variety of water-dependent facilities, and offer an array of 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 due to a combination of climatological and physiographic features which occur no where else. 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 (Comp and Seaman 1985). This region has few direct sources of freshwater inflow to the ocean and is marked by an extensive system of high salinity lagoons. The major estuaries on the Atlantic coast are the St. Johns River and Indian River estuaries and Biscayne Bay. Collectively, these estuaries cover a water surface area of almost 2,000 square kilometers.

Excluding the Florida Keys, the Gulf of Mexico coast of Florida extends some 1,350 km from Florida Bay to Perdido Bay. In general, estuaries along the west coast are located behind low energy barrier islands or at the mouths of rivers that discharge into salt marshes or mangrove-fringed bays (Comp and Seaman 1985). NOAA (1990) identified a total of fourteen major estuaries along Florida's Gulf coast, covering an estuarine water surface area of more than 5,000 square kilometers.

Coastal and estuarine sediments of Florida span a significant geochemical range, from silica- and aluminum-rich sediments of northeastern and northwestern Florida to carbonate-rich sediments of south Florida and the Florida Keys. The geochemistry of estuarine sediments in northern and central Florida reflects the siliceous, aluminous composition of presently eroding uplands of the southeastern United States. In contrast, the calcium carbonate-rich coastal sediments of Florida Bay and the Florida Keys are formed as the remains of diverse marine flora and fauna slowly accumulate in these estuarine waters. The remains of marine plants and animals create virtually all of the sediment volume in this part of Florida, with a minor terrestrial sediment input from riverine and salt marsh systems such as the Everglades.

## 2.2 Biological Features of Florida's Coast

The brackish water habitats that ring the Florida mainland are vital to the state's natural plant and animal communities. Florida's coastal environments are comprised of myriad salt marshes, mangrove forests, and open water communities that support a diverse array of aquatic and terrestrial organisms (Comp and Seaman 1985).

The salt marshes of Florida, which cover approximately 170,000 ha of land, are coastal ecosystems with communities of non-woody, salt-tolerant plants occupying intertidal zones that are occasionally or periodically inundated with salt water (Montague and Wiegert 1990). These areas provide such beneficial features as sediment stabilization, storm protection, aesthetic values, and wildlife habitat. The rate of primary productivity in salt marshes is among the highest of the world's ecosystems. This productivity forms the basis of aquatic and terrestrial food webs that include many unique and economically important plant and animal species.

Estuaries are dominant features along the Florida coastline. Estuaries are among the most productive natural systems and their role in sustaining the health and abundance of marine fishes, shellfish, and wildlife has long been recognized (NOAA 1990). The importance of estuarine habitats for marine fishes and invertebrates is emphasized by the fact that up to 97.5% of the total commercial fisheries catch in the Gulf of Mexico is comprised of species that are dependant on estuaries during some portion of their life cycle (Comp and Seaman 1985). The Gulf of Mexico coastline supports one of the most productive fisheries in the world, with shrimp, oysters, lobsters, scallops, clams, and menhaden being the most important commercial species. While the Atlantic coast fishery is somewhat less important than Gulf coast fisheries, substantial quantities of shrimp, crab, clams, and menhaden are harvested on an annual basis.

In addition to diverse and abundant fisheries, Florida's coastal areas support a wide variety of plant and animal species. Wetland habitats are utilized by numerous species of wading birds, waterfowl, raptors, and a variety of mammalian species. All of these organisms are dependent, to a greater or lesser extent, on the productivity of Florida's coastal waters. In turn, the aquatic organisms that support the impressive communities of higher organisms are dependent on coastal sediments to provide feeding, spawning, and rearing habitats.

## **2.3 Anthropogenic Influences on Florida's Coast**

Environmental management and pollution control issues affecting Florida's coast differ from those affecting coastal areas in other portions of the United States. While there are many common issues, land uses in Florida differ significantly from those in other states. In the northeastern and northwestern portions of the United States coastal ecosystems are influenced by myriad point sources of pollution, primarily from municipal and industrial sources. For example, Hudson River/Raritan Bay (New York) and Chesapeake Bay (Virginia/Delaware) contain more point sources of pollution than any other estuarine areas in the nation (NOAA 1990). Similarly, areas like Puget Sound (Washington) and San Francisco Bay (California) are highly industrialized, with large quantities of effluents discharged into receiving water systems (NOAA 1990). In addition, these systems are often adjacent to older, highly populated cities, which exacerbates the stresses on coastal waters.

Florida ranks fourth among states in terms of population, with nearly 13 million persons in 1990 (USDC 1990). This population was expected to increase by nearly 4% by the year 1992 (USDC 1990). Most of the population of the state currently resides near the coast and population densities in these areas are predicted to increase by over 30% in the next 20 years (Culliton *et al.* 1990). As indicated by FDEP coastal contaminants surveys, non-point discharges are the major sources of contaminants to coastal waterbodies. If population and land use trends continue, inputs of contaminants to coastal waters due to the deposition of atmospheric pollutants and stormwater runoff from urban and suburban areas are likely to

escalate. Likewise, the capacity of municipal wastewater treatment plants will have to increase to accommodate the needs of the burgeoning population. Coastal waters may be used as receiving water systems for many of these point and non-point source discharges.

Manufacturing has traditionally played a smaller role in the Florida economy than in the economy of other states (Fernald 1981). For this reason, Florida's coastal waters are not severely influenced by industrial effluents. Nonetheless, Farrow (1990) indicated that 615 billion gallons of industrial effluent were released into Florida's coastal waters in 1982. The sources of these effluents included the pesticides, organic chemicals and plastics industries, and a variety of other discharges. Of these, the pulp and paper industry may be of particular importance in certain areas of Florida due to its discharges of toxic and bioaccumulative substances into coastal waters (Farrow 1989).

Florida is second (after North Carolina) among southeastern states with respect to the economic value of its agricultural production (Fernald 1981) and is renowned for its production of citrus fruits. While Florida has an excellent climate for the culture of agricultural products, it is dominated by sandy soils with relatively low fertility (Ewel 1990). As such, maintenance of high rates of productivity necessitates the application of large quantities of fertilizers and pesticides. The combination of high irrigation rates, high soil porosity, and low organic content in the soil enhances the potential for the mobilization of many agricultural chemicals. Overland or subsurface transport of many of these substances could ultimately lead to the contamination of coastal waters.

While municipal, industrial and agricultural sources probably represent the major inputs of contaminants to coastal ecosystems, other sources of contamination in Florida's coastal waters may include leachates from landfill sites, dredge and fill activities, and the operation of ships and pleasure craft. Together, anthropogenic influences in estuarine and nearshore marine environments represent a potential hazard to the health and integrity of coastal ecosystems. Ongoing monitoring of environmental conditions provides a means of assessing the nature and extent of environmental contamination and a basis for managing the valuable resources that currently exist in Florida's coastal ecosystems.

## **2.4 Sediment Quality Issues and Concerns**

Natural resource protection is one of the most important goals in managing Florida's coastal waters. These resources include wetlands, floodplains, estuaries, beaches, dunes, barrier islands, coral reefs, and fish and wildlife. Habitats that support the production of fish and wildlife are of fundamental importance and have been identified as natural resources that require special consideration for protection and enhancement (Comp and Seaman 1985).

Maintenance and enhancement of the diversity and abundance of biological resources in coastal ecosystems in Florida requires an integrated approach to environmental management. Implementing an integrated environmental management system is dependent on the development of a comprehensive understanding of the fate and effects of environmental contaminants. While many contaminants are released into the aqueous component of coastal ecosystems, not all of these substances dissolve easily in water. For instance, hydrophobic substances tend to adhere (or adsorb) to particulate matter that becomes deposited on the bottom. This tendency for many toxic substances to form associations with sediments can result in elevated concentrations of certain contaminants in bed sediments. Elevated levels of sediment-associated contaminants may represent hazards to ecosystem integrity by affecting aquatic organisms directly or by limiting the use of those resources by human consumers.

Sediments contaminated with toxic substances have been found in coastal areas throughout the world. Bolton *et al.* (1985) reported that metals, polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), and DDT were chemicals of major concern at a number of marine and estuarine sites in America. A more recent evaluation, using data collected in the National Status and Trends Program (NSTP), has confirmed that bed sediments at sites within the Hudson-Raritan estuary, Boston Harbor, Long Island Sound, and San Francisco Bay are highly contaminated with an assortment of toxic substances (Long and Morgan 1990).

While concerns over contaminated coastal sediments in the United States have been focused primarily in the northeast and on the west coast, the results of recent studies are beginning to indicate that sites in the southeast have also been affected by anthropogenic activities. For example, Long and Morgan (1990) reported that sites in St. Andrews Bay were highly contaminated with pesticides, metals, PCBs, and PAHs (i.e., levels of contaminants in sediment equalled or exceeded the concentrations that were frequently or always associated with toxic effects). These contaminants have also been detected at levels of concern at sites in the vicinity of Miami, Jacksonville, Daytona Beach, Tampa Bay and Pensacola Bay (MacDonald 1994; Long *et al.* 1991).

Sediment-associated contaminants have been connected with a wide range of impacts on the plants and animals that live within and upon bed sediments. Acute and, in some cases, chronic toxicity of sediment-associated contaminants to algae, invertebrates, fish, and other organisms have been measured in laboratory toxicity tests. Field surveys have identified more subtle effects of environmental contaminants, such as the development of tumors and other abnormalities in bottom feeding fish (Goyette *et al.* 1988; Malins *et al.* 1985). In addition, many of these substances can accumulate in fish and shellfish tissues. At elevated levels, these contaminants represent hazards to sensitive wildlife species that rely on these organisms for food. Furthermore, bioaccumulation may result in impairments to human uses of coastal ecosystems. In many areas of North America, health departments have advised residents to limit their consumption of seafood. For example, the Florida Department of Health and Rehabilitative Service and the Florida Department of Agriculture and Consumer Services (Mercury Technical Committee 1991) have issued a health advisory on consumption of shark

meat based on mercury levels in samples of sharks obtained in retail markets. In addition, observations of elevated mercury levels in Florida freshwater fish have resulted in the issuance of health advisories that recommended avoidance of fish consumption in specified state waterbodies.

## **2.5 Sediment Quality: An Indicator of Ecosystem Health**

While evaluations of sediment quality are often used to address site-specific management needs, sediment quality is also used as a sensitive indicator of overall environmental quality. Sediments influence the environmental fate of many toxic and bioaccumulative substances in aquatic ecosystems. As such, sediments integrate contaminant inputs over time and may also represent long-term sources of contamination. In addition to the physical and chemical relationships between sediments and contaminants, sediments are of fundamental importance to benthic communities in terms of providing suitable habitats for essential biological processes (such as, spawning, incubation, rearing, etc.). Therefore, sediments provide an essential link between chemical and biological processes. By developing an understanding of this link, environmental scientists are developing assessment tools and conducting monitoring programs that enable them to make rapid and accurate evaluations of the state of the environment and the health of aquatic ecosystems.

## **Chapter 3**

### **An Evaluation of Existing Approaches to Developing Numerical Sediment Quality Guidelines**

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#### **3.0      Introduction**

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

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

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

### **3.1 Sediment Background Approach (SBA)**

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

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

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

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

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

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

### **3.2 Spiked-Sediment Bioassay Approach (SSBA)**

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

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

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

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

### 3.3 Equilibrium Partitioning Approach (EqPA)

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

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

$$\text{SQG} = \text{K}_p \text{ FCV}$$

where:

|              |   |  |
|--------------|---|--|
| SQG          | = | Sediment quality guideline (in $\mu\text{g}/\text{kg}$ );          |
| $\text{K}_p$ | = | Partition coefficient for the chemical (in $\text{L}/\text{kg}$ ); |
| FCV          | = | and,<br>Final chronic value (in $\mu\text{g}/\text{L}$ ).          |

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

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

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

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

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

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

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

### **3.4 Tissue Residue Approach (TRA)**

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

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

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

This approach has been used on several occasions to develop water quality guidelines for the protection of human health (most notably for DDT, Hg, and PCBs). In addition, sediment contamination limits for 2,3,7,8 tetrachlorodibenzo-*p*-dioxin (T<sub>4</sub>CDD) have been established for Lake Ontario on the basis of fish tissue residues (Endicott *et al.* 1989; Cook *et al.* 1989).

Using a risk assessment approach, EPA (1993) has also derived SQGs for mammalian and avian wildlife species for this substance. The applicability of this approach for deriving SQGs is supported by data which demonstrate that declines in DDT residues in fish and birds (since its use was banned) are strongly correlated with declining concentrations of this substance in surficial sediments in the Great Lakes and Southern California Bight. As such, this approach is a logical companion for the effects-based approaches to deriving SQGs.

### **3.5 Screening Level Concentration Approach (SLCA)**

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

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

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

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

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

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

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

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

### **3.6 Sediment Quality Triad Approach (SQTA)**

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

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

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

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

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

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

### 3.7 Apparent Effects Threshold Approach (AETA)

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

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

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

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

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

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

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

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

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

### **3.8 Weight of Evidence Approach (WEA)**

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

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

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

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

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

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

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

### **3.9 Summary**

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

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

| Approach | Strengths  | Limitations  |
|----------|--|--|
| SBA      | <ul style="list-style-type: none"> <li>▪ Sufficient data are generally available.</li> </ul>   | <ul style="list-style-type: none"> <li>▪ Not based on biological effects.</li> </ul>   |
| SSBA     | <ul style="list-style-type: none"> <li>▪ Based on biological effects.</li> <li>▪ Suitable for all classes of chemicals and most types of sediments.</li> <li>▪ Supports cause and effect evaluations.</li> </ul>   | <ul style="list-style-type: none"> <li>▪ Sufficient data are not generally available.</li> <li>▪ Implementation costs are high.</li> <li>▪ Spiking procedures are not yet standardized.</li> </ul>   |
| EqPA     | <ul style="list-style-type: none"> <li>▪ Based on biological effects.</li> <li>▪ Suitable for all classes of chemicals and most types of sediments.</li> <li>▪ Bioavailability is considered.</li> <li>▪ EPA will support research to validate this approach.</li> <li>▪ Supports cause and effect evaluations.</li> </ul> | <ul style="list-style-type: none"> <li>▪ Few sediment quality criteria are currently available.</li> <li>▪ Water quality criteria are not available for some substances.</li> <li>▪ In situ sediments are rarely at equilibrium.</li> </ul>  |
| TRA      | <ul style="list-style-type: none"> <li>▪ Simple to apply.</li> <li>▪ Bioaccumulation is considered.</li> <li>▪ A protocol for the derivation of TRGs is available.</li> </ul>  | <ul style="list-style-type: none"> <li>▪ Tissue residue guidelines for wildlife are not yet available.</li> <li>▪ In situ sediments are rarely at equilibrium.</li> </ul>  |
| SLCA     | <ul style="list-style-type: none"> <li>▪ Based on biological effects.</li> <li>▪ Sufficient data are generally available.</li> <li>▪ Suitable for all classes of chemicals and most types of sediments.</li> </ul>   | <ul style="list-style-type: none"> <li>▪ Not possible to establish cause and effect relationships.</li> <li>▪ Large database is required.</li> <li>▪ End point used is insensitive.</li> <li>▪ Bioavailability is not considered.</li> </ul> |

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

| Approach | Strengths  | Limitations   |
|----------|--|---|
| SQTA     | <ul style="list-style-type: none"> <li>▪ Based on biological effects.</li> <li>▪ Chemistry, bioassay and in situ biological effects are integrated.</li> <li>▪ Provides a weight of evidence.</li> </ul>   | <ul style="list-style-type: none"> <li>▪ Difficult to derive numerical SQAGs.</li> <li>▪ Labour intensive and expensive.</li> <li>▪ Statistical criteria for evaluating TRIAD have not been established.</li> <li>▪ Extensive site-specific database is required.</li> <li>▪ Not possible to establish cause and effect relationships.</li> <li>▪ Bioavailability is not considered.</li> </ul> |
| AETA     | <ul style="list-style-type: none"> <li>▪ Based on biological effects.</li> <li>▪ All types of biological data are considered.</li> <li>▪ Suitable for all classes of chemicals and most types of sediments.</li> </ul>   | <ul style="list-style-type: none"> <li>▪ Extensive site-specific database is required.</li> <li>▪ Not possible to establish cause and effect relationships.</li> <li>▪ Risk of under- or over- protection of resource.</li> <li>▪ Not applicable to the derivation of broadly applicable SQAGs.</li> <li>▪ Bioavailability is not considered.</li> </ul>  |
| WEA      | <ul style="list-style-type: none"> <li>▪ Based on biological effects.</li> <li>▪ All types of biological data are considered.</li> <li>▪ Suitable for all classes of chemicals and most types of sediments.</li> <li>▪ Provides a weight of evidence.</li> <li>▪ Provides data summaries for evaluating sediment quality.</li> <li>▪ May be implemented with existing data.</li> </ul> | <ul style="list-style-type: none"> <li>▪ Large database is required.</li> <li>▪ Not possible to establish cause and effect relationships.</li> <li>▪ Amalgamation of data from multiple sources could result in unknown biases in the database.</li> <li>▪ Bioavailability is not considered.</li> </ul>  |

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## **Chapter 4**

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

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#### **4.0 Introduction**

This chapter completes the evaluation of approaches for developing SQAGs conducted in Chapter 3 and provides an overview of, and rationale for, the recommended strategy for deriving and validating numerical SQAGs for Florida coastal waters. As indicated in Chapter 3, no effects-based SQAGs exist which apply directly to conditions in Florida. While effects-based SQAGs have been developed for specific parts of the country (e.g., 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 partitioning approach; EqPA), although a more recent review is less critical (Sediment Quality Subcommittee 1992). These evaluations by the SAB suggest that the SQAGs that are under development in other jurisdictions are not likely to address Florida's immediate requirements for sediment assessment tools.

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

A total of eight approaches for deriving numerical SQAGs 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. In this chapter, criteria are provided to evaluate candidate approaches and select a relevant strategy for deriving guidelines (Table 2). The primary considerations in the selection of the recommended strategy were related to practicality, cost-effectiveness, scientific defensibility, and broad applicability to the assessment of sediment quality. Each of these factors are discussed below.

**Table 2. Evaluation of approaches for deriving sediment quality assessment guidelines.**

| Evaluation Criteria   | SBA | SSTA | EqPA | TRA | SLCA | SQTA | AETA | WEA  |
|---|-----|------|------|-----|------|------|------|------|
| <b>Practicality</b>   |     |      |      |     |      |      |      |      |
| Supports development of numerical SQA?  | Y   | Y    | Y    | Y   | Y    | Y    | Y    | Y    |
| Feasible to implement in the near term?   | Y   | N    | Y/N  | Y/N | Y/N  | N    | N    | Y    |
| <b>Cost Effectiveness</b>   |     |      |      |     |      |      |      |      |
| Expensive to implement?   | N   | Y    | N    | Y   | Y    | Y    | Y    | N    |
| Requires generation of new data?  | N   | Y    | N    | Y   | Y    | Y    | Y    | N    |
| <b>Scientific Defensibility</b>   |     |      |      |     |      |      |      |      |
| 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   | Y    | Y    | Y    | Y    |
| Considers data from South East?   | Y   | N    | N    | N   | N    | N    | N    | Y    |
| Provides weight of evidence?  | N   | N    | N    | N   | N    | Y    | Y    | Y    |
| Support definition of ranges of concentrations rather than absolute assessment values | N   | N    | Y/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   | Y/N  | Y/N  | N   | N    | Y    | Y    | N    |
| Applicable to all classes of chemicals?   | Y   | N    | Y/N  | Y   | Y    | Y    | Y    | Y    |
| <b>Applicability</b>  |     |      |      |     |      |      |      |      |
| 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

Practicality is one of the central considerations with respect to the development of SQAGs. Numerical SQAGs 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 in the near term.

In Florida, limited agency resources make collection of a significant quantity of additional data improbable. Therefore, the approach that is recommended must support the development of numerical SQAGs with data that are currently available. In addition, it must be amenable to re-evaluation of the SQAGs as new data become available.

