APPENDIX A

BACKGROUND INFORMATION ON SEDIMENTS

APPENDIX A

BACKGROUND INFORMATION ON SEDIMENTS

A.1 ROLE OF SEDIMENTS IN AQUATIC ECOSYSTEMS

The particulate materials that lie below the water in ponds, lakes, streams, rivers, and other aquatic systems are called sediments. Sediments represent essential elements of aquatic ecosystems because they support both autotrophic and heterotrophic organisms. Autotrophic organisms are those that are able to synthesize food from simple inorganic substances (e.g., carbon dioxide, nitrogen, and phosphorus) and the sun's energy. Green plants, such as algae, bryophytes (e.g., mosses and liverworts), and aquatic macrophytes (e.g., sedges, reeds, and pond weed), are the main autotrophic organisms in aquatic ecosystems. In contrast, heterotrophic organisms utilize, transform, and decompose the materials that are synthesized by autotrophic organisms (i.e., by consuming or decomposing autotrophic and other heterotrophic organisms). Some of the important heterotrophic organisms that can be present in aquatic ecosystems include bacteria, epibenthic and infaunal invertebrates, fish, amphibians, and reptiles. Birds and mammals can also represent important heterotrophic components of aquatic food webs (i.e., through the consumption of aquatic organisms).

Sediments support the production of food organisms in several ways. For example, hard bottom sediments, which are characteristic of fast-flowing streams and are comprised largely of gravel, cobbles, and boulders, provide stable substrates to which periphyton (i.e., algae that grow on rocks) can attach. Soft sediments, which are common in ponds, lakes, and slower-flowing sections of rivers, are comprised largely of sand, silt, and clay and provide suitable substrates for aquatic macrophytes. By providing habitats and nutrients for aquatic plants, sediments support autotrophic production (i.e., the primary production of energy) in aquatic systems. Sediments can also support prolific bacterial communities. Bacteria represent important elements of aquatic ecosystems because they assist in the decomposition of organic matter on the surface of the sediment. In so doing, bacteria release nutrients to the water column and increase bacterial biomass. Bacteria represent the primary heterotrophic elements of aquatic ecosystems. The role that sediments play in supporting primary productivity is essential because green plants and bacteria represent the foundation of food webs upon which all other aquatic organisms depend (i.e., they are consumed by many other aquatic species).

In addition to their role in supporting primary productivity, sediments also provide essential habitats for many benthic-dwelling invertebrates and fish. Some of these invertebrate species live on the sediments (i.e., epibenthic species), while others live in the sediments (i.e., infaunal species). Both epibenthic and infaunal invertebrates consume plants, bacteria, and other organisms that are associated with the sediments. Invertebrates represent important elements of aquatic ecosystems because a wide range of wildlife species consumes them. For example, virtually all fish species consume aquatic invertebrates during a portion of, and often throughout, their life cycles. In addition, many birds (e.g., dippers, sand pipers, and swallows) consume either aquatic or emergent invertebrates. Similarly, aquatic invertebrates represent important food sources for both amphibian (e.g., frogs and salamanders) and reptile (e.g., turtles and snakes) species. Therefore, sediment quality is of critical importance to many wildlife species due to the fundamental role that sediments play in the production of aquatic invertebrates.

Importantly, sediments can also provide habitats for many wildlife species during portions of their life cycle. For example, a variety of fish species utilize sediments for spawning and incubation of their eggs and alevins (e.g., trout, salmon, and whitefish). In addition, juvenile fish often find refuge from predators in sediments and in the aquatic vegetation supported by sediments. Furthermore, many amphibian species burrow into sediments in the fall and remain there throughout the winter months. In these instances, sediments serve as important overwintering habitats. Therefore, sediments play a variety of essential roles in terms of maintaining the structure (i.e., assemblage of organisms in a system) and function (i.e., the processes that occur in a system) of aquatic ecosystems.

A.2 SEDIMENT QUALITY ISSUES AND CONCERNS

Traditionally, concerns relative to the management of aquatic resources in freshwater systems have focused primarily on water quality. However, the importance of sediments in determining the harmful effects of chemical contaminants on aquatic organisms (including plants, invertebrates, amphibians, fish, and reptiles), wildlife (birds and mammals), or human health has become more apparent in recent years (Long and Morgan 1991; USEPA 1994a,b, 1997a, 1998). Specifically, sediment quality is important because many toxic contaminants (such as metals, PAHs, PCBs, chlorophenols, and pesticides), found in only trace amounts in water, can accumulate to elevated levels in sediments. Consequently, sediments can serve both as reservoirs, and as potential sources, of contaminants to the water column. In addition, sediment-associated contaminants have the potential to adversely affect sediment-dwelling organisms (e.g., by causing direct toxicity or altering benthic invertebrate community structure; Chapman

1989). Therefore, sediment quality data (i.e., information on the concentrations of chemical substances) provide essential information for evaluating ambient environmental quality conditions in freshwater systems (i.e., determining if sediments, sediment-dwelling organisms, wildlife, or human health have been injured by releases of toxic or bioaccumulative substances into the environment).