For SQAGs 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-associated contaminants, to establish cause and effect relationships, and to apply to priority classes of chemicals and mixtures of contaminants that are expected to occur in Florida. They must be compatible with other interpretive tools, such as the metals interpretive tool that has already been developed by the Florida Department of Environmental Protection (FDEP). 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 uncertainty associated with 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 (as was recommended by the SAB; Sediment Quality Subcommittee 1992). 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 needs of the agencies that are charged with managing environmental quality. For example, SQAGs should be relevant to designing, implementing, and evaluating environmental monitoring programs by helping identify contaminants that are likely to be associated with adverse biological effects. This would help determine 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 the need for further biological and chemical testing to support regulatory decisions.

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

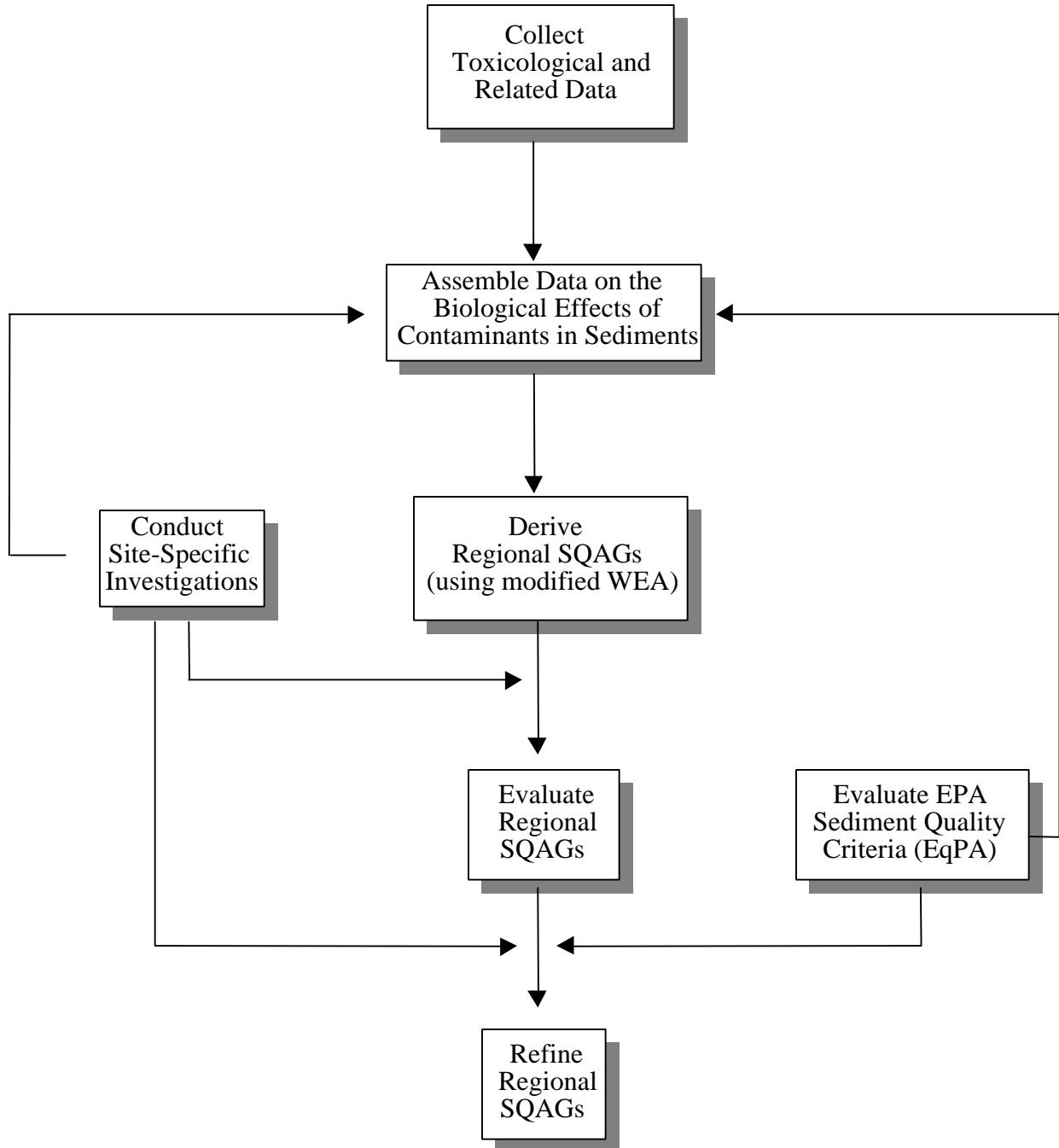
Ideally, SQAGs should be developed from detailed dose-response data which describe the acute and chronic toxicity of individual contaminants and chemical mixtures to sensitive life stages of resident species of aquatic organisms. These data should be generated in controlled laboratory studies, in which 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 be validated in field trials to ensure that any guidelines derived from these data are applicable to a broad range of locations.

Unfortunately, insufficient data are available to support deriving 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-associated contaminants on estuarine and marine organisms (Long and Morgan 1990). However, in spite of this limitation, other types of data are routinely collected which contribute to our understanding of the toxic effects of these contaminants. Specifically, whole sediment toxicity tests have been conducted to assess the biological significance of concentrations of contaminants in sediments collected from numerous 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 been collected. Studies which report matching sediment chemistry and biological effects data are highly relevant to the SQAGs derivation process.

In recommending a suitable strategy for deriving SQAGs, it is important to recognize the limitations of the existing information for evaluating the potential biological effects of sediment-associated contaminants. In addition, the strategy must address 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 influencing the bioavailability of sediment-associated contaminants).

Evaluation of each of the approaches for deriving SQAGs in the context of the requirements for Florida (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 data become available (Figure 1).

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



The weight of evidence approach (WEA; 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 SQAGs. However, several modifications (described in Chapter 5) to this approach are recommended to increase the applicability of the WEA to Florida. These modifications increase the quantity and suitability of data used to evaluate the biological significance of sediment-associated contaminants (i.e., to incorporate data from Florida, other southeastern areas, and 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 WEA for deriving SQAGs is provided in Chapter 5.

## **Chapter 5**

# **Derivation of Numerical Sediment Quality Assessment Guidelines for Florida Coastal Waters Using the Weight of Evidence Approach**

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### **5.0 Introduction**

The National Oceanic and Atmospheric Administration (NOAA) National Status and Trends Program (NSTP) weight of evidence approach (WEA; Long and Morgan 1990) is chosen as a basis for developing 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 North America to establish relationships between concentrations of sediment-associated contaminants and their potential for adverse biological effects. A modified version of the WEA is recommended for deriving numerical SQAGs.

### **5.1 Modification of the Original Weight of Evidence Approach for Deriving Florida Sediment Quality Assessment Guidelines**

A modified version of the WEA is recommended to derive preliminary SQAGs due to its practicality for developing guidelines, 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 original WEA, however, the modifications increase the relevance of the resultant guidelines to Florida coastal sediments. Specifically, the modifications to this approach increase the level of internal consistency in the database (by establishing additional screening criteria), verify and expand the information contained in the original NSTP database, and use all of the information in the database to derive SQAGs (in contrast, only data which had concordance between sediment chemistry and biological effects were used to derive the original NSTP guidelines). In addition, user access to the information from individual studies has been improved by providing expanded data tables (see Volumes 3 and 4).

### **5.1.1 Procedures and Criteria for Screening Candidate Data Sets**

The modified WEA integrates a diverse assortment of data to derive numerical SQAGs. 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, SQCs 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 was used to derive SQAGs 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 experimental designs, 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. 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 rendered these data consistent with the other information included in the database (which had an average TOC of 1.2 +/- 1.8%).

### **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-associated contaminants, with respect to deriving SQAGs 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-associated contaminants from sites located in the southeastern United States. Therefore, collection of acceptable data from Florida and other areas in the southeast was a priority in the present study.

To address the need for additional information on the biological effects of sediment-associated 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 involved identifying and retrieving candidate data sets from sites 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. Contacts in the southeast included representatives from United States Environmental Protection Agency, United States Army Corps of Engineers, Florida Department of Environmental Protection, National Biological Survey, United States 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. Each of the 350 members of the ASTM Subcommittee E47.03 on Sediment Toxicology was contacted to obtain relevant data. Furthermore, a total of 12 bibliographic databases were searched electronically to obtain information that was published in the primary scientific literature.

Over the course of this study, more than 700 publications were retrieved and evaluated to determine their suitability for use in the derivation of SQAGs. More than 120 of these publications were used to verify and expand the original NSTP database. Roughly 35% of the publications used in the present study were from studies conducted in the southeastern and Gulf of Mexico regions 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 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-associated 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 sediment toxicity database provides a basis for deriving SQAGs for priority substances in Florida.

## 5.2 Derivation of Numerical Sediment Quality Assessment Guidelines

The expanded NSTP database consists of information from three types of studies, including equilibrium partitioning (EqP) modelling, laboratory spiked-sediment bioassays, and field investigations of sediment toxicity and benthic community composition. Equilibrium partitioning concentrations, if expressed in units of organic carbon, were converted to units

of dry weight assuming a TOC concentration of 1%. Data from spiked-sediment bioassays were incorporated into the database directly. Field-collected data were treated with a variety of methods. Apparent effects thresholds (AET) and national screening level concentrations (SLC), both of which were based on evaluations of large, merged data sets, were entered directly into the NSTP database. Raw data from other individual surveys were evaluated using co-occurrence analyses (COA), using one of two procedures (Long 1992). If the authors of the reports identified samples that were statistically significantly different from the other groups of samples or from controls, then the mean chemical concentrations in the statistical groups were compared. If no such comparisons were reported, the frequency distributions of the biological data and the mean concentrations in subjectively determined groups of samples were compared (e.g., relatively highly toxic versus least toxic). Data entries were prepared for each endpoint measured in the study (e.g., survival, growth, reproduction, etc.), so that multiple entries for a single geographic area are common in the database.

The expanded NSTP database is a comprehensive source of information on the potential effects of sediment-associated contaminants. Each record in the database contains detailed information on the location of the study, species affected, endpoint measured, particle size distribution, factors that could affect bioavailability of the contaminants (such as TOC and acid volatile sulfides; AVS), and concentrations of the contaminants. Importantly, each entry in the database was assigned an 'effects/no effects' descriptor, based on the degree of concordance between the concentration of the chemical and the endpoint measured in the investigation. Those entries in which the chemical concentrations were considered to be associated with the biological effect measured were designated with an asterisk or "hit" (\*; see Volumes 3 and 4). The descriptors, 'no gradient (NG), small gradient (SG), no concordance (NC), or no effect (NE)', were assigned when either the chemical concentrations were not strongly associated with the biological effect measured or no adverse effects were observed (see below). The data on each substance were then sorted, in ascending order of concentration, 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 'effects data set' (EDS) was comprised primarily of information from 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 EDS 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 biological effects were observed was a factor of two or more greater than the mean concentration at the sites at which no biological effects were observed (this criterion was adopted directly from 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 EDS.

A separate data set was also established, the 'no effects data set' (NEDS), to include the balance of the assembled data. Data on the concentrations of specific substances that were not associated with adverse effects provides important information for defining the relationships between contaminant challenges and biotic responses. For this reason, it was considered appropriate to include the no effects data set in the guidelines derivation process. Several types of information were included in this data set. In general, these entries consisted of data from bioassays in which exposure of aquatic organisms to test sediments did not result in significant biological effects (i.e., no effect). In addition, the descriptors, 'no gradient, small gradient, or no concordance,' were assigned when no differences in the concentration of a particular chemical were reported between stations, the mean chemical concentrations between groups of samples differed by less than a factor of two, or there was no concordance between the severity of the effect and the chemical concentration, respectively. 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.* 1994) but were not included to derive the SQAGs.

Both the effects and the no effects data sets were used to derive numerical sediment quality assessment guidelines for Florida coastal waters, using the steps presented in Figure 2. The arithmetic procedures used in the guidelines derivation process were designed to define three distinct ranges of contaminant concentrations; a minimal effects range, a possible effects range, and a probable effects range. A conceptual representation of the three ranges of contaminant concentrations defined by the guidelines is provided in Figure 3. This figure illustrates the concept that the probability of observing adverse biological effects increases with increasing contaminant concentration.

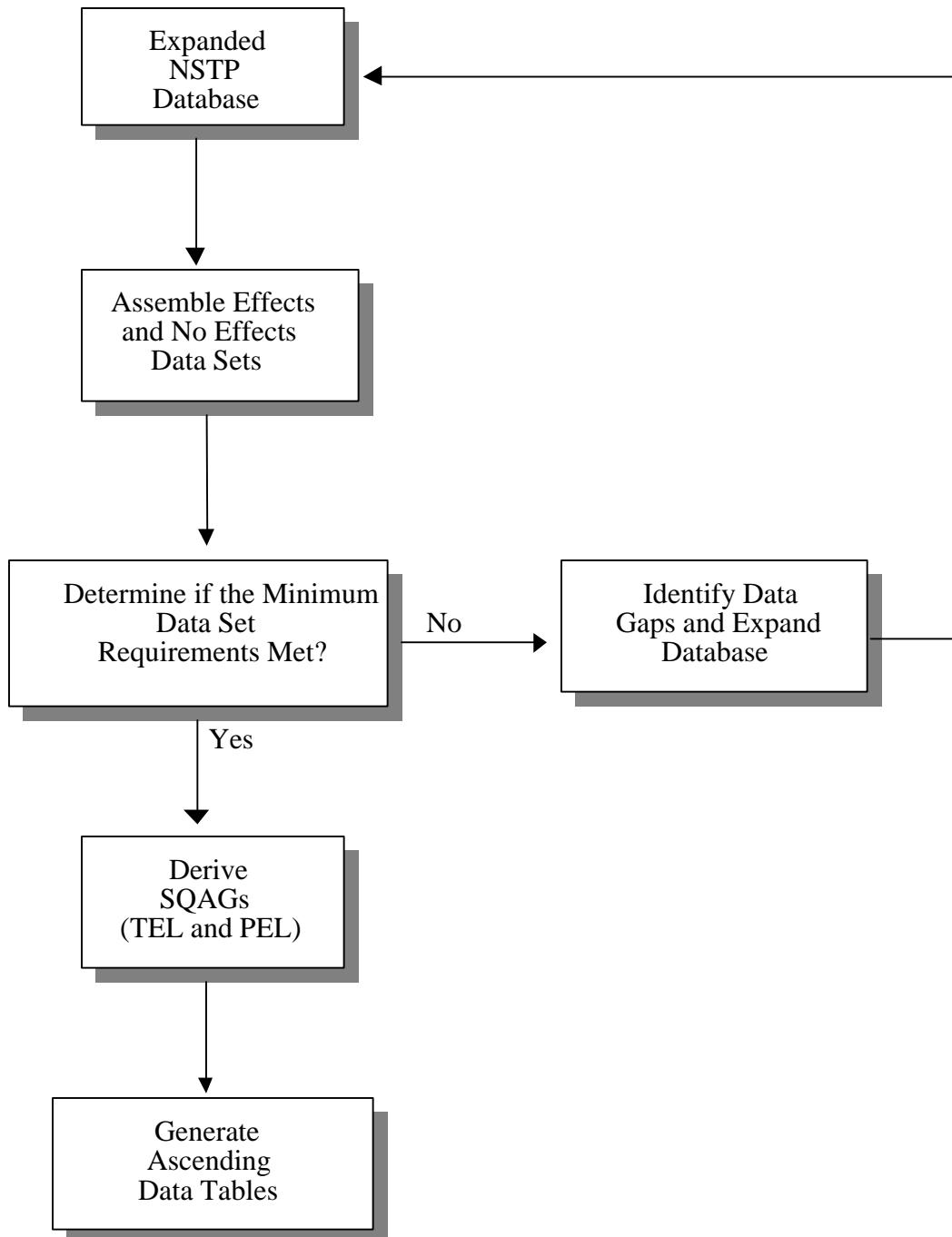
The range of sediment contaminant concentrations that are not likely to be associated with adverse biological effects on aquatic organisms (i.e., the minimal effects range) was defined using a two step process. First, a threshold effects level (TEL) was calculated. The TEL represents the upper limit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

$$\text{TEL} = \sqrt{\text{EDS-L} \text{ NEDS-M}}$$

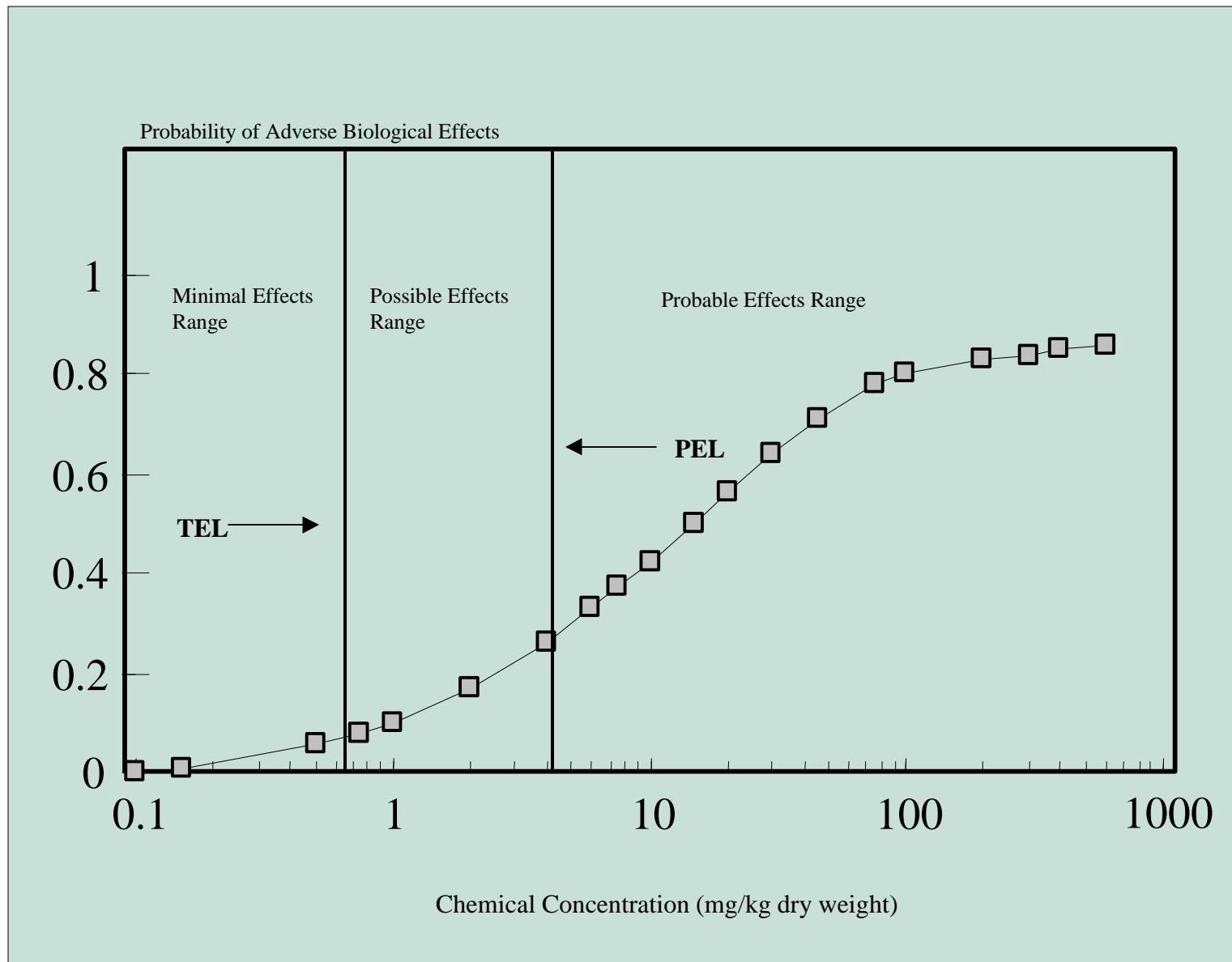
where:

|         |   |   |
|---------|---|---|
| TEL     | = | Threshold effect level;                                     |
| EDS-L = |   | 15th percentile concentration in the effects data set; and, |
| NEDS-M  | = | 50th percentile concentration in the no effects data set.   |

**Figure 2. An overview of the modified NSTPA for deriving numerical sediment quality assessment guidelines in Florida.**



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



The geometric mean, rather than the arithmetic mean, of EDS-L and NEDS-M is calculated because these data are not, necessarily, normally distributed (Sokal and Rohlf 1981; see Appendix 2 for an example of this calculation).

A probable effects level (PEL) was 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 EDS. Within the probable effects range, concentrations of sediment-associated contaminants are considered to represent significant and immediate hazards to aquatic organisms. The PEL was calculated as follows:

$$\text{PEL} = \sqrt{\text{EDS-M} \text{ NEDS-H}}$$

where:

|        |   |   |
|--------|---|---|
| PEL    | = | $\sqrt{\text{EDS-M} \text{ NEDS-H}}$                        |
| PEL    | = | Probable effect level;                                      |
| EDS-M  | = | 50th percentile concentration in the effects data set; and, |
| NEDS-H | = | 85th percentile concentration in the no 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 TEL (lower limit) and the PEL (upper limit). Within this range of concentrations, adverse biological effects are possible; however, it is difficult to predict the occurrence, nature, and/or severity of these effects. 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-associated contaminants represent significant hazards to aquatic organisms. Such investigations would focus first on the determination of the probable origin of the contaminant (i.e., through the use of the metals interpretive tool; Schropp *et al.* 1989), and then on the toxicity of *in situ* sediments (i.e., using bioassessment techniques), as required. See Volume 2 for further guidance on applying the SQAGs. *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 web.*

### 5.3 Rationale for Establishing the Procedure for Deriving the Guidelines

Several arithmetic and statistical procedures were considered for deriving numerical guidelines from the expanded NSTP database. One of the statistical procedures that was considered involved the determination of guideline values based on the percent incidence of

effects associated with increasing contaminant concentrations, which could be determined directly from the database. For example, Long *et al.* (In press) suggested that the ER-M for a specific substance would be considered reliable if there is a relatively high incidence of effects above that concentration (i.e., > 75% hits). Therefore, the percent incidence of effects was considered for deriving the SQAGs, as well. While this option was initially attractive, an evaluation of the relative distribution of the effects and no effects data entries indicated that its implementation would be challenging. Specifically, calculations using the data available on several substances (cadmium and chromium) revealed that, for example, a 75% incidence of adverse effects occurs within concentration ranges that could be delineated by several different guideline values. For this reason, it was decided that an arithmetic procedure that could be applied more consistently to each chemical should be established. While numerous arithmetic procedures are possible, the methods utilized by Long and Morgan (1990) appeared to be the most promising.

Long and Morgan (1990) used the 10th (ER-L) and 50th (ER-M) percentile values in the effects data set 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. Klapow and Lewis (1979) reasoned that the use of percentiles of aquatic toxicity data minimized the influence of single (potentially outlier) data points on the resultant assessment values. For this reason, a percentile-based procedure was selected to support the derivation of SQAGs for Florida coastal waters.

Environmental quality guidelines are, typically, derived from the LC<sub>50</sub>s, EC<sub>50</sub>s, or lowest observed effect levels that are determined from acute and chronic bioassays (Stephan *et al.* 1985; CCME 1991). In addition to this type of information, the expanded NSTP database also includes data from studies in which exposure of organisms to sediments did not result in significant biological effects. The CCREM (1987) recognized the value of using additional information for deriving water quality guidelines (WQGs) for irrigation and livestock watering. In this case, WQGs were derived by applying a safety factor to the geometric mean of the lowest and no observed effects levels that are obtained from acceptable toxicological studies. Likewise, lowest and no observed effect concentrations from tests conducted on contaminant-spiked or amended soils have been employed to derive soil quality criteria for the protection of agricultural, residential, and other land uses (CCME 1993). Furthermore, Kenaga (1982) recommended the use of acute to chronic ratios (ACRs) for estimating long-term no effect concentrations of environmental contaminants in water from median lethal or effective endpoints. In this context, chronic no observed effect levels are used in conjunction with LC<sub>50</sub>s or EC<sub>50</sub>s to calculate the ACRs. Therefore, it was considered appropriate to modify the original procedure from Long and Morgan (1990) to facilitate incorporation of both the effects and the no effects data into the guidelines derivation process.

A variety of arithmetic procedures, which rely on calculations of specific percentiles of the effects and no effects data distributions, could be used to derive SQAGs. To develop a screening tool, the Florida Department of Environmental Protection identified the need to

develop two guideline values, which would define ranges of concentrations that would infrequently, sometimes, and frequently be associated with adverse effects. First, a low incidence of adverse effects was required in the minimal effects range. This objective could be achieved by determining chemical concentrations falling within the low end of the effects range (i.e., effects data set) and the middle of the no effects range (i.e., no effects data set). Subsequently, the mean of these values would establish a threshold effects level for each chemical. The geometric mean of these two values should be calculated because the two data distributions are not likely to be normally distributed. Using similar logic, the upper guideline value could be derived using a procedure that would establish a probable effects range that has a high incidence of adverse effects. This objective could be met by determining chemical concentrations falling within the middle of the effects range and the upper end of the no effects range. Likewise, the geometric mean of these values could be calculated to establish a probable effects level.

A sequential testing approach was used to determine which percentile values should be used to derive the numerical SQAGs. For example, the 5th, 10th, 15th, and 20th percentiles of the effects data set, in conjunction with the 50th percentile of the no effects data set, were all used to derive candidate TEL values for several indicator substances (cadmium, copper, fluoranthene, and phenanthrene). Several procedures for deriving the PEL were also proposed and tested in a similar manner. Subsequently, each of the candidate procedures was evaluated to determine which technique would be most appropriate for use in Florida.

The incidence of effects within each of the ranges of contaminant concentrations provides a powerful tool for evaluating the reliability of SQAGs (Long *et al.* In press). Therefore, the percent incidences of adverse effects within the minimal and probable effects ranges were calculated for each of the TELs and PELs, respectively, derived using the candidate procedures. The TELs were considered to be reliable if the incidence of adverse effects was less than 10% within the minimal effects range, while an incidence of adverse effects of 65% or more within the probable effects range was considered indicative of reliable PELs. Only those procedures that satisfied the criteria for reliability were further considered.

Next, the remaining procedures were evaluated in terms of their overall applicability. In this respect, candidate procedures were considered to be broadly applicable if they supported the derivation of guidelines that could be used to effectively screen sediment chemistry data at a large number of sites. That is, the guidelines would be most useful if they defined only a limited range of contaminant concentrations between the TEL and PEL. By tightening the possible effects range (i.e., reducing the range of uncertainty), the guidelines could be used to classify a larger number of samples in terms of their potential toxicity. Of all the procedures tested, the methods described in section 5.2 appeared to provide the most reliable and appropriate means of deriving SQAGs for Florida coastal waters.

There is a great deal of variability in the quantity of information available for each chemical, ranging from less than twenty data entries for 2,3,7,8-T<sub>4</sub>CDD to several hundred data entries for cadmium. Due to the uncertainty associated with the evaluation of matching sediment chemistry and biological effects data, minimum data requirements were established to ensure

that any guidelines developed were supported by the weight of evidence. To adhere to this principle, SQAGs were derived only for those contaminants which had at least 20 entries in both the effects and no effects data sets.

The number of data entries required to support the derivation of preliminary SQAGs was established based on the results of sequential calculations of guidelines for a total of four substances (cadmium, chromium, fluoranthene, and PCBs) using data sets of various sizes. Using the guideline derivation procedure described above, guidelines were sequentially calculated using randomly selected data sets of various sizes (ranging from 2 to 30 data entries). This procedure was repeated 10 times for each chemical to support the calculation of the mean guideline value and its standard deviation for each data set size. The results of this investigation indicated that the estimate of the guideline value stabilized when the data set contained 15 to 20 entries. The variability in this estimate was not significantly reduced over the range of 20 to 30 entries. Therefore, it was concluded that at least 20 entries from each data set were required to support the derivation of SQAGs.

## **5.4 Strengths and Limitations of the Recommended Approach for Developing Sediment Quality Assessment Guidelines**

The following discussion provides a general critique of the recommended approach for deriving SQAGs in the context of Florida's requirements.

### **5.4.1 Strengths of the Approach**

The WEA is characterized by a number of attributes that make it an attractive choice for deriving SQAGs for Florida coastal waters. First, the approach is supported by a comprehensive database on the biological effects of sediment-associated contaminants. Interpretation of the information contained in the expanded NSTP database provides relevant tools for evaluating the potential for biological effects at various contaminant concentrations.

Such interpretations are supported by detailed summaries of a large volume of data linking contaminant concentrations to biological effects (MacDonald *et al.* 1994). In this way, the WEA provides an effective basis for evaluating the degree of confidence that should be placed on the resultant guidelines. Confidence in these data is enhanced by the rigorous screening procedures that were used to evaluate candidate data sets.

Unlike other approaches to the development of SQAGs, the WEA does not attempt to establish absolute sediment quality assessment values. Instead, the approach delineates ranges of contaminant concentrations that are probably, possibly, and not likely to be associated with adverse biological effects. This approach explicitly recognizes the

uncertainty associated with the prediction of biological effects from chemical concentration data, thereby enhancing the defensibility of the guidelines for identifying priority conditions with respect to contaminated sediments.

One of the more important attributes of the WEA is its overall practicality. The expanded NSTP database supports the derivation of numerical SQAGs for a variety of chemical substances. Many of these chemicals are known to occur in Florida's coastal sediments at elevated levels. Sediment quality assessment guidelines are required to support assessments of the potential for biological effects in these sediments. In addition, by considering matching sediment chemistry and biological effects data from studies conducted in the field, the influence of mixtures of chemicals in sediments is incorporated in the resultant SQAGs. Furthermore, the information in the expanded NSTP database is highly relevant to the guidelines derivation process because it applies to a wide range of biological organisms and endpoints, and incorporates a large number of direct measurements on organisms that are normally associated with bed sediments. The database also includes a significant quantity of data from studies conducted in the southeastern United States (including Florida). These attributes are likely to give the SQAGs derived using the WEA broad applicability. As such, there is a high probability that these guidelines will be appropriate for implementation in Florida.

In addition to the other advantages of the approach, the procedure recommended for calculating SQAGs considers both the EDS and NEDS for each chemical constituent. And, in contrast to the AETA, this procedure does not rely heavily on individual data points. Therefore, outliers do not carry much weight in the overall guidelines derivation process. Integration of the effects and no effects data sets into a single data set allows preparation of ascending data tables for each contaminant. These data tables provide detailed information on specific biological responses that have been observed at various concentrations of the contaminant and, in themselves, can contribute significantly to sediment assessments. As such, these data tables are useful tools for evaluating the potential biological significance of contaminant concentrations that fall within the three ranges described above (i.e., no effects, possible effects, and probable effects ranges). Many reviewers of the original NSTP document (Long and Morgan 1990) indicated that the data tables (with contaminant concentrations arranged in ascending order) were extremely useful tools for evaluating sediment quality data from specific sites (E. Long, NOAA, Seattle, Washington. Personal communication).

In addition to supporting the derivation of numerical SQAGs, the expanded NSTP database provides a basis for evaluating the guidelines. In this respect, it is possible to use the database to calculate the distribution of effects and no effects entries within each range of contaminant concentrations (i.e., the minimal effects, possible effects, and probable effects ranges). These distributions describe the percent incidence of effects (i.e., number of biological effects entries divided by the total number of entries times 100) within each range of concentrations and, as such, provide an estimate of the probability of observing adverse biological effects when the concentration of a contaminant falls within a specific range of

concentrations (Long *et al.* In press). For example, if the incidence of effects within the probable effects range for cadmium was 68%, then there is a high likelihood that adverse biological effects would be observed at sites with concentrations of cadmium equalling or exceeding the PEL. This feature of the approach provides environmental managers a measure of confidence for ranking the relative priority of contaminated sediments. This tool can also be used within a risk assessment framework for assessing contaminated sediments.

Lastly, the WEA 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; Long *et al.* In press). Further, it has been described and evaluated in the *Sediment Classification Methods Compendium* (EPA 1992). 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 1993) as part of their guidelines derivation process. These favourable assessments emphasize the importance and utility of this procedure for deriving numerical SQAGs.

#### **5.4.2 Limitations of the Approach**

In spite of the obvious benefits associated with the WEA for deriving SQAGs, 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-associated contaminants. Ancillary data on grain size, levels of TOC, and concentrations of AVS were not provided in a large number of the reports reviewed. As such, it is not currently possible to express the guidelines in terms of the factors that could influence the bioavailability of these contaminants. 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 under- or over-protection of aquatic resources exists if guidelines are implemented that do not consider the bioavailability of sediment-associated contaminants.

The Florida Department of Environmental Protection (FDEP) recognizes the importance of accounting for the bioavailability of sediment-associated contaminants and is participating with NOAA in bio-effects studies to help address this issue. In addition, FDEP 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-associated metals (Schropp and Windom 1988; Schropp *et al.* 1989; Schropp *et al.* 1990). This tool emphasizes the importance of 'normalizers' in the interpretation of sediment chemistry data and provides a practical tool that complements the effects-based guidelines. A detailed

discussion on how these tools may be used together to assess sediment quality is provided in Volume 2.

It is anticipated that the bioavailability of sediment-associated contaminants will be one of the principal issues addressed in refining the preliminary guidelines. Currently, there is little comprehensive information with which to reliably predict the bioavailability of sediment-associated contaminants. Nonetheless, EPA has developed sediment quality criteria for five non-polar organic substances that are expressed on an OC-normalized basis. Data from other studies conducted in Florida and elsewhere may provide more relevant information for determining the factors that influence bioavailability in Florida coastal sediments.

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. For example, Jones-Lee and Lee (1993) identified ammonia as a factor that could potentially affect the toxicity of aquatic sediments. More recent results from more intensive monitoring in areas such as Tampa Bay and Los Angeles Harbor indicate that unionized ammonia levels in porewater were rarely in the range that would be associated with biological effects (Long Unpublished data). Nonetheless, variables such as ammonia, hydrogen sulfide, and dissolved oxygen, have the potential to affect benthic organisms and should be evaluated in site-specific assessments of sediment quality. In the WEA, the assembly of extensive information from numerous estuarine and marine sites in North America into a single database serves to minimize the impacts of unmeasured contaminants on the guidelines; however, there is still an undefined level of uncertainty associated with the application of the SQAGs at specific sites.

Application of the recommended approach may also be restricted by other limitations on the available information. Currently, limited data are available on the chronic toxicity of sediment-associated contaminants to marine and estuarine organisms. However, information on the responses of *in situ* benthic invertebrate communities to contaminant challenges partially mitigates this limitation on the database. In addition, only limited data are available on some potentially important sediment contaminants in Florida (including a variety of pesticides, dioxins and furans, etc.).

The results of this preliminary evaluation indicate that SQAGs 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. In particular, these guidelines may not be directly applicable to sediments with atypical levels of the factors that may influence the bioavailability of contaminants (e.g., very high or very low levels of TOC). Detailed discussions on the application of SQAGs in regional and site-specific sediment quality assessments are provided in Volume 2. In addition, a more

thorough evaluation of the SQAGs comparability, reliability, and predictability is presented in Chapter 7.

## **Chapter 6**

# **Numerical Sediment Quality Assessment Guidelines for Florida Coastal Waters**

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### **6.0 Introduction**

State water quality criteria are one of the major management tools for protecting designated uses of coastal ecosystems, including maintenance of acceptable conditions for living resources. While state water quality criteria provide effective tools for managing water quality, they provide little guidance for managing sediment quality. Therefore, numerical sediment quality assessment guidelines (SQAGs) are required to help address concerns relative to contamination of coastal ecosystems with substances that tend to be associated with bed sediments. In particular, there is a need for SQAGs that apply to substances that are known or suspected to be present in Florida coastal sediments.

This Chapter presents the numerical SQAGs that have been developed for assessing sediment quality in Florida coastal waters. In total, threshold effect levels (TELs) and probable effects levels (PELs) have been derived for 34 substances or groups of substances. In addition, an indication of the subjective degree of confidence that should be placed on the SQAGs is provided. The detailed evaluation of these guidelines is presented in Chapter 7.

### **6.1 Identification of Priority Contaminants in Florida Coastal Waters**

Generally, Florida's coast has not been subjected to extensive industrial developments. Therefore, the types of persistent, bioaccumulative, and highly toxic contaminants that are known to occur elsewhere in the United States are not likely to be distributed widely in its coastal areas. Nonetheless, various land uses and other coastal activities in the state have contributed significant quantities of environmental contaminants into coastal waters; therefore, sediments in the vicinity of major point and non-point sources may be severely contaminated. Concerns relative to the contamination of coastal ecosystems fall into four general categories: urban stormwater runoff; agricultural runoff; domestic wastewater; and, industrial wastewater (Hand *et al.* 1990). Consideration of each of these potential sources of environmental contaminants provides a basis for identifying chemical concerns in the Florida coastal zone.

It would be virtually impossible to develop SQAGs for every substance that could, potentially, be released into Florida coastal waters. For this reason, the evaluation of chemical concerns in Florida coastal systems is focused on identifying priority substances that are known to be released in significant quantities into receiving water systems and to form associations with sediments (Table 3). These substances are considered to be of highest priority with respect to developing numerical SQAGs 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 proximity of urban developments to the coast, urban stormwater represents a major source of contaminants into coastal ecosystems in Florida. Florida Department of Environmental Protection (FDEP 1994), 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 at elevated levels in Florida coastal sediments (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 Florida coastal ecosystems (Volume 2).

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, suspended solids, herbicides, insecticides, and other pesticides. While agricultural runoff is known to have 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 with significant potential to contaminate sediments in Florida's coastal areas are listed in Table 3. This list was assembled by considering present and historical pesticide use patterns (Pait *et al.* 1989), in conjunction with the physical/chemical properties of each substance (Worthing and Hance 1991). In addition, the pesticides which have been detected in coastal sediments (Long and Morgan 1990; Long *et al.* 1991; Delfino *et al.* 1991) or in aquatic biota (Trefry *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 potentially significant sources of environmental contaminants. While wastewater treatment plant (WWTP) upgrades have resulted in improved water quality in many areas, progress towards 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 generally not characterized by high densities of heavy manufacturing industries, substantial quantities of industrial wastewater are discharged into Florida waters

**Table 3. Preliminary identification of chemicals of concern in Florida coastal waters.**

| Substance                                      | Reference/Rationale   |
|--|---|
| <b>Metals</b>                                  |   |
| Arsenic  | Long et al. (1991); FDEP (1994).  |
| Cadmium  | Long et al. (1991); FDEP (1994).  |
| Chromium                                       | Long and Morgan (1990); Long et al. (1991); FDEP (1994).  |
| Copper   | Used in aquatic herbicides/found in fish; Long et al. (1991); Trefry et al. (1983); Leslie (1990); FDEP (1994). |
| Lead   | Long and Morgan (1990); Long et al. (1991); FDEP (1994).  |
| Mercury  | Long and Morgan (1990); Long et al. (1991); FDEP (1994).  |
| Nickel   | Long et al. (1991); FDEP (1994).  |
| Silver   | Long and Morgan (1990); FDEP (1994).  |
| Tributyltin                                    | Used as an antifoulant on ships.  |
| Zinc   | Long and Morgan (1990); Long et al. (1991); FDEP (1994).  |
| <b>Polycyclic Aromatic Hydrocarbons (PAHs)</b> |   |
| Acenaphthene                                   | Delfino et al. (1991); FDEP (1994).   |
| Acenaphthylene                                 | Delfino et al. (1991); FDEP (1994).   |
| Anthracene                                     | Long and Morgan (1990); Delfino et al. (1991); FDEP (1994).   |
| Benz(a)anthracene                              | Long and Morgan (1990); Delfino et al. (1991); FDEP (1994).   |
| Benzo(a)pyrene                                 | Long and Morgan (1990); Delfino et al. (1991); FDEP (1994).   |
| Chrysene                                       | Long and Morgan (1990); Delfino et al. (1991); FDEP (1994).   |
| Dibenzo(a,h)anthracene                         | Long and Morgan (1990); Delfino et al. (1991); FDEP (1994).   |
| Fluorene                                       | Long and Morgan (1990); Delfino et al. (1991); FDEP (1994).   |
| Fluoranthene                                   | FDEP (1994).  |
| Napthalene                                     | Long and Morgan (1990); Delfino et al. (1991); FDEP (1994).   |
| 2-methylnaphthalene                            | Long and Morgan (1990).   |
| Phenanthrene                                   | Long and Morgan (1990); Delfino et al. (1991); FDEP (1994).   |
| Pyrene   | Long and Morgan (1990); Delfino et al. (1991); FDEP (1994).   |
| Total PAHs                                     | Long and Morgan (1990); Long et al. (1991); FDEP (1994).  |
| <b>Polychlorinated Biphenyls (PCBs)</b>        |   |
| Total PCBs                                     | Long and Morgan (1990); Long et al. (1991); Delfino et al. (1991); FDEP (1994).                                 |

**Table 3. Preliminary identification of chemicals of concern in Florida coastal waters (continued).**

| Substance               | Reference/Rationale   |
|-------------------------|---|
| <b>Pesticides</b>       |   |
| Aldrin/Dieldrin         | Long and Morgan (1990); Long et al.(1991); FDEP (1994).                         |
| Azinphos-methyl (guthi) | Organophosphorous insecticide* (Kow > 10,000?)                                  |
| Chlordane               | Long and Morgan (1990); Long et al. (1991); FDEP (1994).                        |
| Chlorothalonil          | Chlorophenyl fungicide (Kow = 20,000)   |
| Chlorpyrifos            | Organophosphorous insecticide (Kow > 50,000)                                    |
| DDT and metabolites     | Long and Morgan (1990); Long et al. (1991); FDEP (1994); Delfino et al. (1991). |
| Disulfoton              | Organophosphorous insecticide (Kow > 10,000)                                    |
| Endosulfan              | Delfino et al. (1991); FDEP (1994).   |
| Endrin                  | Organochlorine insecticide (Kow > 10,000?); FDEP (1994).                        |
| Heptachlor              | Organochlorine insecticide (Kow > 10,000?); FDEP (1994).                        |
| Heptachlor epoxide      | Organochlorine insecticide (Kow > 10,000?); FDEP (1994).                        |
| Lindane (gamma-BHC)     | Organochlorine insecticide (Kow > 10,000?); FDEP (1994).                        |
| Mirex                   | Organochlorine insecticide (Kow > 10,000?); FDEP (1994).                        |
| Phorate                 | Organophosphorous insecticide (Kow > 10,000?).                                  |
| Quintozene (PCNB)       | Chlorophenyl fungicide (Kow = 10,000).  |
| Toxaphene (alpha-BHC)   | Organochlorine insecticide; FDEP (1994).  |
| Trifluralin             | Dinitroanaline herbicide (Kow > 200,000); FDEP (1994).                          |

\* Kow = Octanol-water partition coefficient which provides an indication of the hydrophobicity of a substance;  
Criteria for selection of pesticides: Kow > 5,000.