The presence of elevated concentrations of chemical substances in aquatic sediments represents an environmental concern because sediments provide essential and productive habitats for communities of sediment-dwelling organisms, including epibenthic and infaunal species. These species fall into several taxonomic groups including scuds (amphipods), mayflies (ephemeropterans), stoneflies (plecopterans), caddisflies (trichopterans), dragonflies and damselflies (odonatans), midges (dipterans), water fleas (cladocerans), worms (oligochaetes), snails (gastropods), and clams (bivalves). The aforementioned organisms are important elements of freshwater ecosystems, representing important sources of food for many fish and wildlife species. However, the presence of sediment-associated contaminants in freshwater ecosystems can be harmful to sediment-dwelling organisms. Certain sediment-associated contaminants can also bioaccumulate in the tissues of aquatic organisms and, as a result, pose a potential hazard to those species that consume aquatic organisms, including piscivorous wildlife and humans.

APPENDIX B

SEDIMENT ASSESSMENT TOOLS

APPENDIX B

SEDIMENT ASSESSMENT TOOLS

There are a number of tools that can be used to carry out the assessment of sediment quality at contaminated sites. The following descriptions of each of these tools provide information on their derivation (as appropriate), applications, advantages, and limitations.

B.1 SEDIMENT CHEMISTRY DATA

Sediment chemistry is determined by extracting and measuring contaminants from the sediment matrix through various analytical techniques. Sediment chemistry measurements are used to determine the spatial and temporal extent of contamination at a site. In general, the target analytes measured in sediment quality surveys are determined based on land and water use information, in conjunction with existing sediment chemistry data and fish advisory information. However, the final list of analytes to be measured is also influenced by the equipment, technology, facilities, and funds that are available. The chemicals that are typically analyzed in sediments collected near urbanized and industrial areas include trace metals, PAHs, PCBs, organochlorine pesticides, and several other organic substances (e.g., TCDDs/TCDFs; chlorophenols, phthalates, etc.). Several other conventional variables, such as sediment particle size, TOC, AVS, aluminum, lithium, sulfides, and ammonia, are also usually measured to provide ancillary information for interpreting the resultant information on contaminant concentrations. While chemical concentrations are generally reported on a dry weight basis from extracted sediment samples, concentrations of contaminants in porewater and elutriate samples may also be determined to yield information on the bioavailable fraction of contaminants.

Sediment contaminant data may be normalized to specific variables to either reduce variance in the data or to further define the bioavailability of sediment-bound contaminants. For example, the concentrations of non-ionic organic contaminants may be normalized to TOC concentrations in sediment (Swartz *et al.* 1987; Di Toro *et al.* 1991). In addition, AVS-normalization procedures may be used to interpret extractable metals data (Di Toro *et al.* 1992). Alternatively, metal concentrations may be normalized to those of a reference element (such as aluminum or lithium; Schropp *et al.* 1990; Loring 1991) or to sediment particle size (Chapman 1992).

One of the principal strengths of using sediment chemistry is that it provides direct information on the presence of elevated contaminants in the sediments, relative to anthropogenic background or reference conditions. In addition, standard methods have been established for determining the concentrations of many analytes in sediment. Because measurements of sediment chemistry can be both accurate and precise, they provide a reliable basis for discriminating between contaminated and uncontaminated sites. Furthermore, analytical methods have been developed which may provide an indication of the potential bioavailability of certain substances (e.g., metals); however, the applicability of these methods (such as weak acid digestions) has not been fully established.

One of the main limitations of sediment chemistry data is that, by itself, it can not provide a basis for assessing the potential biological effects of contaminated sediments without the development or utilization of SQGs. The suite of analytes that is selected for determination may also limit the utility of these data. For example, important chemicals may be missed if the available land use data are not collected and appropriately interpreted (i.e., dioxins and furans should be measured in the vicinity of pulp mills; pesticides should be measured near agricultural areas). In some cases, the utility of these data is also limited by the use of inappropriate methods or by inadequate quality assurance practices.