**Chlorinated Organic Compounds**

|                   |                                     |
|-------------------|-------------------------------------|
| 2,3,7,8-T4CDD     | Pulp and paper industry.            |
| 2,3,7,8-T4CDF     | Pulp and paper industry             |
| Pentachlorophenol | Delfino et al. (1991); FDEP (1994). |

**Phthalates**

|                            |                        |
|----------------------------|------------------------|
| Bis(2-ethylhexyl)phthalate | Delfino et al. (1991). |
| Dimethyl phthalate         | Delfino et al. (1991). |
| Di-n-butylphthalate        | Delfino et al. (1991). |

(Farrow 1990). The major sources of these effluents are the 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 are likely to have resulted in the release 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 typically associated with specific industrial wastewaters).

## 6.2 Numerical Sediment Quality Assessment Guidelines

Numerical sediment quality assessment guidelines have been developed for a total of 34 high priority substances in Florida. Using the procedures described in Chapter 5, a threshold effect level and a probable effect level were derived using the information contained in the expanded NSTP database. These numerical guidelines, which are expressed on a dry weight basis, are presented in Table 4. A brief discussion of the sources, fate, and effects of each substance (or group of substances) has also been prepared to provide additional information for applying the SQAGs. Lastly, the results of the evaluation of the reliability of the SQAGs are expressed as "high", "medium", and "low confidence" to provide guidance on the application of the SQAGs. However, the reader is urged to read Chapter 7 for the details of this evaluation, as well as an assessment of the comparability and predictability of the SQAGs.

### 6.2.1 Metals

Numerical SQAGs have been derived for nine metals that occur in Florida coastal sediments. As is the case for the other substances, the SQAGs are reported on a dry weight basis. While it is possible that further research could support the derivation of effects-based guidelines that are expressed in terms of the factors that influence bioavailability, such as acid volatile sulfide (AVS), the necessary data are not yet available. As discussed in Chapter 5, the preliminary guidelines should be used in conjunction with other assessment tools (such as the FDEP metals interpretive tool described in Volume 2) to evaluate sediment quality conditions in coastal waters.

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

| Substance  | Total Number<br>of Records | Number of Entries<br>in the EDS | Number of Entries<br>in the NEDS | Sediment Quality Assessment Guidelines |       |
|--|----------------------------|---------------------------------|----------------------------------|--|-------|
|  |                            |                                 |                                  | TEL                                    | PEL   |
| <b>Metals (SQAGs in mg/kg)</b>                                 |                            |                                 |                                  |  |       |
| Arsenic  | 295                        | 39                              | 256                              | 7.24                                   | 41.6  |
| Cadmium  | 433                        | 107                             | 326                              | 0.676                                  | 4.21  |
| Chromium   | 354                        | 53                              | 301                              | 52.3                                   | 160   |
| Copper   | 440                        | 105                             | 335                              | 18.7                                   | 108   |
| Lead   | 402                        | 95                              | 307                              | 30.2                                   | 112   |
| Mercury  | 331                        | 66                              | 265                              | 0.13                                   | 0.696 |
| Nickel   | 355                        | 23                              | 332                              | 15.9                                   | 42.8  |
| Silver   | 190                        | 35                              | 155                              | 0.733                                  | 1.77  |
| Tributyltin  | 72                         | 6                               | 66                               | ID                                     | ID    |
| Zinc   | 411                        | 96                              | 315                              | 124                                    | 271   |
| <b>Polychlorinated Biphenyls (PCBs; SQAGs in µg/kg)</b>        |                            |                                 |                                  |  |       |
| Total PCBs   | 199                        | 65                              | 134                              | 21.6                                   | 189   |
| <b>Polycyclic Aromatic Hydrocarbons (PAHs; SQAGs in µg/kg)</b> |                            |                                 |                                  |  |       |
| Acenaphthene   | 240                        | 62                              | 178                              | 6.71                                   | 88.9  |
| Acenaphthylene   | 209                        | 36                              | 173                              | 5.87                                   | 128   |
| Anthracene   | 259                        | 70                              | 189                              | 46.9                                   | 245   |
| Fluorene   | 263                        | 73                              | 190                              | 21.2                                   | 144   |
| 2-methylnaphthalene  | 189                        | 40                              | 149                              | 20.2                                   | 201   |
| Naphthalene  | 256                        | 57                              | 199                              | 34.6                                   | 391   |
| Phenanthrene   | 268                        | 74                              | 194                              | 86.7                                   | 544   |
| Sum LMW-PAHs   | 274                        | 69                              | 205                              | 312                                    | 1442  |

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

| Substance  | Total Number<br>of Records | Number of Entries<br>in the EDS | Number of Entries<br>in the NEDS | Sediment Quality Assessment Guidelines |       |
|--|----------------------------|---------------------------------|----------------------------------|--|-------|
|  |                            |                                 |                                  | TEL                                    | PEL   |
| <b>Polycyclic Aromatic Hydrocarbons (PAHs; SQAGs in µg/kg)</b> |                            |                                 |                                  |  |       |
| Benz(a)anthracene  | 249                        | 63                              | 186                              | 74.8                                   | 693   |
| Benzo(a)pyrene   | 259                        | 68                              | 191                              | 88.8                                   | 763   |
| Chrysene   | 258                        | 68                              | 190                              | 108                                    | 846   |
| Dibenzo(a,h)anthracene   | 246                        | 54                              | 192                              | 6.22                                   | 135   |
| Fluoranthene   | 279                        | 85                              | 194                              | 113                                    | 1494  |
| Pyrene   | 263                        | 70                              | 193                              | 153                                    | 1398  |
| Sum HMW-PAHs   | 274                        | 64                              | 210                              | 655                                    | 6676  |
| Total PAHs   | 250                        | 58                              | 192                              | 1684                                   | 16770 |
| <b>Pesticides (SQAGs in µg/kg)</b>                             |                            |                                 |                                  |  |       |
| Aldrin   | 180                        | 15                              | 165                              | ID                                     | ID    |
| Azinphos-methyl (Guthion)                                      | 0                          | 0                               | 0                                | ID                                     | ID    |
| Chlordane  | 203                        | 25                              | 178                              | 2.26                                   | 4.79  |
| Chlorthalonil  | 0                          | 0                               | 0                                | ID                                     | ID    |
| Chlorpyrifos   | 1                          | 1                               | 0                                | ID                                     | ID    |
| p,p'-DDD   | 173                        | 22                              | 151                              | 1.22                                   | 7.81  |
| p,p'-DDE   | 211                        | 37                              | 174                              | 2.07                                   | 374   |
| p,p'-DDT   | 175                        | 26                              | 149                              | 1.19                                   | 4.77  |
| Total DDT  | 89                         | 37                              | 52                               | 3.89                                   | 51.7  |
| Dieldrin   | 181                        | 25                              | 156                              | 0.715                                  | 4.3   |
| Disulfoton   | 0                          | 0                               | 0                                | ID                                     | ID    |
| Endosulfan   | 6                          | 4                               | 2                                | ID                                     | ID    |
| Endrin   | 146                        | 14                              | 132                              | ID                                     | ID    |

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

| Substance  | Total Number<br>of Records | Number of Entries<br>in the EDS | Number of Entries<br>in the NEDS | Sediment Quality Assessment Guidelines |      |
|--|----------------------------|---------------------------------|----------------------------------|--|------|
|  |                            |                                 |                                  | TEL                                    | PEL  |
| <b>Pesticides (SQAGs in µg/kg)</b>                     |                            |                                 |                                  |  |      |
| Heptachlor   | 168                        | 14                              | 154                              | ID                                     | ID   |
| Heptachlor epoxide                                     | 137                        | 9                               | 128                              | ID                                     | ID   |
| Lindane (gamma-BHC)                                    | 181                        | 21                              | 160                              | 0.32                                   | 0.99 |
| Mirex  | 120                        | 3                               | 117                              | ID                                     | ID   |
| Phorate  | 0                          | 0                               | 0                                | ID                                     | ID   |
| Quintozene (PCNB)                                      | 0                          | 0                               | 0                                | ID                                     | ID   |
| Toxaphene (alpha-BHC)                                  | 133                        | 4                               | 129                              | ID                                     | ID   |
| Trifluralin  | 0                          | 0                               | 0                                | ID                                     | ID   |
| <b>Chlorinated Organic Substances (SQAGs in µg/kg)</b> |                            |                                 |                                  |  |      |
| 2,3,7,8-Tetrachlorodibenzo-p-dioxin                    | 18                         | 2                               | 16                               | ID                                     | ID   |
| 2,3,7,8-Tetrachlorodibenzofuran                        | 17                         | 1                               | 16                               | ID                                     | ID   |
| Pentachlorophenol                                      | 82                         | 7                               | 75                               | ID                                     | ID   |
| <b>Phthalates (SQAGs in µg/kg)</b>                     |                            |                                 |                                  |  |      |
| Bis(2-ethylhexyl)phthalate                             | 131                        | 31                              | 100                              | 182                                    | 2647 |
| Dimethyl phthalate                                     | 86                         | 10                              | 76                               | ID                                     | ID   |
| Di-n-butyl phthalate                                   | 79                         | 7                               | 72                               | ID                                     | ID   |

Total Number of Records = Number of data records in the expanded biological effects database for sediments.

All of the sediment quality assessment guidelines are expressed on a dry weight basis, as potential normalizers (e.g., Al, TOC, AVS) were rarely reported.

EDS = Effects data set; NEDS = No effects data set; TEL = Toxic effect level; PEL = Probable effect level.

ID = insufficient data to derive sediment quality assessment guidelines.

SQAG = Sediment quality assessment guidelines

### **Arsenic**

Arsenic is released naturally into the environment due to the weathering of arsenic-rich rocks and volcanic activity. In addition to the natural sources of this substance, however, arsenic is released into the environment as a result of human activities. For example, arsenic is used in pigments, for medical purposes, in glass making, 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 co-precipitated 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 co-precipitates 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 web (Jaagumagi 1990a).

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.

Consideration of the available information on the toxicity of sediment-associated arsenic to aquatic biota results in the derivation of a ***TEL of 7.2 mg/kg and a PEL of 41.6 mg/kg***. An evaluation of the reliability of the SQAGs for arsenic suggests that a moderate degree of confidence can be placed on these guidelines.

### **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 co-precipitation 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) presented evidence on the role of AVS in controlling the availability of cadmium. In general, cadmium is considered to have an extensive residence time and can accumulate to significant levels in biological tissues (Jaagumagi 1990a).

Exposure of aquatic organisms to cadmium can result in various 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 behavior of marine amphipods have also been observed in spiked-sediment bioassays with cadmium (Long and Morgan 1990).

Consideration of the available information on the toxicity of sediment-associated cadmium to aquatic biota results in the derivation of a ***TEL of 0.68 mg/kg and a PEL of 4.2 mg/kg.*** An evaluation of the reliability of the SQAGs for cadmium suggests that a high degree of confidence can be placed on these guidelines.

### ***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 co-precipitates 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 ( $BCF < 3$ ; BCFs, bioconcentration factors are the ratio of tissue concentrations to concentrations in water), 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).

Consideration of the available information on the toxicity of sediment-associated chromium to aquatic biota results in the derivation of a ***TEL of 52.3 mg/kg and a PEL of 160 mg/kg.*** An evaluation of the reliability of the SQAGs for chromium suggests that a moderate degree of confidence can be placed on these guidelines.

### **Copper**

Copper is a common metallic element in crustal rocks and minerals. Natural sources of copper 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 can 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 co-precipitates 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).

Consideration of the available information on the toxicity of sediment-associated copper to aquatic biota results in the derivation of a ***TEL of 18.7 mg/kg and a PEL of 108 mg/kg***. An evaluation of the reliability of the SQAGs for copper suggests that a moderate degree of confidence can be placed on these guidelines.

### ***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. 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 in algae may be as high as 20,000; however, BCFs for fish and invertebrates tend to be much lower (500 to 1700; CCREM 1987).

Consideration of the available information on the toxicity of sediment-associated lead to aquatic biota results in the derivation of a ***TEL of 30.2 mg/kg and a PEL of 112***

**mg/kg.** An evaluation of the reliability of the SQAGs for lead suggests that a moderate degree of confidence can be placed on these guidelines.

### ***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). Anthropogenic sources to aquatic ecosystems can include waste incineration, coal combustion, paints, mining and smelting, 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.

Consideration of the available information on the toxicity of sediment-associated mercury to aquatic biota results in the derivation of a ***TEL of 0.13 mg/kg and a PEL of 0.70 mg/kg.*** An evaluation of the reliability of the SQAGs for mercury suggests that a low degree of confidence can be placed on these guidelines.

### ***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 industrial processes and in oil refining. More recently, it has been

used in nuclear power 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 co-precipitation 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).

Consideration of the available information on the toxicity of sediment-associated nickel to aquatic biota results in the derivation of **a TEL of 15.9 mg/kg and a PEL of 42.8 mg/kg**. An evaluation of the reliability of the SQAGs for nickel suggests that a low degree of confidence can be placed on these guidelines.

### *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 chemical species. Silver has a fairly low potential to accumulate in aquatic organisms, with BCFs ranging from less than 1 to 240 (CCREM 1987).

Consideration of the available information on the toxicity of sediment-associated silver to aquatic biota results in the derivation of a ***TEL of 0.73 mg/kg and a PEL of 1.77 mg/kg***. An evaluation of the reliability of the SQAGs for silver suggests that a moderate degree of confidence can be placed on these guidelines.

### ***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 SQAGs (TEL 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 a 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).

Consideration of the available information on the toxicity of sediment-associated zinc to aquatic biota results in the derivation of a ***TEL of 124 mg/kg and a PEL of 271 mg/kg.*** An evaluation of the reliability of the SQAGs for zinc suggests that a moderate degree of confidence can be placed on these guidelines.

### 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.* 1992). In addition, WWTP effluents and runoff from urban areas, particularly from roads, are known to contain significant quantities of PAHs. Furthermore, 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, phenanthrene, and pyrene (Delfino *et al.* 1991). In general, elevated levels of sediment-associated 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 may depend on the concentration of TOC in the sediment (Bolton *et al.* 1985; Lyman *et al.* 1987).

### ***Acenaphthene***

Consideration of the available information on the toxicity of sediment-associated acenaphthene to aquatic biota results in the derivation of a ***TEL of 6.7 µg/kg and a PEL of 88.9 µg/kg***. An evaluation of the reliability of the SQAGs for acenaphthene suggests that a moderate degree of confidence can be placed on these guidelines.

### ***Acenaphthylene***

Consideration of the available information on the toxicity of sediment-associated acenaphthylene to aquatic biota results in the derivation of a ***TEL of 5.9 µg/kg and a PEL of 128 µg/kg***. An evaluation of the reliability of the SQAGs for acenaphthylene suggests that a moderate degree of confidence can be placed on these guidelines.

### ***Anthracene***

Consideration of the available information on the toxicity of sediment-associated anthracene to aquatic biota results in the derivation of a ***TEL of 46.9 µg/kg and a PEL of 245 µg/kg***. An evaluation of the reliability of the SQAGs for anthracene suggests that a high degree of confidence can be placed on these guidelines.

### ***Fluorene***

Consideration of the available information on the toxicity of sediment-associated fluorene to aquatic biota results in the derivation of a ***TEL of 21.2 µg/kg and a PEL of 144 µg/kg***. An evaluation of the reliability of the SQAGs for fluorene suggests that a moderate degree of confidence can be placed on these guidelines.

### ***2-methylnaphthalene***

Consideration of the available information on the toxicity of sediment-associated 2-methylnaphthalene to aquatic biota results in the derivation of a ***TEL of 20.2 µg/kg and a PEL of 201 µg/kg***. An evaluation of the reliability of the SQAGs for 2-methylnaphthalene suggests that a high degree of confidence can be placed on these guidelines.

### ***Naphthalene***

Consideration of the available information on the toxicity of sediment-associated naphthalene to aquatic biota results in the derivation of a ***TEL of 34.6 µg/kg and a PEL of 391 µg/kg***. An evaluation of the reliability of the SQAGs for naphthalene suggests that a high degree of confidence can be placed on these guidelines.

### ***Phenanthrene***

Consideration of the available information on the toxicity of sediment-associated phenanthrene to aquatic biota results in the derivation of a ***TEL of 86.7 µg/kg and a PEL of 544 µg/kg***. An evaluation of the reliability of the SQAGs for phenanthrene suggests that a high degree of confidence can be placed on these guidelines.

### ***Total 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). Consideration of the available information on the toxicity of sediment-associated total LMW PAHs to aquatic biota results in the derivation of a ***TEL of 312 µg/kg and a PEL of 1440 µg/kg***. An evaluation of the reliability of the SQAGs for total LMW PAHs suggests that a high degree of confidence can be placed on these guidelines.

### ***Benz(a)anthracene***

Consideration of the available information on the toxicity of sediment-associated benz(a)anthracene to aquatic biota results in the derivation of a ***TEL of 74.8 µg/kg and a PEL of 693 µg/kg.*** An evaluation of the reliability of the SQAGs for benz(a)anthracene suggests that a moderate degree of confidence can be placed on these guidelines.

### ***Benzo(a)pyrene***

Consideration of the available information on the toxicity of sediment-associated benzo(a)pyrene to aquatic biota results in the derivation of a ***TEL of 88.8 µg/kg and a PEL of 763 µg/kg.*** An evaluation of the reliability of the SQAGs for benzo(a)pyrene suggests that a high degree of confidence can be placed on these guidelines.

### ***Chrysene***

Consideration of the available information on the toxicity of sediment-associated chrysene to aquatic biota results in the derivation of a ***TEL of 108 µg/kg and a PEL of 846 µg/kg.*** An evaluation of the reliability of the SQAGs for chrysene suggests that a high degree of confidence can be placed on these guidelines.

### ***Dibenzo(a,h)anthracene***

Consideration of the available information on the toxicity of sediment-associated dibenzo(a,h)anthracene to aquatic biota results in the derivation of a ***TEL of 6.2 µg/kg and a PEL of 135 µg/kg.*** An evaluation of the reliability of the SQAGs for dibenzo(a,h)anthracene suggests that a moderate degree of confidence can be placed on these guidelines.