B.2 NUMERICAL SEDIMENT QUALITY GUIDELINES

B.2.1 Sediment Quality Guidelines for the Protection of Aquatic Life

Sediment-associated contaminants can potentially cause adverse effects to sediment-dwelling organisms, depending on the exposure, hazard, and availability of the chemical. Numerical SQGs provide a basis for interpreting sediment chemistry by defining threshold concentrations of sediment-associated contaminants associated with adverse effects. Certain SQGs represent chronic toxicity thresholds, below which adverse effects on sediment-dwelling organisms are unlikely to occur. Other types of SQGs are intended to identify the concentrations of sediment-associated contaminants above which adverse effects are likely to be observed. As such, numerical SQGs represent valuable tools for assessing the potential effects of contaminated sediments.

A variety of theoretically-based and empirically-based approaches have been devised to formulate SQGs in North America and elsewhere in the world. The equilibrium partitioning approach represents the main theoretically-based approach for deriving numerical SQGs. The empirically-based approaches include the spiked-sediment toxicity test approach, screening level concentration approach, apparent effects threshold (AET) approach, sediment background approach, and the weight-of-evidence approach (see Beak Consultants Ltd. 1987, 1988; Chapman 1989; Persaud *et al*. 1989; Sediment Criteria Subcommittee 1989, 1990; USEPA 1989b, 1992a,b; MacDonald *et al*. 1992; MacDonald 1994; and Ingersoll *et al.* 1997 for further information on the various approaches).

The weight-of-evidence approach to the derivation of empirically-derived SQGs was originally developed to provide tools for assessing the potential for biological effects of sediment-associated contaminants tested in NOAA's National Status and Trends Program (Long and Morgan 1991; MacDonald 1994; Long *et al.* 1995). As a first step, a biological effects database for sediments (i.e., BEDS) was compiled which contained information on the effects of sediment-associated contaminants, including spiked-sediment toxicity test data, matching sediment chemistry and biological effects data from field studies, and SQGs derived using various approaches. The data on each chemical substance were retrieved from the database, incorporated into data tables, and sorted in ascending order of the chemical's concentration. Using the weight-of-evidence approach, numerical SQGs are derived using the information in both the effects data set and the no effects data set. Statistical evaluation of the information in these data sets facilitates the development of two sediment quality assessment values for each analyte, including a TEL and a PEL (CCME 1995; Smith *et al.* 1996). The TEL is intended to estimate the concentration of a chemical below which adverse effects occur only rarely (i.e., a minimal effects range). The PEL is intended to provide an estimate of the concentration above which adverse effects occur frequently (i.e., probable effects range) and was intended to be used in conjunction with the TEL to assess sediment quality conditions. As such, the TEL and PEL define three concentrations ranges for a chemical, including those that were rarely (below the TEL), occasionally (above a TEL but below a PEL), and frequently (above the PEL) associated with adverse effects.

The modified weight-of-evidence approach is characterized by a number of attributes that make it an attractive choice for deriving empirically-based SQGs for assessing sediment quality in freshwater ecosystems. First, this approach recognizes the uncertainty associated with the prediction of biological effects under a variety of field conditions and relies upon the evidence assembled from numerous independent studies. Second, reliance on field-collected data assures that the resultant SQGs incorporate the influence of mixtures of chemicals in sediments. Furthermore, the information in the BEDS is highly relevant to the guidelines derivation process because it applies to a wide range of biological organisms and endpoints, incorporating a large number of direct measurements on organisms that are normally associated with sediments. In addition, the information in the BEDS is representative of a wide range of sediment conditions. These attributes are likely to give the SQGs derived using the weight-of-evidence approach broad applicability, increasing the probability that the guidelines would be appropriate for implementation in Minnesota.

In spite of the benefits associated with this approach, a number of limitations are evident which could restrict application of these guidelines. First, the weight-of-evidence approach does not fully support the quantitative evaluation of *cause and effect* relationships between contaminant concentrations and biological responses. Although information from spiked-sediment toxicity tests and equilibrium partitioning models is included in the BEDS, the weight-of-evidence approach is still largely based on *associations* between contaminant concentrations and biological responses. Application of the recommended approach may also be restricted by other limitations on the available information, which restrict our ability to develop empirically-derived SQGs for some potentially important sediment contaminants in Minnesota, including mercury, PAHs, PCBs, trace metals, dioxins and furans, and a suite of pesticides. Furthermore, the effects-based SQGs do not consider the potential for bioaccumulation nor the associated effects on those species that consume aquatic organisms (i.e., wildlife and humans). Finally, the empirically-derived SQGs do not address the potential bioavailability of sediment-associated contaminants (Di Toro *et al.* 1990) and, therefore, may over- or under-estimate the toxicity of sediment-associated contaminants.

B.2.2 Sediment Quality Guidelines for the Protection of Wildlife and Human Health

Tissue residue-based SQGs provide practical tools for evaluating sediment quality relative to the potential for bioaccumulation (Cook *et al.* 1992). Tissue residue-based SQGs define the concentrations for individual chemicals or classes of chemicals in sediments that will not result in unacceptable levels of that substance in the tissues of aquatic organisms. The first step in their development involves the derivation or selection of an appropriate tissue residue guideline (TRG) for the substance or substances under consideration. These TRGs may be selected for the protection of human health (e.g., Food and Drug Administration action levels; USEPA 1989a), wildlife (New York State Department of Environmental Conservation fish flesh criteria for piscivorous wildlife; Newell *et al.* 1987), or both. In addition, relationships between concentrations of contaminants in sediments and contaminant residues in aquatic biota must be established. In general, the necessary biota-to-sediment accumulation factors (BSAFs) are determined from field studies or estimated using various modeling approaches. The SQGs are then derived by dividing the TRG by the BSAF (Cook *et al.* 1992).

Residue-based SQGs are important tools for conducting sediment quality assessments for several reasons. First and foremost, unlike the other SQGs, the tissue residue-based SQGs explicitly consider the potential for bioaccumulation and effects on higher trophic levels. In addition, the residue-based SQGs provide a basis for interpreting sediment chemistry data, in terms of the potential for adverse effects on human health and wildlife. Such assessments should be supported by direct measurements of contaminant concentrations in the tissues of aquatic organisms and wildlife species to assure that actual hazards are identified.

The chief disadvantage of this tool is that TRGs for the protection of wildlife have not been developed for most contaminants (Newell *et al.* 1987; Cook *et al.* 1992). Therefore, SQGs must be developed from TRGs applicable to the protection of human health. While guidelines, so developed, would adequately address human health concerns, other components of the ecosystem (e.g., fish-eating birds and mammals which have high daily consumption rates of aquatic organisms) may not be adequately protected.

B.3 TOXIC UNIT MODELS

The quality of aquatic ecosystems is commonly affected by multiple point and non-point sources, which have the potential to release a wide variety of chemical substances into receiving water systems. As such, sediments frequently contain hundreds of individual contaminants. Evaluation of the effects of individual substances in such complex mixtures is challenging because certain SQGs are intended to provide a basis for assessing the effects of individual contaminants when they occur by themselves (e.g., EqP-based SQGs). None of the guidelines that are currently available provide a basis for assessing the effects of the entire mixture of contaminants on sediment-dwelling organisms.

One of the most direct means of evaluating the relative hazards of individual chemicals in mixtures of sediment-associated contaminants is to normalize concentrations to independently-derived benchmark concentrations (Swartz and Di Toro 1997). Benchmarks that are commonly used in such assessments include SQGs, no effect concentrations (NOECs), and median lethal concentrations (LC_{50}s). Such normalized concentrations are expressed as toxic units (TUs), where the number of TUs of a chemical in a sample is equal to the measured concentration divided by the benchmark concentration (Sprague 1970). In this way, the relative potential contribution of each chemical substance to the observed or predicted effect can be compared (Swartz and Di Toro 1997). The number of toxic units within each class of contaminant or within the entire mixture can then be summed to evaluate the relative potential for toxicity among various environmental samples. This procedure can be used to assess the biological significance of contaminant concentrations in porewater or whole sediments.

Recently, Long *et al.* (1998a) used this procedure to evaluate the relationship between chemical contamination and acute toxicity to sediment-dwelling organisms (i.e., amphipods). In this study, mean SQG-quotients were calculated for each sediment sample by first determining a SQG-quotient for each chemical substance (i.e., by dividing the chemical concentration by the mid-range SQG for that substance; i.e., ERM and PEL values) and then calculating the arithmetic mean of all of the SQG-quotients determined. The results of this study indicated that mean SQG-quotients provided effective tools for assessing the potential effects of contaminant mixtures on sediment-dwelling organisms, as indicated by the consistent and marked increase in the incidence of toxicity with increasing mean SQG-quotients.