### ***Fluoranthene***

Consideration of the available information on the toxicity of sediment-associated fluoranthene to aquatic biota results in the derivation of a ***TEL of 113 µg/kg and a PEL of 1490 µg/kg.*** An evaluation of the reliability of the SQAGs for fluoranthene suggests that a high degree of confidence can be placed on these guidelines.

### ***Pyrene***

Consideration of the available information on the toxicity of sediment-associated pyrene to aquatic biota results in the derivation of a ***TEL of 153 µg/kg and a PEL of 1400 µg/kg***. An evaluation of the reliability of the SQAGs for pyrene suggests that a high degree of confidence can be placed on these guidelines.

### ***Total 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). Consideration of the available information on the toxicity of sediment-associated total HMW PAHs to aquatic biota results in the derivation of a ***TEL of 655 µg/kg and a PEL of 6680 µg/kg***. An evaluation of the reliability of the SQAGs for total HMW PAHs suggests that a moderate degree of confidence can be placed on these guidelines.

### ***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). Consideration of the available information on the toxicity of sediment-associated total PAHs to aquatic biota results in the derivation of a ***TEL of 1680 µg/kg and a PEL of 16800 µg/kg***. An evaluation of the reliability of the SQAGs for total PAHs suggests that a high degree of confidence can be placed on these guidelines.

### **6.2.3 Polychlorinated Biphenyls**

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 may depend 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  $4.4 \times 10^7$  in laboratory studies and biomagnification in higher trophic levels has been demonstrated (Moore and Walker 1991).

Consideration of the available information on the toxicity of sediment-associated total PCBs to aquatic biota results in the derivation of a ***TEL of 21.6 µg/kg and a PEL of 189 µg/kg.*** An evaluation of the reliability of the SQAGs for total PCBs suggests that a low degree of confidence can be placed on these guidelines.

#### 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 sediments is provided in Table 3. 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<sub>ow</sub>), and existing sediment quality monitoring data (Long and Morgan 1990; Long *et al.* 1991; Delfino *et al.* 1991).

Sufficient toxicological data exist to develop SQAGs 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 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).

Consideration of the available information on the toxicity of sediment-associated dieldrin to aquatic biota results in the derivation of a ***TEL of 0.72 µg/kg and a PEL of 4.3 µg/kg***. An evaluation of the reliability of the SQAGs for dieldrin suggests that a moderate degree of confidence can be placed on these guidelines.

### ***Azinphos-methyl***

Insufficient data were available to develop SQAGs for azinphos-methyl, 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).

Consideration of the available information on the toxicity of sediment-associated total chlordane to aquatic biota results in the derivation of a ***TEL of 2.3 µg/kg and a PEL of 4.8 µg/kg***. An evaluation of the reliability of the SQAGs for chlordane suggests that a low degree of confidence can be placed on these guidelines.

#### ***Chlorthalonil***

Insufficient data were available to develop SQAGs for chlorthalonil.

#### ***Chlorpyrifos***

Insufficient data were available to develop SQAGs 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***

Consideration of the available information on the toxicity of sediment-associated p,p'-DDD to aquatic biota results in the derivation of a ***TEL of 1.2 µg/kg and a PEL of 7.8 µg/kg***. An evaluation of the reliability of the SQAGs for p,p'-DDD suggests that a moderate degree of confidence can be placed on these guidelines.

### *p,p'-DDE*

Consideration of the available information on the toxicity of sediment-associated p,p'-DDE to aquatic biota results in the derivation of a ***TEL of 2.1 µg/kg and a PEL of 374 µg/kg***. An evaluation of the reliability of the SQAGs for p,p'-DDE suggests that a moderate degree of confidence can be placed on these guidelines.

### *p,p'-DDT*

Consideration of the available information on the toxicity of sediment-associated p,p'-DDT to aquatic biota results in the derivation of a ***TEL of 1.2 µg/kg and a PEL of 4.8 µg/kg***. An evaluation of the reliability of the SQAGs for p,p'-DDT suggests that a moderate degree of confidence can be placed on these guidelines.

### *Total DDT*

Consideration of the available information on the toxicity of sediment-associated total DDT (the sum of the concentrations of *p,p'-DDT*, *o,p'-DDT*, *p,p'-DDE*, *o,p'-DDE*, *p,p'-DDD*, *o,p'-DDD*) to aquatic biota results in the derivation of a ***TEL of 3.89 µg/kg and a PEL of 51.7 µg/kg***. An evaluation of the reliability of the SQAGs for total DDT suggests that a low degree of confidence can be placed on these guidelines.

### *Disulfoton*

Insufficient data were available to develop SQAGs for disulfoton.

### *Endosulfan*

Insufficient data were available to develop SQAGs for endosulfan. McLeese *et al.* (1982) reported a 12 day LC<sub>50</sub> of 340 µg/kg for the sandworm, *Nereis virens*. Chandler and Scott (1991) reported effects on colonization of polychaetes in South Carolina at 50 µg/kg and mortality to copepods at 200 µg/kg.

### *Endrin*

Insufficient data were available to develop SQAGs 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). More recently, Hansen *et al.* (1993a) reported a chronic SQC of 7.6 µg/kg at 1% OC.

### ***Heptachlor***

Insufficient data were available to develop SQAGs 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 SQAGs for heptachlor epoxide.

### ***Lindane***

Consideration of the available information on the toxicity of sediment-associated lindane (gamma-BHC) to aquatic biota results in the derivation of a ***TEL of 0.32 µg/kg and a PEL of 0.99 µg/kg***. An evaluation of the reliability of the SQAGs for lindane suggests that a low degree of confidence can be placed on these guidelines.

### ***Mirex***

Insufficient data were available to develop SQAGs for mirex.

### ***Phorate***

Insufficient data were available to develop SQAGs for phorate.

### ***Toxaphene***

Insufficient data were available to develop SQAGs 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 SQAGs 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. 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; EPA 1985; NRCC 1981; NRCC 1984). Dioxins and furans 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). Dioxins and furans may also be formed during the disinfection of complex effluents (e.g. pulp and paper effluents) containing many organic constituents.

Dioxins and furans 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).

Dioxins and furans may be distributed throughout the environment via air, water, soil, and sediments. Dioxins and furans 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 SQAGs for any of the 75 dioxin or furan congeners that could be present in Florida coastal sediments.

### *Pentachlorophenol*

Insufficient data were available to develop SQAGs 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 SQAGs for these substances are required to assess the hazards posed to aquatic organisms.

#### *Bis(2-ethylhexyl)phthalate*

Consideration of the available information on the toxicity of sediment-associated bis(2-ethylhexyl)phthalate to aquatic biota results in the derivation of a ***TEL of 182 µg/kg and a PEL of 2650 µg/kg***. An evaluation of the reliability of the SQAGs for bis(2-ethylhexyl)phthalate suggests that a high degree of confidence can be placed on these guidelines.

#### *Dimethyl phthalate*

Insufficient data were available to develop SQAGs 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 SQAGs 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 Evaluation of the Sediment Quality Assessment Guidelines for Florida Coastal Waters**

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#### **7.0 Introduction**

The numerical sediment quality assessment guidelines (SQAGs) presented in Chapter 6 were derived to help FDEP assess sediment quality in Florida coastal waters. When used with high quality sediment chemistry data and in conjunction with other assessment techniques, these guidelines provide a sound technical basis for assessing the potential for adverse biological effects in contaminated sediments (see Volume 2 for more information on the potential uses of the SQAGs). This chapter provides further information on the reliability, comparability, and predictive ability of these management tools. The following evaluation should be reviewed by potential users to judge the appropriateness of the SQAGs for their specific situations.

A variety of procedures have been used by state and federal agencies to evaluate the applicability of SQAGs. For example, the California State Water Resources Control Board (Lorenzato *et al.* 1991) considered it important to evaluate the precision or comparability of SQAGs. The Board defined precision as the level of agreement among guidelines based on different data and/or different approaches. Long *et al.* (In press) considered that reliability was an essential attribute of SQAGs and evaluated reliability by determining the extent to which guidelines satisfied previously established narrative objectives (i.e., does the PEL define the lower limit of a range of contaminant concentrations that is usually associated with adverse biological effects). In turn, this information may be used to assess the degree of confidence that could be placed in numerical SQAGs for each substance. Finally, SQAGs have been evaluated in terms of their ability to identify impacted and unimpacted sites, using sediment chemistry data alone (PTI 1988). The present evaluation addresses all three of these considerations and includes the following elements:

- ▶ An assessment of the comparability of the SQAGs with similar assessment tools derived using different approaches and procedures;
- ▶ An assessment of the reliability of the SQAGs using the matching sediment chemistry and biological effects data contained within the expanded NSTP database; and,

- ▶ An assessment of the predictive capability of the SQAGs using independent data sets (i.e., not included in the expanded NSTP database) from the southeast and elsewhere in the United States.

The methods used for each of these assessments and their results are discussed in the following sections. Potential users should consider the results of these evaluations before using the SQAGs in the applications outlined in Volume 2.

## 7.1 Comparability of the Sediment Quality Assessment Guidelines

Several governmental agencies in North America have derived sediment quality criteria, guidelines, objectives, or standards to support the assessment, management, and/or remediation of coastal sediments. These sediment management tools, developed using a wide array of approaches and procedures, often span several orders of magnitude for any given substance (MacDonald *et al.* 1992; Haines *et al.* 1994). Nonetheless, many of these guidelines share common assumptions. For example, certain sediment quality guidelines are assumed to provide a high level of protection for marine organisms, while other guidelines are intended to identify the concentrations of contaminants above which adverse effects are expected. Determination of the level of agreement among similar sediment quality guidelines provides a basis for assessing their comparability (Long *et al.* In press).

In this section, the SQAGs are compared with various sediment quality criteria, guidelines, objectives, and standards developed for other applications. The first step in this process involved collating available North American sediment quality assessment values. The expanded NSTP database was a major source of these assessment values; however, several documents containing compilations of sediment quality criteria, guidelines, objectives, and standards also were used (MacDonald *et al.* 1992; Haines *et al.* 1994). Next, the guideline values that were similar to the threshold effect levels (TELs) were identified. Likewise, guidelines that were similar to the PELs were assembled for each substance. Freshwater guidelines were not considered in this evaluation. Subsequently, the guideline values from each source were tabulated on a substance by substance basis for comparison (Tables 5 and 6).

Threshold effect levels derived from the expanded NSTP database are considered to provide a high level of protection for aquatic organisms. A total of four sets of similar guidelines were identified for comparison with the TELs, including:

- ▶ The effects range-low values (ER-Ls) promulgated under the NOAA National Status and Trends Program (Long and Morgan 1990);

**Table 5. A comparison of sediment quality assessment guidelines (SQAGs) applicable to coastal and marine waters: Threshold effect level (TEL) equivalents.**

| Substance  | TEL  | ER-L  | PSDDA - SL | SQC - Chronic | SQO   | Number of SQAGs Comparable to TEL |
|--|------|-------|------------|---------------|-------|-----------------------------------|
| <b>Metals (SQAGs in mg/kg)</b>                                 |      |       |            |               |       |                                   |
| Arsenic  | 7.24 | 8.2*  | 70         | 8.2*          | 20    | 2                                 |
| Cadmium  | 0.68 | 1.2*  | 0.96*      | 7.7           | 1*    | 3                                 |
| Chromium   | 52.3 | 81*   | NG         | NG            | 60*   | 2                                 |
| Copper   | 18.7 | 34*   | 81         | 34*           | 100   | 2                                 |
| Lead   | 30.2 | 46.7* | 66*        | 33*           | 30*   | 4                                 |
| Mercury  | 0.13 | 0.15* | 0.21*      | 0.01          | 0.15* | 3                                 |
| Nickel   | 15.9 | 20.9* | NG         | NG            | 45*   | 2                                 |
| Silver   | 0.73 | 1*    | 1.2*       | NG            | NG    | 2                                 |
| Zinc   | 124  | 150*  | 160*       | 190*          | 150*  | 4                                 |
| <b>Polycyclic Aromatic Hydrocarbons (PAHs; SQAGs in µg/kg)</b> |      |       |            |               |       |                                   |
| Acenaphthene   | 6.71 | 16*   | 63         | 2400          | 50    | 1                                 |
| Acenaphthylene   | 5.87 | 44    | 64         | NG            | 60    | 0                                 |
| Anthracene   | 46.9 | 85.3* | 130*       | 190           | 10    | 2                                 |
| Fluorene   | 21.2 | 19*   | 64         | 59            | 50*   | 2                                 |
| Naphthalene  | 34.6 | 160   | 210        | 500           | 200   | 0                                 |
| 2-Methylnaphthalene  | 20.2 | 70    | 67         | NG            | NG    | 0                                 |
| Phenanthrene   | 86.7 | 240*  | 320        | 2400          | 15    | 1                                 |
| Total Low Molecular Weight PAHs                                | 312  | 552*  | 610*       | NG            | NG    | 2                                 |
| Benz(a)anthracene  | 74.8 | 261   | 450        | 1600          | 130*  | 1                                 |
| Benzo(a)pyrene   | 88.8 | 430   | 680        | 18000         | 160*  | 1                                 |
| Chrysene   | 108  | 384   | 670        | 1200          | 140*  | 1                                 |
| Dibenz(a,h)anthracene  | 6.22 | 63.4  | 120        | 12000         | 60    | 0                                 |

**Table 5. A comparison of sediment quality assessment guidelines (SQAGs) applicable to coastal and marine waters: Threshold effect level (TEL) equivalents (continued).**

| Substance   | TEL  | ER-L  | PSDDA - SL | SQC - Chronic | SQO  | Number of SQAGs Comparable to TEL |
|---|------|-------|------------|---------------|------|-----------------------------------|
| <b>Polycyclic Aromatic Hydrocarbons (PAHs; SQAGs in µg/kg; cont.)</b> |      |       |            |               |      |                                   |
| Fluoranthene  | 113  | 600   | 630        | 1600          | 170* | 1                                 |
| Pyrene  | 153  | 665   | 430*       | 850           | 260* | 2                                 |
| Total High Molecular Weight PAHs                                      | 655  | 1700* | 1800*      | NG            | NG   | 2                                 |
| Total PAHs  | 1684 | 4022* | NG         | NG            | NG   | 1                                 |
| <b>Polychlorinated Biphenyls (PCBs; SQAGs in µg/kg)</b>               |      |       |            |               |      |                                   |
| Total PCBs  | 21.6 | 22.7* | 130        | NG            | 30*  | 2                                 |
| <b>Pesticides (SQAGs in µg/kg)</b>                                    |      |       |            |               |      |                                   |
| Chlordane   | 2.26 | 0.5   | NG         | 0.3           | NG   | 0                                 |
| Dieldrin  | 0.72 | 0.02  | NG         | 200           | NG   | 0                                 |
| p,p'-DDD  | 1.22 | 2*    | NG         | NG            | NG   | 1                                 |
| p,p'-DDE  | 2.07 | 2.2*  | NG         | NG            | NG   | 1                                 |
| p,p'-DDT  | 1.19 | 1*    | NG         | 1.6*          | NG   | 2                                 |
| Total DDT   | 3.89 | 1.58* | NG         | 1.6*          | NG   | 2                                 |
| Lindane (gamma-BHC)   | 0.32 | NG    | NG         | 3.1           | NG   | 0                                 |
| <b>Phthalates (SQAGs in µg/kg)</b>                                    |      |       |            |               |      |                                   |
| Bis(2-ethylhexyl)phthalate  | 182  | NG    | 3100       | NG            | NG   | 0                                 |

ER-L = Effects range low (Long and Morgan 1990).

PSDDA - SL = Screening level used in the Puget Sound Dredged Disposal Analysis program (USACOE 1988).

SQC - Chronic = Chronic sediment quality criterion (EqPA; Lyman et al. 1987; Pavlou et al. 1987; Hansen et al. 1993a,b,c,d,e).

SQO = Sediment quality objective (Swain and Nijman 1991).

\* = Indicates that the SQAGs are within a factor of three of the PEL.

NG = No guideline.

**Table 6. A comparison of sediment quality guidelines (SQAGs) applicable to coastal and marine waters: Probable effect level (PEL) equivalents.**

| Substance  | PEL  | ER-M  | LAET   | SQC - Acute | SLC   | Number of SQAGs Comparable to PEL |
|--|------|-------|--------|-------------|-------|-----------------------------------|
| <b>Metals (SQAGs in mg/kg)</b>                                 |      |       |        |             |       |                                   |
| Arsenic  | 41.6 | 70*   | 57*    | 16          | NG    | 2                                 |
| Cadmium  | 4.21 | 9.6*  | 5.1*   | 24          | NG    | 2                                 |
| Chromium   | 160  | 370*  | 260*   | NG          | NG    | 2                                 |
| Copper   | 108  | 270*  | 390    | 54          | NG    | 1                                 |
| Lead   | 112  | 218*  | 450    | 840         | NG    | 1                                 |
| Mercury  | 0.7  | 0.71* | 0.41*  | 0.15        | NG    | 2                                 |
| Nickel   | 42.8 | 51.6* | > 140  | NG          | NG    | 1                                 |
| Silver   | 1.77 | 3.7*  | > 0.56 | NG          | NG    | 1                                 |
| Zinc   | 271  | 410*  | 410*   | 560*        | NG    | 3                                 |
| <b>Polycyclic Aromatic Hydrocarbons (PAHs; SQAGs in µg/kg)</b> |      |       |        |             |       |                                   |
| Acenaphthene   | 88.9 | 500   | 500    | NG          | NG    | 0                                 |
| Acenaphthylene   | 128  | 640   | > 560  | NG          | 47.4* | 1                                 |
| Anthracene   | 245  | 1100  | 960    | NG          | 163*  | 1                                 |
| Fluorene   | 144  | 540   | 540    | NG          | 101*  | 1                                 |
| Naphthalene  | 391  | 2100  | 2100   | 10500       | 414*  | 1                                 |
| 2-Methylnaphthalene  | 201  | 670   | 670    | NG          | NG    | 0                                 |
| Phenanthrene   | 544  | 1500* | 1500*  | 14000       | 368   | 2                                 |
| Total Low Molecular Weight PAHs                                | 1442 | 3160* | 5200   | NG          | NG    | 1                                 |
| Benz(a)anthracene  | 693  | 1600* | 1300*  | 55000       | 261   | 2                                 |
| Benzo(a)pyrene   | 763  | 1600* | 1600*  | 450000      | 397   | 2                                 |
| Chrysene   | 846  | 2800  | 1400*  | 115000      | 384   | 1                                 |
| Dibenz(a,h)anthracene  | 135  | 260*  | 230*   | NG          | NG    | 2                                 |

**Table 6. A comparison of sediment quality guidelines (SQAGs) applicable to coastal and marine waters: Probable effect level (PEL) equivalents (continued).**

| Substance   | PEL   | ER-M   | LAET   | SQC - Acute | SLC  | Number of SQAGs Comparable to PEL |
|---|-------|--------|--------|-------------|------|-----------------------------------|
| <b>Polycyclic Aromatic Hydrocarbons (PAHs; SQAGs in µg/kg; cont.)</b> |       |        |        |             |      |                                   |
| Fluoranthene  | 1494  | 5100   | 1700*  | 9000        | 644* | 2                                 |
| Pyrene  | 1398  | 2600*  | 2600*  | 49500       | 665  | 2                                 |
| Total High Molecular Weight PAHs                                      | 6676  | 9600*  | 12000* | NG          | NG   | 2                                 |
| Total PAHs  | 16770 | 44792* | NG     | NG          | NG   | 1                                 |
| <b>Polychlorinated Biphenyls (PCBs; SQAGs in µg/kg)</b>               |       |        |        |             |      |                                   |
| Total PCBs  | 189   | 180*   | 130*   | NG          | 36.6 | 2                                 |
| <b>Pesticides (SQAGs in µg/kg)</b>                                    |       |        |        |             |      |                                   |
| Chlordane   | 4.79  | 6*     | NG     | NG          | NG   | 1                                 |
| Dieldrin  | 4.3   | 8*     | NG     | NG          | NG   | 1                                 |
| p,p'-DDD  | 7.81  | 20*    | 16*    | NG          | NG   | 2                                 |
| p,p'-DDE  | 374   | 27     | 9      | NG          | NG   | 0                                 |
| p,p'-DDT  | 4.77  | 7*     | 34     | NG          | NG   | 1                                 |
| Total DDT   | 51.7  | 46.1*  | NG     | 210         | 505  | 1                                 |
| Lindane (gamma-BHC)   | 0.99  | NG     | NG     | NG          | NG   | 0                                 |
| <b>Phthalates (SQAGs in µg/kg)</b>                                    |       |        |        |             |      |                                   |
| Bis(2-ethylhexyl)phthalate  | 2647  | NG     | 1900*  | NG          | NG   | 1                                 |

ER-M = Effects range median (Long and Morgan 1990).