The main advantage of toxic units models is that they provide a basis for evaluating the relative effects of various contaminants and classes of contaminants in complex contaminant mixtures in sediments. Calibration of the toxic units model using field-collected data also provides a means of evaluating the probability of observing toxicity in sediments that contain complex mixtures of contaminants (i.e., the cumulative effects of contaminant mixtures). As such, TU models provide an effective means of increasing the applicability of SQGs and the results of spiked-sediment toxicity tests.

One of the chief limitations of TU models is that their reliability is driven, to a large extent by the reliability of values that are used in the denominator (i.e., SQGs, NOECs, or LC_{50}s). In addition, TUs provide a basis for evaluating the relative potential for toxicity in each sample only, unless they are calibrated with independent data from field studies.

B.4 SEDIMENT TOXICITY TESTS

Laboratory toxicity tests have been developed to assess lethal and sublethal endpoints in surrogate organisms exposed to sediments under controlled conditions. These tests include short-term (≤10 days) and long-term (>10 days) exposure periods that are used to evaluate the biological significance of sediment contamination. These tests may be as simple as short-term tests on a single contaminant using a single species or as complex as mesocosm studies in which the long-term effects of mixtures of contaminants on ecosystem dynamics are investigated. In addition, tests may be designed to assess the toxicity of whole sediments (solid phase), suspended sediments, elutriates, sediment extracts, or porewater. The organisms that are routinely tested include microorganisms, algae, invertebrates, and fish.

Whole sediment toxicity tests are the most relevant for assessing the effects of contaminants that are associated with bottom sediments. The U.S. EPA (2000b) and American Society for Testing and Materials (ASTM 1997a,b, 1999 have developed and approved whole sediment tests for assessing the toxicity of freshwater sediments. For example, standard methods have been established for assessing the acute and/or short-term chronic toxicity of sediment-associated contaminants on the amphipod, *Hyalella azteca*, the midges, *Chironomus tentans* and *C. riparius*, the mayfly, *Hexagenia limbata*, and several other species (ASTM 1997a; USEPA 2000b). These procedures may be modified to assess toxicity to other benthic invertebrate species that occur in freshwater environments (ASTM 1997b). Similar guidance has also been developed under the U.S. EPA's Assessment and Remediation of Contaminated Sediments (ARCS) program (USEPA 1994b). Ten-day freshwater acute toxicity tests using *H. azteca* and *C. tentans* have been selected by the U.S. EPA's Sediment Tiered Testing Committee for Agency-wide use (USEPA 1998).

In addition to whole sediment toxicity tests, various procedures are available for assessing the potential for adverse effects on aquatic organisms due to the resuspension of sediments or partitioning of contaminants into the aqueous phase (ASTM 1997b). Perhaps the most sensitive and frequently used of these are the bacterial luminescence test (Microtox®; Schiewe *et al.* 1985; Burton and Stemmer 1988). Tests using algae, invertebrates, and fish also have been employed to assess the toxicity of the suspended and/or aqueous phases, including porewater. In addition, formal procedures for conducting water column bioassays and bioaccumulation tests have been recommended by the ASTM (1997a,c).

Toxicity tests have a number of advantages that make them particularly relevant for evaluating the effects of contaminated sediments on aquatic organisms. First, they provide quantitative information on sediment toxicity that provides a basis for discriminating between impacted and unimpacted sites. In addition, standard methods have been established to support the generation of reliable and comparable data, as well as to minimize the effects of the physical characteristics of the sediments. The results of these tests are also ecologically-relevant because they commonly employ resident species, and the tests provide a way to compare the sensitivities of different organisms. Furthermore, studies conducted throughout North America have demonstrated that aquatic organisms in standard sediment toxicity tests respond primarily to the contaminants in the sediments and porewater (i.e., not physical factors or other variables). These characteristics make them relevant for evaluating contaminant-related impacts in freshwater systems. Moreover, techniques for identifying the chemicals that are causing toxicity are being refined [i.e., toxicity identification evaluation (TIE) procedures and sediment spiking], which further support the identification of contaminants of concern.