LAET = Lowest apparent effects threshold (PTI 1988).

SQC - Acute = Acute sediment quality criterion (EqPA; Lyman et al. 1987; Pavlou et al. 1987).

SLC = National screening level concentration (Neff et al. 1987).

\* = Indicates that the SQGs are within a factor of three of the TEL.

NG = No guideline available.

- ▶ The screening levels (PSDDA-SL) developed for use in Washington State under the Puget Sound Dredged Disposal Analysis Program (USACOE 1988);
- ▶ The USEPA chronic sediment quality criteria (SQC - Chronic) developed using the equilibrium partitioning approach (assuming 1% TOC; Lyman *et al.* 1987; Pavlou *et al.* 1987; Hansen *et al.* 1993a, b, c, d, and e); and,
- ▶ The sediment quality objectives (SQOs) developed for Burrard Inlet, British Columbia (Swain and Nijman 1991). These objectives were derived using a variation of the sediment background approach and are intended to protect the most sensitive use of bed sediments.

Probable effect levels define the lower limit of the range of contaminant concentrations that has a high probability of being associated with adverse biological effects. Hence, the PELs are considered to provide a low level of protection for aquatic organisms. The four sets of guidelines identified for comparison with the PELs included:

- ▶ The effects range median (ER-M) values promulgated under the NOAA National Status and Trends Program (Long and Morgan 1990);
- ▶ The lowest apparent effects threshold values (LAET) calculated for Puget Sound (PTI 1988);
- ▶ The acute sediment quality criteria (SQC - Acute) developed using the equilibrium partitioning approach (assuming 1% TOC; Lyman *et al.* 1987; Pavlou *et al.* 1987); and,
- ▶ The national screening level concentrations developed using the screening level concentration (SLC) approach (Neff *et al.* 1987).

While it is difficult to define an acceptable level of comparability for the SQAGs, a panel of experts assembled in California recommended that agreement within a factor of three or less among various guidelines was indicative of good concordance (Lorenzato *et al.* 1991). Using this guidance, the comparability of the SQAGs was evaluated by comparing them with those that have been developed using other approaches and other procedures. Comparable guidelines (i.e., agreeing within a factor of three of the SQAGs) were designated with an asterisk (\*) in Tables 5 and 6.

Evaluation of the comparability of the SQAGs was impaired by the lack of guidelines for certain substances. For example, four or more guidelines were available for only 19 and 18 of the substances for which TELs and PELs, respectively, have been developed. An adequate

number of guidelines were not available for chromium, nickel, silver, bis(2-ethylhexyl)phthalate, several PAHs, and most of the pesticides. Nonetheless, the results of this evaluation indicate that many of the SQAGs are comparable to the guidelines that have been derived for other applications and, hence, provide a precise basis for evaluating sediment quality conditions.

The recommended TELs for 17 of the 34 substances were comparable to two or more other guidelines, which were intended to provide a high level of protection for aquatic biota (Table 5). The best agreement was observed for metals, while the poorest agreement was observed for high molecular weight PAHs. This evaluation revealed that the TELs were usually lower than the other guidelines with which they were compared.

The recommended PELs for 14 of the 34 substances were comparable to two or more other guidelines (Table 6). Once again, the best agreement among the various guidelines was observed for metals. Relatively poor agreement was observed among the guidelines for pesticides and the low molecular weight PAHs. As was the case for the TELs, the PELs were generally lower than the other guidelines that were derived using different approaches and different procedures.

While agreement between guideline values provides users with additional confidence in these assessment tools, comparability does not guarantee that the guidelines will accurately identify impacted and unimpacted sites. Therefore, the reliability and predictability of the guidelines were also evaluated to determine if they can be used to identify impacted and unimpacted sites in the field.

## 7.2 Reliability of the Sediment Quality Assessment Guidelines

An evaluation of the reliability of the SQAGs was conducted to provide practitioners with an indication of the level of confidence that should be placed in these management tools. The recommended SQAGs were derived by applying a consistent arithmetic procedure to the information contained in the expanded NSTP database. This procedure was designed to support the derivation of two guideline values for each substance, which could be used together to define three ranges of contaminant concentrations: the minimal effects range; the possible effects range; and, the probable effects range. A high degree of confidence in the guidelines would be indicated if a low, moderate, and high incidence of adverse biological effects were actually observed within these three ranges of concentrations, respectively. In addition, confidence in the SQAGs would be warranted if the incidence of effects increased consistently and markedly with increasing chemical concentrations (Long *et al.* In press).

In this section, the reliability of the SQAGs was evaluated using information in the expanded NSTP database. To conduct this evaluation, a scoring system was devised to integrate the following information:

- (i) The incidence of adverse biological effects within the minimal effects range;
- (ii) The incidence of adverse biological effects within the probable effects range; and,
- (iii) The degree of concordance between the concentrations of sediment-associated contaminants and the incidence of adverse biological effects.

The first step in applying this scoring system was to determine the TEL Score (TS). This parameter quantifies the extent to which the TEL fulfills the objective of defining the upper limit of a range of chemical concentrations within which adverse biological effects occurred only infrequently. The key metric in this evaluation was the incidence of adverse biological effects within the minimal effects range, as calculated using the expanded NSTP database. Specifically, the number of "hits" (\*) and total number of records within the minimal effects range were identified on a substance by substance basis. Subsequently, the percent incidence of adverse effects was calculated for each substance by dividing the number of hits by the total number of records in the minimal effects range and multiplying by 100 (Table 7). A TS of two (2), one (1), or zero (0) was assigned if the incidence of adverse biological effects was < 10%, 10 to 25%, or > 25%, respectively, within the minimal effects range (Table 8; Long *et al.* In press).

Next, PEL Scores (PS) were determined for each substance and group of substances for which SQAGs were derived. This second parameter assesses the extent to which the PEL fulfills the objective of defining the lower limit of a range of chemical concentrations within which adverse biological effects usually or always occurred. Consistent with the procedures used to determine the TS, the percent incidence of adverse biological effects within the probable effects range was calculated on a substance by substance basis (i.e., by dividing the number of hits by the total number of records in the probable effects range and multiplying by 100). A PS of two (2), one (1), or zero (0) was assigned if the incidence of adverse biological effects was > 65%, 50 to 65%, or < 50%, respectively, within the probable effects range.

A Concordance Score (CS) was then determined to assess the agreement between contaminant concentrations and the incidence of adverse biological effects. The first step in this process was to calculate the incidence of adverse biological effects within the possible effects range (i.e., between the TEL and PEL). Next, the percent incidence of adverse biological effects within each of the three ranges of contaminant concentrations were compared. As there should be a consistent and marked increase in the incidence of hits

**Table 7. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments.**

| Substance                                      | % 'Hits' in the Minimum Effects Range (< =TEL) | % 'Hits' in the Possible Effects Range (>TEL to <PEL) | % 'Hits' in the Probable Effects Range (>= PEL) |
|--|--|---|---|
| <b>Metals</b>                                  |  |   |   |
| Arsenic  | 2.7  | 12.9  | 46.8  |
| Cadmium  | 5.6  | 20.1  | 70.8  |
| Chromium                                       | 3.5  | 15.4  | 52.9  |
| Copper   | 9.0  | 21.9  | 55.9  |
| Lead   | 5.8  | 25.8  | 58.4  |
| Mercury  | 7.8  | 23.6  | 36.7  |
| Nickel   | 3.3  | 8.4   | 9.4   |
| Silver   | 6.6  | 9.8   | 60.5  |
| Zinc   | 3.8  | 27.2  | 64.8  |
| <b>Polycyclic Aromatic Hydrocarbons (PAHs)</b> |  |   |   |
| Acenaphthene                                   | 7.5  | 29.1  | 57.4  |
| Acenaphthylene                                 | 7.4  | 13.9  | 51.4  |
| Anthracene                                     | 8.7  | 20.5  | 75.0  |
| Fluorene                                       | 11.7   | 20.5  | 70.0  |
| 2-methylnaphthalene                            | 0.0  | 23.4  | 81.5  |
| Naphthalene                                    | 2.6  | 19.3  | 71.2  |
| Phenanthrene                                   | 8.0  | 22.8  | 77.8  |
| Sum LMW-PAHs                                   | 8.7  | 19.4  | 65.6  |
| Benz(a)anthracene                              | 8.7  | 15.7  | 78.4  |
| Benzo(a)pyrene                                 | 8.5  | 22.1  | 70.9  |
| Chrysene                                       | 9.2  | 18.8  | 72.4  |
| Dibenzo(a,h)anthracene                         | 15.8   | 11.6  | 65.1  |
| Fluoranthene                                   | 9.5  | 20.2  | 79.7  |
| Pyrene   | 7.4  | 19.3  | 83.0  |
| Sum HMW-PAHs                                   | 9.5  | 15.0  | 65.5  |
| Total PAHs                                     | 7.3  | 19.3  | 76.7  |
| <b>Polychlorinated Biphenyls (PCBs)</b>        |  |   |   |
| Total PCBs                                     | 15.7   | 36.9  | 54.9  |

**Table 7. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments (continued).**

| Substance               | % 'Hits' in the Minima Effects Range<br>(< =TEL) | % 'Hits' in the Possible Effects Range<br>(>TEL to <PEL) | % 'Hits' in the Probable Effects Range<br>(>= PEL) |
|-------------------------|--|--|--|
| <b>Pesticides</b>       |  |  |  |
| Chlordane               | 9.0  | 12.1   | 17.0   |
| Dieldrin                | 3.5  | 13.2   | 50.0   |
| Lindane                 | 2.9  | 21.1   | 25.6   |
| p,p'-DDD                | 3.6  | 10.9   | 46.2   |
| p,p'-DDE                | 5.3  | 16.5   | 50.0   |
| p,p'-DDT                | 7.9  | 4.8  | 58.6   |
| Total DDT               | 47.6   | 25.6   | 64.0   |
| <b>Phthalates</b>       |  |  |  |
| Bis(2-ethylhexyl)phthal | 8.5  | 21.2   | 66.7   |

% 'Hits' = Number of effects entries/total number of data entries within each range.

TEL = Threshold effect level.

PEL = Probable effect level.

H = High; M = Moderate; L = Low.

Confidence in the TEL 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.

**Table 8. An evaluation of the reliability of the sediment quality assessment guidelines for Florida coastal waters.**

| Substance                                      | TEL<br>Score<br>(TS) | PEL<br>Score<br>(PS) | Concordance<br>Score<br>(CS) | Total Reliability<br>Score<br>(TRS) | Degree of<br>Reliability |
|--|----------------------|----------------------|------------------------------|-------------------------------------|--------------------------|
| <b>Metals</b>                                  |                      |                      |                              |                                     |                          |
| Arsenic  | 2                    | 0                    | 2                            | 4                                   | M                        |
| Cadmium  | 2                    | 2                    | 2                            | 6                                   | H                        |
| Chromium                                       | 2                    | 1                    | 2                            | 5                                   | M                        |
| Copper   | 2                    | 1                    | 2                            | 5                                   | M                        |
| Lead   | 2                    | 1                    | 2                            | 5                                   | M                        |
| Mercury  | 2                    | 0                    | 1                            | 3                                   | L                        |
| Nickel   | 2                    | 0                    | 1                            | 3                                   | L                        |
| Silver   | 2                    | 1                    | 1                            | 4                                   | M                        |
| Zinc   | 2                    | 1                    | 2                            | 5                                   | M                        |
| <b>Polycyclic Aromatic Hydrocarbons (PAHs)</b> |                      |                      |                              |                                     |                          |
| Acenaphthene                                   | 2                    | 1                    | 1                            | 4                                   | M                        |
| Acenaphthylene                                 | 2                    | 1                    | 1                            | 4                                   | M                        |
| Anthracene                                     | 2                    | 2                    | 2                            | 6                                   | H                        |
| Fluorene                                       | 1                    | 2                    | 1                            | 4                                   | M                        |
| Naphthalene                                    | 2                    | 2                    | 2                            | 6                                   | H                        |
| 2-Methylnaphthalene                            | 2                    | 2                    | 2                            | 6                                   | H                        |
| Phenanthrene                                   | 2                    | 2                    | 2                            | 6                                   | H                        |
| Total Low Molecular Weight PAH                 | 2                    | 2                    | 2                            | 6                                   | H                        |
| Benz(a)anthracene                              | 2                    | 2                    | 1                            | 5                                   | M                        |
| Benzo(a)pyrene                                 | 2                    | 2                    | 2                            | 6                                   | H                        |
| Chrysene                                       | 2                    | 2                    | 2                            | 6                                   | H                        |
| Dibenz(a,h)anthracene                          | 1                    | 2                    | 1                            | 4                                   | M                        |
| Fluoranthene                                   | 2                    | 2                    | 2                            | 6                                   | H                        |
| Pyrene   | 2                    | 2                    | 2                            | 6                                   | H                        |
| Total High Molecular Weight PAH                | 2                    | 2                    | 1                            | 5                                   | M                        |
| Total PAHs                                     | 2                    | 2                    | 2                            | 6                                   | H                        |

**Table 8. An evaluation of the reliability of the sediment quality assessment guidelines for Florida coastal waters (continued).**

| Substance                               | TEL<br>Score<br>(TS) | PEL<br>Score<br>(PS) | Concordance<br>Score<br>(CS) | Total Reliability<br>Score<br>(TRS) | Degree of<br>Reliability |
|---|----------------------|----------------------|------------------------------|-------------------------------------|--------------------------|
| <b>Polychlorinated Biphenyls (PCBs)</b> |                      |                      |                              |                                     |                          |
| Total PCBs                              | 1                    | 1                    | 1                            | 3                                   | L                        |
| <b>Pesticides</b>                       |                      |                      |                              |                                     |                          |
| Chlordane                               | 2                    | 0                    | 0                            | 2                                   | L                        |
| Dieldrin                                | 2                    | 1                    | 2                            | 5                                   | M                        |
| p,p'-DDD                                | 2                    | 0                    | 2                            | 4                                   | M                        |
| p,p'-DDE                                | 2                    | 1                    | 2                            | 5                                   | M                        |
| p,p'-DDT                                | 2                    | 1                    | 1                            | 4                                   | M                        |
| Total DDT                               | 0                    | 1                    | 0                            | 1                                   | L                        |
| Lindane (gamma-BHC)                     | 2                    | 0                    | 1                            | 3                                   | L                        |
| <b>Phthalates</b>                       |                      |                      |                              |                                     |                          |
| Bis(2-ethylhexyl)phthalate              | 2                    | 2                    | 2                            | 6                                   | H                        |

H = High (TRS = 6); M = Moderate (TRS = 4-5); L = Low (TRS <4).

TEL = Threshold effect level; PEL = Probable effect level.

within the three concentration ranges (Long *et al.* In press), the presence of at least a two fold increase in the incidence of hits between adjacent ranges of concentrations was used as an indicator of concordance between increasing chemical concentrations and increased incidence of adverse biological effects. A CS of two (2) was assigned if the percent incidence of adverse biological effects was a factor of two or more higher in the possible effects range compared to the minimal effects range *and* a factor of two or more higher in the probable effects range compared to the possible effects range. Lower CSs were assigned if this criterion was not met. Thus, a CS of 1 was assigned if the factor of two difference was apparent between only two of the ranges, while a CS of 0 was assigned if there was no apparent concordance.

Finally, the overall reliability of the SQAGs for each substance and group of substances was evaluated by calculating a Total Reliability Score (TRS). The TRS was determined by calculating the sum of the TEL, PEL, and Concordance Scores. The SQAGs were considered to have a high degree of reliability if they met all of the evaluation criteria identified previously (i.e., TRS = 6). A moderate degree of reliability was considered to exist when intermediate scores were obtained for one or two of the parameters, or a low score was obtained for one parameter but high scores were assigned for the other two parameters resulting in a TRS of four (4) or five (5). A lower degree of reliability was assigned to SQAGs when TRSs of less than four (4) were calculated.

The results of this evaluation indicate that, in general, the recommended SQAGs provide reliable tools for evaluating sediment quality in Florida coastal waters. The TELs for a total of 30 substances were considered to have a high degree of reliability (TS = 2), as indicated by the low incidence of effects within the minimal effects range. Nine trace metals, 14 individual PAHs or groups of PAHs, six pesticides, and bis(2-ethylhexyl)phthalate were included in this group (Table 8). Moderate reliability (TS = 1) was indicated for fluorene, dibenzo(a,h)anthracene, and total PCBs. A low degree of confidence was placed on the TEL for only one substance, total DDT (TS = 0).

In general, the reliability of the PELs was lower than that for the TELs. The PELs for 16 substances were considered to have a high degree of reliability (PS = 2), as indicated by a high incidence of adverse biological effects within the probable effects ranges. Of the highly reliable PELs, 14 were for individual PAHs or groups of PAHs (Table 8). The PELs for cadmium and bis(2-ethylhexyl)phthalate were also considered to be highly reliable. A moderate degree of reliability (PS = 1) was indicated for the PELs for most of the trace metals (5 of 9); while a low degree of reliability (PS = 0) was apparent for three metals (arsenic, mercury, and nickel). The PELs for pesticides and total PCBs were considered to have either a moderate or low level of reliability.

A high degree of concordance between contaminant concentrations and the incidence of adverse biological effects was observed for the majority of the recommended SQAGs (Tables 6 and 7). The incidence of adverse effects increased consistently and markedly with

increasing concentrations of all trace metals, except mercury, nickel, and silver. Two-fold increases in the incidence of effects between the minimal and possible effects ranges, *and* the possible and probable effects ranges were also observed for ten of the 16 individual PAHs and groups of PAHs. The concordance scores for dieldrin, p,p-DDD, and p,p-DDE were also high (CS = 2), while those for four other pesticides and total PCBs were lower (CS = 0 or 1).

As indicated previously, the overall reliability of the SQAGs for each substance and group of substances was determined using all three measures of reliability. The results of this evaluation indicate that the majority of the SQAGs are highly or moderately reliable ( $TRS \geq 4$ ) and can be used with confidence to conduct sediment quality assessments (Table 8). A high degree of reliability ( $TRS = 6$ ) is indicated for the guidelines for one trace metal (cadmium), ten individual PAHs or groups of PAHs, and bis(2-ethylhexyl)phthalate. The SQAGs for 16 other substances were considered to be moderately reliable ( $TRS = 4$  or 5), including those for six trace metals, five individual PAHs, total HMW-PAHs, dieldrin, p,p-DDE, p,p-DDD, and p,p-DDT. The guidelines for mercury, nickel, total PCBs, chlordane, lindane, and total DDT were all considered to have a low level of reliability ( $TRS \leq 3$ ). It is important to consider the results of this evaluation when applying the SQAGs.

### 7.3 Predictability of the Sediment Quality Assessment Guidelines

The SQAGs are intended to provide effective tools for screening sediment quality data to identify priorities for further actions. For this objective to be realized, the SQAGs must support accurate predictions regarding the biological effects of sediment-associated contaminants in the field. Therefore, an initial evaluation is provided in this section to determine their ability to correctly identify biologically impacted and non-impacted sites. As the applicability of the SQAGs to Florida coastal waters and other areas in the southeastern portion of the United States is the focus of this evaluation, the predictability of the guidelines was assessed using the results of field studies from Florida (Tampa Bay and Pensacola Bay) and nearby areas (Gulf of Mexico). However, data from field studies conducted in California (San Pedro Bay) and in New York (Hudson-Raritan Estuary) were also used to evaluate the predictability of the SQAGs in more highly contaminated areas. It should be noted that these independent data sets were not used to derive the SQAGs (i.e., they are independent data sets).