Toxicity tests also have several limitations which influence their application in sediment quality assessments. For example, many of the tests that are currently available involve short-term exposures (i.e., 10-day) and, hence, may not be sensitive enough to detect subtle, chronic effects on sensitive species resulting from long-term exposure to chemicals of concern. In addition, field-collected sediments are manipulated prior to testing, which may affect their integrity and toxicity (Burton 1991). Similarly, certain sediment phases (e.g., organic extracts, elutriates) may be less relevant for evaluating the *in situ* effects of toxic substances in sediments. Likewise, the ecological relevance of certain tests has not been fully established (e.g., Microtox®; although it was not intended for this purpose but rather as an indicator of potential exposure). Importantly, certain test organisms may be more sensitive to certain classes of contaminants than others; therefore, it is necessary to use a suite of tests to cover the range of sensitivities exhibited by sediment-dwelling species in the field. However, limitations on the availability of resources often necessitate striking a balance between the number of samples that are collected within an area (which determines the ability to characterize spatial and temporal patterns of sediment toxicity) and the number of species and endpoint tests that are tested (which determines the ability to determine if individual samples are toxic).

B.5 TOXICITY IDENTIFICATION EVALUATION PROCEDURES

Toxicity identification evaluation (TIE) involves the application of specialized toxicity tests to identify the classes of contaminants (possibly individual substances) that are causing toxicity in sediment samples that contain complex contaminant mixtures. These procedures recognize that the results of toxicity tests cannot, by themselves, be used to identify the substance or substances that caused the observed effect. However, toxicity tests can be used to implicate the causative agents in environmental samples, if such samples are repeatedly manipulated to alter the availability of certain classes of chemicals. These procedures are conducted in three phases to characterize (Phase I), identify (Phase II), and confirm (Phase III) the substances that are responsible for causing the observed effects (Swartz and Di Toro 1997).

In Phase I of the process, the sediment sample is manipulated to alter or render unavailable generic classes of chemicals with similar properties (Ankley and Schubauer-Berigan 1995). For example, addition of EDTA to the sample is likely to bind cationic metals and render then biologically unavailable. If the sample is no longer toxic following the addition of EDTA, then cationic metals are implicated as the causative agent. Similarly, pH manipulations can be used to determine if ammonia is causing toxicity, aeration can be used to evaluate the role of volatile or oxidizable substances in the observed toxicity, and C_{18} columns can be used to extract nonionic compounds. The results of toxicity tests conducted following each of these manipulations provide information on the classes of chemicals that are contributing to the observed toxicity.

A combination of analytical and toxicological methods is used in the second phase of the process to identify the specific chemicals that may be contributing to toxicity. For example, if EDTA chelation reduced or eliminated the toxicity of the sample, then the chemical analyses would then focus on determining the concentrations of cationic metals. The concentrations of each metal would then be compared to the results of toxicity tests conducted to assess the toxicity of single substances (e.g., which result in an LC_{50} for a particular metal like zinc). If the concentration of a metal exceeds the concentration that is known to be toxic to the species tested, then that metal is implicated in the toxic response that was observed.

The third phase of the TIE process is designed to confirm that the suspect contaminants (i.e., which were identified in Phase II) are the actual toxicants (Ankley and Schubauer-Berigan 1995). A number of procedures are used in Phase III to provide a weight-of-evidence for confirming the role of suspect contaminants in the environmental sample. These methods include correlation analysis of toxicity and chemistry data, evaluation of species sensitivities to different chemicals, alteration of pH or other water characteristics that influence the toxicity of specific chemicals, and several other procedures (Swartz and Di Toro 1997). Together, the results of these investigations can be used to confirm or reject the conclusions reached in the first two phases of the process.

TIE procedures were originally developed for identifying the substances that caused toxicity in effluents. More recently, these procedures have been applied to porewater that is extracted from aqueous sediments. As the concentrations of chemicals in whole sediments are generally correlated with those in porewater, TIE procedures can be used to identify the substances that are likely causing toxicity in whole sediments. However, methods have not yet been established for routinely applying TIE procedures directly to whole sediments. The U.S. EPA is in the process of developing guidance for conducting TIEs on marine and freshwater sediments (Scott Ireland, U.S. EPA, personal communication, 2000).

The main limitations of TIE procedures are associated with the level of effort, and associated costs, that are required to generate reliable results. Because multiple manipulations, toxicity tests, chemical analyses, and confirmatory analyses are required for each environmental sample, the costs of processing numerous samples using these procedures are likely to be prohibitive. In addition, TIE procedures are most effective when a single substance or a limited number of contaminants is responsible for the observed toxicity; discrimination of the effect of individual substances is difficult when many chemicals are contributing to sediment toxicity (Swartz and Di Toro 1997).

B.6 BENTHIC MACROINVERTEBRATE COMMUNITY ASSESSMENTS

Benthic communities are assemblages of organisms that live in or on the bottom sediment. In most benthic community assessments, the primary objective is to determine the identity, abundance, and distribution of the species that are present. Because most benthic macroinvertebrates are relatively sedentary and are closely associated with the sedimentary environment, they tend to be sensitive to both short-term and long-term changes in habitat, sediment, and water quality conditions (Davis and Lathrop 1992). Therefore, data on the distribution and abundance of these species provides important information on the quality of the aquatic environment.