The predictability of the TELs and PELs was determined separately for each of the data sets considered in this evaluation. The first step in the evaluation process was to assemble the sediment chemistry data for each geographic area. Next, the concentrations of each chemical in each sediment sample were compared to the SQAGs. These comparisons formed the basis of the predictions that were made regarding the toxicity of each sediment sample. Sediment samples with concentrations of one or more substances that exceeded their respective PELs

were predicted to be toxic. Sediment samples were predicted to be non-toxic if the concentrations of all measured substances were below the respective TELs that had been recommended for those substances. Samples with concentrations of one or more substances greater than the TELs and no exceedances of the PELs were considered to be possibly toxic.

These samples were not considered in the evaluation of predictability; nonetheless, the incidence of toxicity was calculated for these samples to provide an indication of the potential for observing toxic effects when contaminant concentrations fall within the possible effects range.

The predictability of the SQAGs was evaluated by comparing the predictions with the results of the biological investigations. The predictability of the PELs and the TELs were calculated as the ratio of the number of stations that were observed to be impacted and the total number of stations that were predicted to be impacted (expressed as a percentage). In this assessment, toxic samples were defined as those in which one or more of the measured bioassay endpoints were significantly different from those in control/reference samples.

During 1991 and 1992, a total of 144 sediment samples were collected to assess the toxicity of Tampa Bay sediments (Long *et al.* 1994). A suite of bioassays was used in this assessment, including a 10-d amphipod (*Ampelisca abdita*) survival test, a 1-hr sea urchin (*Arbacia punctulata*) fertilization test (using porewater), and a Microtox test (using organic extracts). Matching sediment chemistry data, including metals, PAHs, PCBs, and a suite of pesticides, were collected on 61 of these samples. Based on a comparison of the sediment chemistry data with the PELs, a total of 46 samples were predicted to be toxic (Table 9). Of these, 40 samples were observed to be toxic. Therefore, the predictability of the PELs was roughly 87%. Only two sites were predicted to be not toxic, based on comparisons of the sediment chemistry data with the TELs. The results of the biological tests conducted on these samples revealed that neither sample was toxic. Hence, the predictability of the TELs was calculated to be 100% in Tampa Bay.

In 1993, a total of 40 samples were collected to assess sediment quality in Pensacola Bay (Long Unpublished data). Sediment chemistry data were obtained on 20 of these samples, with the analytes including metals, PAHs, and a suite of pesticides. In this study, toxicity was assessed using a 10-d amphipod (*Ampelisca abdita*) survival test, a 48-hr sea urchin (*Arbacia punctulata*) fertilization and embryo development test (using porewater), and Microtox (using organic extracts). Based on comparisons of the chemistry data with the PELs, a total of 12 samples were predicted to be toxic (Table 10). Of these, 11 of the samples were observed to be toxic. Therefore, the predictability of the PELs was 92%. Using the TELs, two samples were predicted to be not toxic; both of these were observed to be toxic. Hence, the predictability of the TELs was calculated to be 0%.

As part of the USEPA Environmental Monitoring and Assessment Program (EMAP 1991), matching sediment chemistry and biological effects data were collected from eight areas in the Gulf of Mexico region in 1991. The areas sampled in this survey included Galveston Bay (TX), Matagorda Bay (TX), Mississippi River (LA), Mississippi Sound (LA), Mobile Bay

**Table 9. Predicted toxicity vs. observed toxicity in Tampa Bay sediments (1991 and 1992 survey; Long et al. 1994).**

| Predicted Toxicity              | Number of<br>Toxic Samples<br>Amphipod Test (%) | Number of<br>Toxic Samples<br>Sea Urchin Test (%) | Number of<br>Toxic Samples<br>Microtox Test (%) | Predictability<br>(Using All Tests;<br>% Correct) |
|---------------------------------|---|---|---|---|
| Overall Toxicity                | 8 of 61 (13%)                                   | 50 of 61 (82%)                                    | 7 of 16 (44%)                                   | -   |
| Not Toxic (< TELs)              | 0 of 2 (0%)                                     | 0 of 2 (0%)                                       | 0 of 2 (0%)                                     | 100% (2 of 2)                                     |
| Possibly Toxic (> TELs; < PELs) | 1 of 13 (8%)                                    | 10 of 13 (77%)                                    | 1 of 5 (20%)                                    | -   |
| Toxic (1 PEL exceeded)          | 0 of 22 (0%)                                    | 16 of 22 (73%)                                    | 1 of 2 (50%)                                    | 73% (16 of 22)                                    |
| Toxic (2-5 PELs exceeded)       | 2 of 3 (67%)                                    | 3 of 3 (100%)                                     | 0 of 2 (0%)                                     | 100% (3 of 3)                                     |
| Toxic (6-9 PELs exceeded)       | 1 of 5 (20%)                                    | 5 of 5 (100%)                                     | 1 of 1 (100%)                                   | 100% (5 of 5)                                     |
| Toxic (> 10 PELs exceeded)      | 4 of 16 (25%)                                   | 16 of 16 (100%)                                   | 4 of 4 (100%)                                   | 100% (16 of 16)                                   |
| Toxic (1 or more PELs exceeded) | 7 of 46 (15%)                                   | 40 of 46 (87%)                                    | 6 of 9 (66%)                                    | 87% (40 of 46)                                    |
| Toxic (2 or more PELs exceeded) | 7 of 24 (29%)                                   | 24 of 24 (100%)                                   | 5 of 7 (71%)                                    | 100% (24 of 24)                                   |

TEL = Threshold effect level; PEL = Probable effect level.

**Table 10. Predicted toxicity vs. observed toxicity in Pensacola Bay sediments (Long Unpublished data).**

| Predicted Toxicity              | Number of Toxic Samples<br>Amphipod Test (%) | Number of Toxic Samples<br>Sea Urchin Test (%) | Number of Toxic Samples<br>Microtox Test (%) | Predictability<br>(Using All Tests;<br>% Correct) |
|---------------------------------|--|--|--|---|
| Overall Toxicity                | 0 of 20 (0%)                                 | 13 of 20 (65%)                                 | 15 of 20 (75%)                               | -   |
| Not Toxic (< TELs)              | 0 of 2 (0%)                                  | 1 of 2 (50%)                                   | 2 of 2 (100%)                                | 0% (0 of 2)                                       |
| Possibly Toxic (> TELs; < PELs) | 0 of 6 (0%)                                  | 3 of 6 (50%)                                   | 2 of 6 (33%)                                 | -   |
| Toxic (1 PEL exceeded)          | 0 of 2 (0%)                                  | 2 of 2 (100%)                                  | 2 of 2 (100%)                                | 100% (2 of 2)                                     |
| Toxic (2-5 PELs exceeded)       | 0 of 5 (0%)                                  | 3 of 5 (60%)                                   | 4 of 5 (80%)                                 | 80% (4 of 5)                                      |
| Toxic (6-9 PELs exceeded)       | 0 of 2 (0%)                                  | 2 of 2 (100%)                                  | 2 of 2 (100%)                                | 100% (2 of 2)                                     |
| Toxic (> 10 PELs exceeded)      | 0 of 3 (0%)                                  | 2 of 3 (67%)                                   | 3 of 3 (100%)                                | 100% (3 of 3)                                     |
| Toxic (1 or more PELs exceeded) | 0 of 12 (0%)                                 | 9 of 12 (75%)                                  | 11 of 12 (92%)                               | 92% (11 of 12)                                    |
| Toxic (2 or more PELs exceeded) | 0 of 10 (0%)                                 | 7 of 10 (70%)                                  | 9 of 10 (90%)                                | 90% (9 of 10)                                     |

TEL = Threshold effect level; PEL = Probable effect level.

(AL), Pensacola Bay, Florida Panhandle, and West Central Florida. Sediment chemistry data were collected on metals (including organotins), PAHs, PCBs, a suite of pesticides, and several additional substances. Sediment toxicity was assessed using acute toxicity tests on two crustacean species, including the amphipod, *Ampelisca abdita*, and the mysid shrimp, *Mysidopsis bahia*. As no statistical evaluation of the toxicity test results were reported by the investigators, a 20% difference between the survival of test organisms in Gulf of Mexico sediments and control sediments was considered to be indicative of toxicity. Of the 47 samples collected in this survey, three were predicted to be toxic and 16 were predicted to be not toxic (Table 11). The results of the two bioassays indicated that none of the samples that were predicted to be toxic were observed to be toxic (predictability = 0%). In contrast, 15 of the samples that were predicted to be not toxic were observed to be not toxic (predictability = 94%).

Although additional evaluations of the predictability of the SQAGs are needed, they appear to provide predictive tools for evaluating sediment quality in the southeastern portion of the United States. Considering the field survey results from the Gulf of Mexico region, 51 of the 61 sites that were predicted to be toxic actually were toxic to one or more species (Table 12). Hence, an overall predictability of roughly 84% was calculated for the PELs. The PELs were more predictive of toxicity when two or more of the SQAGs were exceeded in bed sediments (predictability = 97%; Table 12). As might be expected, these data suggest that the likelihood of observing toxic effects increases with the number of chemicals that are present at concentrations that fall within the probable effects range. By comparison, 17 of the 20 samples that were predicted to be not toxic actually were not toxic to the organisms tested. The predictability of the TELs was, therefore, calculated to be 85%. However, nearly half of the samples collected in these surveys had concentrations of one or more contaminants that fell within the possible effects range; it was not possible to predict whether or not these samples would be toxic. It should be noted that roughly 40% of the samples with contaminant concentrations within the possible effects range were observed to be toxic, based on the results of several tests.

While the recommended SQAGs were developed to assess sediment quality in Florida coastal waters, it has become apparent that these guidelines may be used in broader coastal applications throughout the United States. Data from other regions are important because they provide a means of evaluating the predictability of the SQAGs in more contaminated areas and in other sediment types. Therefore, the predictability of the SQAGs was evaluated using matching sediment chemistry and biological effects data from two additional locations, including San Pedro Bay (CA) and the Hudson-Raritan Estuary (NY). In the San Pedro Bay survey (Long Unpublished data), sediment samples were collected from 44 sites, with toxicity evaluated using a 10-d amphipod (*Eohaustorius estuaricus*) survival test and a 48-hr abalone (*Haliotis rufescens*) embryo development test (using porewater). Using the PELs, 36 of these samples were predicted to be toxic (Table 13). Thirty-four of the 36 samples were observed to be toxic in one or more assays, resulting in a predictability of 94%. Based on comparisons of the sediment chemistry data to the TELs, none of the remaining samples had

**Table 11. Predicted toxicity vs. observed toxicity in Gulf of Mexico sediments (EMAP 1991).**

| Predicted Toxicity              | Number of<br>Toxic Samples<br>Amphipod Test (%) | Number of<br>Toxic Samples<br>Mysid Test (%) | Predictability<br>(Using Both Tests;<br>% Correct) |
|---------------------------------|---|--|--|
| Overall Toxicity                | 7 of 47 (15%)                                   | 0 of 47 (0%)                                 | -  |
| Not Toxic (< TELs)              | 1 of 16 (6%)                                    | 0 of 16 (0%)                                 | 94% (15 of 16)                                     |
| Possibly Toxic (> TELs; < PELs) | 6 of 28 (21%)                                   | 0 of 28 (0%)                                 | -  |
| Toxic (1 PEL exceeded)          | 0 of 3 (0%)                                     | 0 of 3 (0%)                                  | 0% (0 of 3)  |
| Toxic (2-5 PELs exceeded)       | -   | -  | -  |
| Toxic (6-9 PELs exceeded)       | -   | -  | -  |
| Toxic (> 10 PELs exceeded)      | -   | -  | -  |
| Toxic (1 or more PELs exceeded) | 0 (0%)  | 0 (0%)                                       | 0% (0 of 3)  |
| Toxic (2 or more PELs exceeded) | -   | -  | -  |

TEL = Threshold effect level; PEL = Probable effect level.

**Table 12. Predicted toxicity vs. observed toxicity in Gulf of Mexico sediment quality surveys.**

| Predicted Toxicity              | Number of Toxic Samples<br>Amphipod Test (%) | Number of Toxic Samples<br>Mysid Test (%) | Number of Toxic Samples<br>Sea Urchin Test (%) | Number of Toxic Samples<br>Microtox Test (%) | Predictability<br>(Using All Tests;<br>% Correct) |
|---------------------------------|--|---|--|--|---|
| Overall Toxicity                | 15 of 128 (12%)                              | 0 of 47 (0%)                              | 63 of 81 (78%)                                 | 22 of 36 (61%)                               | -   |
| Not Toxic (< TELs)              | 1 of 20 (5%)                                 | 0 of 16 (0%)                              | 1 of 4 (25%)                                   | 2 of 4 (50%)                                 | 85% (17 of 20)                                    |
| Possibly Toxic (> TELs; < PELs) | 7 of 47 (15%)                                | 0 of 28 (0%)                              | 13 of 19 (68%)                                 | 3 of 11 (27%)                                | -   |
| Toxic (1 PEL exceeded)          | 0 of 27 (0%)                                 | 0 of 3 (0%)                               | 18 of 24 (75%)                                 | 3 of 4 (75%)                                 | 67% (18 of 27)                                    |
| Toxic (2-5 PELs exceeded)       | 2 of 8 (25%)                                 | -   | 7 of 8 (88%)                                   | 4 of 7 (57%)                                 | 88% (7 of 8)                                      |
| Toxic (6-9 PELs exceeded)       | 1 of 7 (14%)                                 | -   | 7 of 7 (100%)                                  | 3 of 3 (100%)                                | 100% (7 of 7)                                     |
| Toxic (> 10 PELs exceeded)      | 4 of 19 (21%)                                | -   | 19 of 19 (100%)                                | 7 of 7 (100%)                                | 100% (19 of 19)                                   |
| Toxic (1 or more PELs exceeded) | 7 of 61 (11%)                                | 0 (0%)                                    | 51 of 58 (88%)                                 | 17 of 21 (81%)                               | 84% (51 of 61)                                    |
| Toxic (2 or more PELs exceeded) | 7 of 34 (21%)                                | -   | 33 of 34 (97%)                                 | 14 of 17 (82%)                               | 97% (33 of 34)                                    |

TEL = Threshold effect level; PEL = Probable effect level.

**Table 13. Predicted toxicity vs. observed toxicity in San Pedro Bay sediments (Long Unpublished data).**

| Predicted Toxicity              | Number of<br>Toxic Samples<br>Amphipod Test (%) | Number of<br>Toxic Samples<br>Abalone Test (%) | Predictability<br>(Using Both Tests;<br>% Correct) |
|---------------------------------|---|--|--|
| Overall Toxicity                | 22 of 44 (50%)                                  | 40 of 44 (91%)                                 | -  |
| Not Toxic (< TELs)              | -   | -  | -  |
| Possibly Toxic (> TELs; < PELs) | 3 of 8 (38%)                                    | 7 of 8 (88%)                                   | -  |
| Toxic (1 PEL exceeded)          | 8 of 18 (44%)                                   | 16 of 18 (89%)                                 | 94% (17 of 18)                                     |
| Toxic (2-5 PELs exceeded)       | 6 of 12 (50%)                                   | 10 of 12 (83%)                                 | 92% (11 of 12)                                     |
| Toxic (6-9 PELs exceeded)       | 4 of 6 (67%)                                    | 6 of 6 (100%)                                  | 100% (6 of 6)                                      |
| Toxic (> 10 PELs exceeded)      | -   | -  | -  |
| Toxic (1 or more PELs exceeded) | 19 of 36 (53%)                                  | 33 of 36 (92%)                                 | 94% (34 of 36)                                     |
| Toxic (2 or more PELs exceeded) | 10 of 18 (56%)                                  | 16 of 18 (89%)                                 | 94% (17 of 18)                                     |

TEL = Threshold effect level; PEL = Probable effect level.

concentrations of contaminants which all fell within the minimal effects range; therefore, it was not possible to evaluate the predictability of the TELs.

In the Hudson-Raritan Estuary, 38 samples were collected in 1992 to evaluate sediment quality conditions (Long Unpublished data). Four biological tests were conducted to assess the toxicity of these samples, including a 10-d amphipod (*Ampelisca abdita*) survival test, 48-hr bivalve (*Mulinea lateralis*) embryo survival and development tests (using elutriates), and a Microtox assay. Of the 34 samples that were predicted to be toxic, 26 samples exhibited toxicity in one or more of the biological tests (Table 14). Hence, the predictability of the PELs was calculated to be 76%. Two samples were predicted to be not toxic and toxicity was not observed in any of the tests conducted with these samples. Therefore, the predictability of the TELs was determined to be 100%. Of the two samples that were considered to be possibly toxic, both samples were observed to be toxic in one or more biological tests.

In addition to providing predictive tools for evaluating sediment quality conditions in Florida, the results of this preliminary evaluation indicate that the recommended SQAGs may be predictive in other areas, as well. Overall, the predictability of the PELs in San Pedro Bay and the Hudson-Raritan Estuary was greater than 85% (60 of 70 samples were correctly classified as toxic). Both of the samples that were predicted to be not toxic were correctly classified. Nine of the 10 of the samples that were classified as possibly toxic were found to be toxic using a battery of tests.

Considering information from all five areas used in this evaluation, the SQAGs appear to provide predictive tools for assessing sediment quality (Table 15). Of the 22 samples that were predicted to be not toxic, 19 were observed to be not toxic in two or more bioassays. The overall predictability of the TELs was, therefore, calculated to be 88%. It is possible that water-borne contaminants (e.g., ammonia, hydrogen sulfide, etc.), unmeasured contaminants in sediments, or other factors were responsible for the observed toxicity in the samples that were incorrectly classified. Of the 131 samples that were predicted to be toxic (i.e., the concentrations of one or more substances exceeded the PELs), 111 were observed to be toxic in one or more biological tests. Therefore, the PELs were determined to have a predictability of roughly 85%. Predictability increased to 89% when the PELs for two or more substances were exceeded in sediment samples. Approximately 74% of the samples collected had contaminant concentrations that fell within either the minimal or probable effects ranges. Therefore, it was possible to classify as toxic or not toxic the majority of the samples collected in these studies.

#### 7.4 Summary

**Table 14. Predicted toxicity vs. observed toxicity in Hudson-Raritan Estuary sediments (Long Unpublished data).**

| Predicted Toxicity              | Number of<br>Toxic Samples<br>Amphipod Test (%) | Number of<br>Toxic Samples<br>Any of 4 Tests (%) | Predictability<br>(Using All Tests;<br>% Correct) |
|---------------------------------|---|--|---|
| Overall Toxicity                | 21 of 38 (55%)                                  | 28 of 38 (74%)                                   | -   |
| Not Toxic (< TELs)              | 0 of 2 (0%)                                     | 0 of 2 (0%)                                      | 100% (2 of 2)                                     |
| Possibly Toxic (> TELs; < PELs) | 2 of 2 (100%)                                   | 2 of 2 (100%)                                    | -   |
| Toxic (1 PEL exceeded)          | 0 of 1 (0%)                                     | 0 of 1 (0%)                                      | 0% (0 of 1)                                       |
| Toxic (2-5 PELs exceeded)       | 7 of 13 (54%)                                   | 10 of 13 (77%)                                   | 77% (10 of 13)                                    |
| Toxic (6-9 PELs exceeded)       | 4 of 11 (36%)                                   | 7 of 11 (64%)                                    | 64% (7 of 11)                                     |
| Toxic (> 10 PELs exceeded)      | 8 of 9 (89%)                                    | 9 of 9 (100%)                                    | 100% (9 of 9)                                     |
| Toxic (1 or more PELs exceeded) | 19 of 34 (56%)                                  | 26 of 34 (76%)                                   | 76% (26 of 34)                                    |
| Toxic (2 or more PELs exceeded) | 19 of 33 (58%)                                  | 26 of 33 (79%)                                   | 79% (26 of 33)                                    |

TEL = Threshold effect level; PEL = Probable effect level.