Assessments of benthic community structure have been used to describe reference conditions, to establish baseline conditions, and to evaluate the effects of natural and anthropogenic disturbances (Striplin *et al.* 1992; Reynoldson *et al.* 1998). In terms of evaluating sediment quality, such assessments are focused on establishing relationships between various community metrics (e.g., species richness, total abundance, and biomass) and measures of sediment quality (e.g., chemical concentrations and organic content). Data from benthic community assessments have the potential to provide relevant information for identifying impacted sites and, with appropriate supporting data, the factors that are contributing to any adverse effects that are observed.

Benthic community assessments have a number of advantages that make them useful for evaluating the status of benthic communities. First and foremost, the results of these assessments provide information that is directly relevant for evaluating benthic community status. In addition, standard methods for conducting such assessments have been established, facilitating unbiased random sampling, broad geographic coverage (including both contaminated and uncontaminated areas), and reducing variability in the results (i.e., by sampling under consistent hydrological conditions). Furthermore, the information generated is socially relevant (i.e., benthic species represent important food organisms for many sportfish species, such as walleye) and can be used to discriminate between sites that are degraded to various extents.

Benthic community assessments also have a number of limitations that restrict their use for evaluating contaminant-related impacts. Of primary concern, the information on benthic community structure can not be used alone to evaluate the cause of any impacts that are observed. While benthic communities certainly respond to chemical contamination in the sediment (Canfield *et. al.* 1994, 1996), they are also affected by a wide range of physical factors that are not directly related to sediment quality (e.g., low dissolved oxygen levels, grain size differences, and water depth). For the St. Louis River AOC, the results from multivariate redundancy analysis (RDA) on 13 environmental parameters revealed that the majority of variation in benthic community structure was attributed to water depth and site distance from the headwaters (Breneman *et al.* 2000). In addition, benthic community composition exhibits significant spatial, short-term temporal, and seasonal variability; therefore, interpretation of the data relative to contaminant effects can be difficult. The selection of reference sites can also influence the results of benthic community assessments (Crane *et al.* 1997). To complicate matters further, there is little agreement among benthic ecologists on which metrics are the most appropriate for evaluating the status of the community as a whole. Therefore, it is difficult to determine if information on individual organisms (e.g., morphological changes, biomarkers), populations of organisms (e.g., abundance of indicator species, population size structure, etc.), community structure (e.g., species richness, community indices, etc.), or community function (e.g., energy processing, presence of functional groups) should be used as indicators of benthic community status (SETAC 1997).

B.7 SEDIMENT QUALITY TRIAD

The sediment quality triad (Triad) was developed as a tool to support site-specific assessments of sediment quality (Long and Chapman 1985; Long 1989). The Triad is based on correspondences between three measures of sediment quality conditions: sediment chemistry; sediment toxicity tests; 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 toxicity tests provide information that may be used to evaluate the effects of sediment-associated contaminants on resident or indicator species. Measures of *in situ* biological effects, such as benthic infaunal community structure or histopathological abnormalities in benthic fish species, provide information on alterations of resident communities that may be related to sediment chemistry. Integration of these three components provides comprehensive information which may be used to evaluate and rank the relative priority of the areas that have been surveyed.

The major advantage of the sediment quality triad is that it integrates the data generated from the three separate measurements. Thus, natural variability in biotic characteristics can be differentiated from the variability due to the toxic effects of environmental contaminants. For example, variability in benthic community composition may be due to the presence of contaminants in sediments, or it may be related to differences in other aspects of habitat quality (i.e., water depth, organic carbon, grain size). The Triad provides a basis for distinguishing these relationships; however, the approach can not be used alone to establish cause and effect relationships for individual substances. The other advantages of the Triad are that it may be used for any measured contaminant, it may include both acute and chronic effects, and it does not require information on the specific mechanisms of interaction between organisms and toxic contaminants. The integration of the three data types provides a weight-of-evidence regarding contaminant-related impacts on the benthic community.

The major limitations of the Triad are as follows: statistical criteria have not been developed for use with the Triad; a large database is required; 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 (Chapman 1989). In addition, the Triad may not explicitly consider the bioavailability of sediment-associated contaminants.