**Table 15. Overall predictability of the recommended sediment quality assessment guidelines (SQAGs; using observed toxicity in any bioassay).**

| Location                    | Predictability of the SQAGs (% Correct) |                                   |                                    |                                    |
|-----------------------------|---|-----------------------------------|------------------------------------|------------------------------------|
|                             | Not Toxic<br>(< TELs)                   | Possibly Toxic<br>(>TELs, < PELs) | Toxic<br>(1 or More PELs Exceeded) | Toxic<br>(2 or More PELs Exceeded) |
| Tampa Bay (FL)              | 100% (2 of 2)                           | (10 of 13)                        | 87% (40 of 46)                     | 100% (24 of 24)                    |
| Pensacola Bay (FL)          | 0% (0 of 2)                             | (1 of 2)                          | 92% (11 of 12)                     | 90% (9 of 10)                      |
| Gulf of Mexico              | 94% (15 of 16)                          | (6 of 28)                         | 0% (0 of 3)                        | -                                  |
| San Pedro Bay (CA)          | -                                       | (7 of 8)                          | 94% (34 of 36)                     | 94% (17 of 18)                     |
| Hudson-Raritan Estuary (NY) | 100% ( 2 of 2)                          | (2 of 2)                          | 76% (26 of 34)                     | 79% (26 of 33)                     |
| <b>Overall</b>              | <b>86% (19 of 22)</b>                   | <b>(26 of 53)</b>                 | <b>85% (111 of 131)</b>            | <b>89% (76 of 85)</b>              |

TEL = Threshold effect level; PEL = Probable effect level.

Note: Samples with contaminant concentrations within the possible effects range (i.e., > TEL and < PEL) were not predicted to be toxic or not toxic.

Therefore, the predictability of the SQAGs was not evaluated using these samples.

A preliminary evaluation of the recommended sediment quality assessment guidelines was conducted to determine their potential applicability in Florida coastal waters and other areas in the United States. The results of this evaluation indicate that, in general, the SQAGs can be used with a high or moderate degree of confidence to conduct sediment quality assessments. The SQAGs for 18 to 20 substances were comparable to the functionally-similar guidelines that have been developed using different approaches and/or different data. In addition, the SQAGs for 28 substances had a moderate or high degree of reliability, as indicated by the data contained in the expanded NSTP database. Furthermore, the SQAGs for up to 34 substances, when used collectively, were found to provide predictive tools for classifying marine and estuarine sediments in terms of their potential for being associated with adverse biological effects. Overall, the predictability of the PELs and TELs was 85% and 86%, respectively. Therefore, the results of all three evaluations indicated that the SQAGs are likely to be appropriate for use in a variety of applications in Florida and elsewhere.

While the foregoing evaluation provides important information for assessing the SQAGs, additional data are required to fully evaluate the applicability of these assessment tools. Specifically, matching sediment chemistry and biological effects data from additional areas in Florida are needed to determine the predictability of the SQAGs in various types of sediments and in sediments affected by distinct contaminant sources. For example, the predictability of the SQAGs could be different in carbonate-type sediments and/or sediments that are affected principally by industrial or agricultural contaminant sources (i.e., contaminated with substances for which no SQAGs are currently available). Additional data sets, from sediment quality surveys conducted elsewhere in the United States, are also required to evaluate the regional applicability of the SQAGs.

# **Chapter 8**

## **Summary and Recommendations**

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### **8.1 Summary**

This report recommends and evaluates numerical sediment quality assessment guidelines (SQAGs) for priority substances in Florida coastal waters. An approach was selected that would respond to Florida's management requirements. Following refinement of this approach, numerical SQAGs were derived for 34 substances or groups of substances. These SQAGs were subsequently evaluated to determine their comparability, reliability, and predictability. Volume 2, *Applications of the Sediment Quality Assessment Guidelines*, provides guidance on the intended uses of the SQAGs. Volumes 3 and 4, *Supporting Documentation - Biological Effects Database for Sediments* and *Supporting Documentation - Regional Biological Effects Database for Sediments*, summarize the data that were used to develop the SQAGs.

In Florida, conservation and protection of natural resources is a high priority environmental management goal. Achieving 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 understanding (and public recognition) of the role sediments play in coastal ecosystem functions. Sediments are particularly important in determining the fate and effects of environmental contaminants.

Recent monitoring data indicate that concentrations of various contaminants are elevated at numerous locations in Florida coastal sediments. The SQAGs are needed to evaluate the potential for biological effects associated with these contaminants and to provide assistance in managing coastal resources. In this respect, SQAGs provide useful tools for focusing limited resources on the highest management priorities.

To identify an appropriate procedure for deriving SQAGs, the major approaches used in other jurisdictions to derive numerical SQAGs were evaluated in the context of Florida's requirements for sediment quality assessment values. The results of this analysis indicated that a weight of evidence approach (WEA) used by the NOAA National Status and Trends Program (Long and Morgan 1990) would respond most directly to Florida's requirements. Therefore, a strategy that relied on a refined version of the WEA was recommended to derive numerical SQAGs for near-term use in addressing sediment quality concerns.

Sediment quality assessment guidelines have been developed for 34 priority contaminants and groups of contaminants in Florida coastal waters. However, insufficient data were available to derive guidelines for another 20 substances that are known or are suspected to contaminate Florida coastal sediments. The numerical SQAGs were used to define three ranges of concentrations for each of the 34 contaminants: a probable effects range; a possible effects range; and, a minimal effects range.

A preliminary evaluation of the recommended SQAGs was conducted to determine their potential applicability in Florida coastal waters and other areas in the United States. The results of this evaluation indicate that most of the SQAGs can be used with a high or moderate degree of confidence to conduct sediment quality assessments. The SQAGs for 18 to 20 substances were comparable to similar guidelines developed by other investigators using different approaches and procedures. In addition, the SQAGs for 28 substances had a moderate or high degree of reliability, as indicated by analysis of the incidence of adverse effects in the expanded NSTP database. Furthermore, the SQAGs for 34 substances compared to field-collected data, were found to provide predictive tools for classifying sediments in terms of their potential for being associated with adverse biological effects. Therefore, the results of all three evaluations indicated that the SQAGs are likely to be appropriate for use in a variety of applications in Florida and elsewhere in the United States.

The recommended SQAGs support the identification of contaminated sites and priority chemicals of concern in Florida coastal waters. As such, the guidelines can contribute to the design, implementation, and evaluation of local and state sediment quality monitoring programs. In addition, the guidelines may be used in a variety of environmental management applications, including identifying the need for further testing to support regulatory decisions and evaluating the need for remedial actions. Furthermore, SQAGs provide a common basis for reaching 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. 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.

## 8.2 Recommendations

The following recommendations will strengthen the identification and management of contaminated sediments. While several tasks might appear onerous, most will not require significant resources to complete. Importantly, FDEP has forged partnerships with key investigators in this field, including the National Atmospheric and Oceanic Administration,

National Biological Survey, U.S. Environmental Protection Agency, California State Water Resources Board, and Environment Canada. Activities which are being conducted or planned by these and other agencies will provide products which can be used directly to implement the recommendations. The Department is encouraged to strengthen these relationships and develop a plan of action for addressing sediment contamination.

### **8.2.1 Conduct Further Screening and Expansion of the NSTP Database**

An expanded NSTP database was used to support the derivation of numerical SQAGs for Florida coastal waters. This database should be updated and expanded as new information on the biological effects of sediment-associated contaminants becomes available (see discussion in Section 4.3). Specifically, the database should be expanded to include the results of high quality coastal studies that have recently been conducted in Florida (e.g., Pensacola Bay, St. Andrews Bay), the southeast (e.g., Savannah, GA; Charleston, SC), and elsewhere in the United States (e.g., San Pedro Bay and San Diego Harbor, CA; Boston Harbor, MA). The database should also be improved by further screening the data to identify the highest quality data; additional procedures for screening data were recently proposed by the Science Advisory Group on the Assessment of Sediment Quality.

### **8.2.2 Strengthen and Add Sediment Quality Assessment Guidelines**

An updated and further expanded NSTP database should be used to strengthen the SQAGs recommended in this document and to derive guidelines for additional priority substances (for which insufficient data were available for developing this report) identified in Florida coastal sediments. Numerical SQAGs or refined SQAGs are required for the following substances and groups of substances:

- Metals (specifically, mercury and nickel);
- Organotins;
- Polychlorinated biphenyls (PCBs);
  - Pesticides (specifically, azinphos-methyl, chlordane, chlorothalonil, chlorpyrifos, DDTs, disulfoton, endosulfan, endrin, heptachlor, heptachlor epoxide, endrin, lindane, phorate, toxaphene, and trifluralin);
- Polychlorinated dibenzo-*p*-dioxins (PCDDs);
- Polychlorinated dibenzofurans (PCDFs);
- Pentachlorophenol; and,
- Phthalate esters (specifically, dimethyl phthalate and di-n-butyl phthalate).

### **8.2.3 Further Evaluate the Sediment Quality Assessment Guidelines**

In Chapter 7, the SQAGs were evaluated to determine their applicability in Florida coastal waters. This evaluation focused on the comparability, reliability, and predictability of the SQAGs. However, this evaluation was preliminary and additional data from field surveys and laboratory studies are required to further evaluate the SQAGs.

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 different sediment types. Although the information used to derive the SQAGs includes data from a wide variety of sites in North America, it is possible that these data do not adequately represent the range of conditions in Florida. Further biological testing and benthic invertebrate community evaluations should be conducted in various Florida sediments to determine the applicability of SQAGs. These locations should be selected to encompass a wide range of sediment types and should include contaminated sites and uncontaminated "reference" locations.

The recommended guidelines are based on dry weight-normalized contaminant concentrations. The preliminary evaluation of these SQAGs indicated that they were both reliable and predictive. Nonetheless, the results of several studies suggest that certain variables (such as total organic carbon and acid volatile sulfide) could influence the bioavailability of sediment-associated contaminants. Data from other studies have not confirmed these relationships. Therefore, additional data is needed to define the conditions governing the bioavailability of individual contaminants in Florida coastal sediments. The guidelines should be refined appropriately when these relationships become more clearly established.

### **8.2.4 Determine the Sensitivities of Selected Florida Estuarine Invertebrate Species**

The relative sensitivity of species that occur in Florida is a central consideration for further evaluating the applicability of the SQAGs. The SQAGs recommended for assessing the potential for biological effects of sediment-associated 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 are limited. Therefore, it is difficult to determine if the recommended SQAGs would adequately protect aquatic organisms that occur in Florida coastal waters. Additional biological testing and benthic invertebrate community evaluations should be undertaken to determine if aquatic organisms that occur in Florida have sensitivity ranges similar to organisms occurring in other parts of North America.

### **8.2.5 Develop Bioaccumulation-Based Sediment Quality Assessment Guidelines**

The numerical SQAGs recommended in this study were primarily developed from information on the effects of sediment-associated contaminants on sediment-resident organisms. However, these SQAGs do not consider the potential for bioaccumulation of persistent substances in the tissues of aquatic organisms nor the potential for adverse effects on the human and non-human consumers (wildlife) of these aquatic organisms. Therefore, bioaccumulation-based SQAGs should be developed for the most persistent and bioaccumulative substances that occur in Florida coastal sediments and used with the effects-based SQAGs that were derived in this study. The tissue residue approach may provide appropriate procedures for developing the required SQAGs (see Section 3.4).

### **8.2.6 Develop Sediment Quality Assessment Guidelines for Freshwater Ecosystems**

The SQAGs developed in the present study and the Department's metals interpretive approach are tools for evaluating coastal sediment quality conditions. No such tools exist for use in Florida's freshwater ecosystems. Effects-based SQAGs should be developed to evaluate the biological significance of contaminated sediments in freshwater systems. In the past year, Environment Canada has made progress in developing a biological effects database for freshwater sediments. Upon completion, the information contained in this database could be used to derive numerical SQAGs for freshwater ecosystems. Hence, the FDEP is encouraged to cooperate with this agency to take advantage of the considerable cost-savings and head-start in developing freshwater SQAGs.

In addition to effects-based SQAGs, a companion tool is required to identify metals contamination in freshwater sediments. Therefore, it is recommended that the Department's metals interpretive tool and similar procedures for identifying anthropogenic enrichment of sediment-associated metals be evaluated for application in Florida freshwater sediments.

### **8.2.7 Improve Survey Procedures**

In the past, several sampling, chemical analytical, and biological testing procedures have been used in sediment toxic effects studies (e.g., dredged material disposal, etc.) conducted in Florida coastal waters. Often, the procedures have limited the applicability of the studies and hindered their use in broader ecological investigations. Routine practices should be re-evaluated in light of recent information and procedures refined as appropriate.

Standard procedures for collecting, storing, characterizing, and manipulating sediments that are to be used in toxicological testing have been established by the ASTM (1994) and should be followed. The results of the assessment of the SQAGs indicate that aquatic organisms exhibit a range of sensitivities to sediment-associated contaminants. Therefore, there is a need to ensure that biological investigations of sediment quality employ a battery of tests, with a focus on sensitive species and sensitive endpoints. For example, the results of the sea urchin fertilization test (using porewater) was highly correlated with sediment chemistry in Tampa Bay and Pensacola Bay (Long *et al.* 1994; Long Unpublished data). In contrast, the results of short-term toxicity tests which measure the survival of adult sand worms, bivalves, or shrimp provide little information for assessing sediment quality. Standard methods for conducting biological tests with a range of species (e.g., amphipods and polychaetes; ASTM 1993) have been developed to provide guidance to practitioners in this field and adherence to these protocols will improve the quality of data that are generated in Florida.

### **8.2.8 Increase Coordination with Federal Agencies on Contaminated Sediment Management**

In the southeastern United States, there are a relatively large number of independent and loosely-related initiatives involved in evaluating and managing contaminated sediments. Development of a regional intergovernmental strategy for contaminated sediment assessment and management would improve the effectiveness of state and federal programs, encourage greater local support in preventing sediment contamination and restoring sediment quality, and reduce unnecessary regulatory delays. For example, the FDEP coastal contaminants database is used in a variety of applications, including wetland permitting and restoration, contaminated site clean up, and identifying stormwater management priorities. The consolidation of this data with comparable sediment quality data from other programs, such as the EPA Environmental Monitoring and Assessment Program (EMAP), would be of considerable benefit to both the responsible agencies and the regulated entities.

A cooperative effort should be pursued by the National Oceanic and Atmospheric Administration, National Biological Survey, United States Fish and Wildlife Service, United States Environmental Protection Agency, United States Army Corps of Engineers, United States Geological Survey, Florida Department of Environmental Protection, and other affected agencies to identify priority sediment management, monitoring, and regulatory objectives, and to develop an interagency strategy for achieving them.

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

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### A. *Spiked Sediment Bioassay Data*

1. Toxicity tests which follow published protocols set by the ASTM are acceptable. Other tests which employ more novel protocols **should** be evaluated on a case by case basis (e.g., Green Book test protocols are acceptable).
2. Concentrations of the contaminant in sediment **must** be measured (with the number of measurements taken dependent on the nature of the chemical and duration of the test). Calculated (nominal) concentrations of the substances in sediment are not acceptable.
3. 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.
4. 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.
5. Acceptable biological tests **should** demonstrate that adequate environmental conditions for the test species were maintained throughout the test.
6. Preferred endpoints include effects on embryonic development, early survival, growth, reproduction, and adult survival.
7. Responses and survival of controls **must** be reported and within acceptable limits.
8. Appropriate statistical procedures **should** be used and reported in detail.
9. The equilibrium adjustment period (i.e., time between spiking and initiation of the biological test) and information relevant to the determination if equilibrium had been established **should** be reported.

## Appendix 1: Screening Criteria for Evaluating Candidate Data Sets for the Sediment Toxicity (SEDOX) Database (continued)

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### 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 other published 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.

## Appendix 2: Description of the Procedures for Deriving Sediment Quality Assessment Guidelines.

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An overview of the procedures for deriving numerical sediment quality assessment guidelines (SQAGs) using the information in the expanded NSTP database was included in Chapter 5. Additional information on the procedures is provided here. While these procedures have been automated to generate the SQAGs for this report, they may also be conducted manually with satisfactory results. The manual procedures that can be applied to the data contained in Volume 3 of the report (MacDonald *et al.* 1994) are described below.

The information in the expanded NSTP database is presented in tabular form (i.e., the ascending data tables) in MacDonald *et al.* (1994). These data tables contain information on the concentration of the contaminant (sorted from lowest to highest), geographic area investigated, analysis type, test type, endpoint measured, species tested, life stage, TOC and AVS levels, and bibliographic reference (see Section 5.2). In addition, each entry in the ascending data tables was assigned an 'effects/no-effects' descriptor, which indicates whether or not the contaminant was associated with the biological effect (i.e., endpoint) that was measured. An entry was assigned an 'effects' descriptor (\*) if:

- (i) an adverse biological effect, such as acute toxicity, was reported; and,
- (ii) concordance was apparent between the observed biological response and the measured chemical concentration.

In the co-occurrence analysis of field-collected data entered into the BEDS, an effects descriptor was assigned to data entries in which adverse biological effects were observed in association with at least a two-fold elevation in the chemical concentration above reference concentrations. This two-fold criterion provided a consistent basis for identifying chemical concentrations that were strongly associated with the adverse effect that was measured. Data entries from spiked-sediment bioassays were also assigned an 'effects' descriptor if significant biological effects were reported.

A 'no gradient' (NG) descriptor was assigned when no differences in the concentrations of the chemical of concern was reported between the toxic and non-toxic stations. A 'small gradient' (SG) descriptor was assigned when the concentrations of a substance differed by less than a factor of two between the toxic and non-toxic samples. A 'no concordance' (NC) descriptor was assigned when there was no concordance between the severity of the effect and the chemical concentration (i.e., the concentration of a chemical in the toxic samples was lower than the concentration of that substance in the non-toxic samples). In these cases (i.e., NG, SG, and NC), it was assumed that other factors (whether measured or not) were more important in the etiology of the observed effect than the concentration of the contaminant considered. Finally, a 'no effects' (NE) descriptor was applied to biological data from unaffected, background, reference, or control samples.

## Appendix 2: Description of the Procedures for Deriving Sediment Quality Assessment Guidelines (continued).

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Collectively, the 'effects' data entries from laboratory and field studies were included in the **effects data set (EDS)**. Collectively, data assigned 'no gradient', 'small gradient', 'no concordance', and 'no effects' descriptors were included in the **no effects data set (NEDS)**. These latter data entries were included in the no effects data set because the concentrations of the substance were not associated with the observed biological effects. Hence, the substance was not considered to be harmful at the concentration that was measured. Data entries with greater than values (e.g., > 72 mg/kg for arsenic) were assigned dashes (-) and were not included in either data set.

Derivation of numerical SQAGs from information in the ascending data tables requires several steps. First, the number of data entries in the effects data set is determined by counting all asterisks in the ascending data table for the substance under consideration. For example, examination of Table 1 (arsenic) in MacDonald *et al.* (1994) reveals that the effects data set consists of 38 data entries. Then, the 15th percentile (EDS-L) and the 50th percentile (EDS-M) of the effects data set are determined. For example, the EDS-L and EDS-M for arsenic are 8.2 and 54 mg/kg, respectively. Next, the number of data entries in the no effects data set is established (e.g., there are 257 data entries in the no effects data set for arsenic). The next step in this process necessitates identification of the 50th percentile (NEDS-M) and 85th percentile (NEDS-H) of the no effects data set. For arsenic, the NEDS-M and NEDS-H are 6.4 and 32 mg/kg, respectively. These four values are used directly to calculate the SQAGs.

As indicated in Chapter 5, the TEL is calculated by taking the geometric mean of the 15th percentile concentration in the effects data set (EDS-L) and the 50th percentile concentration in the no effects data set (NEDS-M). The geometric mean is calculated because the effects and no effects data sets are probably not normally-distributed. If these data sets were demonstrated to have normal distributions, then an arithmetic mean could be used to calculate the sediment quality assessment guidelines. The geometric mean of two values is determined by calculating the square root of the product of the two values. This procedure is represented by the following equation:

$$\begin{aligned}
 \text{TEL} &= \sqrt{\text{EDS-L} * \text{NEDS-M}} \\
 &= \sqrt{(8.2 * 6.4)} \\
 &= \sqrt{52.48} \\
 &= 7.24
 \end{aligned}$$

**Appendix 2: Description of the Procedures for Deriving Sediment Quality Assessment Guidelines (continued).**

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Similarly, the PEL is calculated as the geometric mean of the 50th percentile concentration in the effects data set (EDS-M) and the 85th percentile concentration in the no effects data set (NEDS-H). This procedure is represented by the following equation:

$$\begin{aligned} \text{PEL} &= \sqrt[4]{(\text{EDS-M} * \text{NEDS-H})} \\ &= \sqrt[4]{(54 * 32)} \\ &= \sqrt[4]{1728} \\ &= 41.6 \end{aligned}$$