B.8 BIOACCUMULATION ASSESSMENTS

In addition to causing direct effects on aquatic biota, contaminants can also accumulate in the tissues of sediment-resident organisms. Because many benthic and epibenthic species represent important components of the food web, sediment-associated contaminants can be transferred from them to higher trophic levels. In this way, contaminated sediments represent a potential hazard to aquatic predators (e.g., fish), to wildlife species that consume aquatic organisms, and to people that consume contaminated fish and piscivorous wildlife. Bioaccumulation tests can be used as one piece of information to assess ecological and human health impacts.

Freshwater bioaccumulation tests can be conducted in several ways (ASTM 1997c; USEPA 2000b) on a number of different organisms (e.g., freshwater clams, fish, chironomids, larval mayflies, amphipods, and oligochaetes). Bioaccumulation testing often involves exposing test organisms to sediments for a period of 28 days, after which their tissues are analyzed for chemical contaminants. The 28-day standard exposure interval was adopted with the recognition that higher molecular weight neutral organic chemicals (i.e., log $K_{OW} \geq 4.7$) are not expected to reach steady-state bioaccumulation in that length of time (Connell 1990). However, for most bioaccumulating chemicals, 28-day exposures will result in a proportion of steady state sufficient to demonstrate the bioavailability, or lack of it, of chemicals associated with sediments. Tissue residues measured in test sediment organisms at the end of the exposure period are compared with residues in organisms exposed to an uncontaminated reference sediment.

Actual bioaccumulation will be influenced by several variables which are dependent on characteristics of the exposure medium and on physiological and metabolic characteristics of the organism (McFarland 1995; U.S. EPA Bioaccumulation Analysis Workgroup 2000). A greater source of variability in both laboratory and field studies may result from the use of organisms that are not in intimate and continuous contact with the sediments. The chemical must be fully bioavailable; if desorption of the chemical from sediment binding sites is very slow, an exposed organism may never reach equilibrium bioaccumulation (McFarland 1995). This will affect the kinetics of the distribution of a chemical between sediment and the test organism.

Several studies have evaluated the selection of organisms for sediment bioaccumulation testing (as cited in USEPA 2000b). *Lumbriculus variegatus* (Oligochaeta) have been selected as a good sediment bioaccumulation testing organism because: they are in contact with the sediments, are easy to culture in the laboratory, and are tolerant of varying physico-chemical characteristics of sediment (USEPA 2000b). The use of *L. variegatus* in laboratory bioaccumulation studies has been field validated with natural populations of oligochaetes. Brunson *et al*. (1998) compared the bioaccumulation of contaminants in laboratory exposed *L. variegatus* and field-collected oligochaetes; select PAH and DDT peak concentrations were similar in field-collected oligochaetes and *L. variegatus* exposed for 28 days in the laboratory. Another advantage of using *L. variegatus* is that it does not biotransform PAHs, a common sediment contaminant (Harkey *et al*. 1994).

The primary disadvantage of conducting sediment bioaccumulation tests is their cost, which is influenced by the time-intensive nature of the tests. In addition, an adequate tissue mass must be obtained for chemical analysis. This could require the use of additional replicates for analyses of multiple chemicals. The test organisms may also avoid the test sediments, thereby reducing bioaccumulation exposure. Another disadvantage is that field collected sediments may include indigenous organisms that would need to be separated from the test organisms at the end of the test. An appropriate reference control site must also be selected. Quality assurance requirements are also more stringent for bioaccumulation tests than for conventional sediment toxicity tests. Finally, the relationships between bioaccumulation and direct toxicity to aquatic organisms are poorly defined.

APPENDIX C

ECOSYSTEM-BASED MANAGEMENT APPROACH

APPENDIX C

ECOSYSTEM-BASED MANAGEMENT APPROACH

C.1 HISTORICAL DEVELOPMENT OF THE ECOSYSTEM APPROACH

The ecosystem approach to planning, assessment, and management is the most recent phase in an historical succession of approaches to environmental management. Previously, humans had been considered to be separate from the environment in which they lived. This *egocentric approach* viewed the external environment only in terms of human uses. For example, untreated sewage and industrial waste used to be discharged into the St. Louis River without regard to its environmental consequences. In the early 1900s, pollution was noticeable in the river and harbor; this prompted the U.S. Geological Survey and Minnesota Department of Health (MDH) to investigate the sources and nature of pollution during 1903-1904 (MPCA and WDNR 1992). Additional surveys conducted by MDH during 1928-1929, 1947-1948, 1954, and 1961 documented further degradation of water quality in the St. Louis River (MPCA and WDNR 1992). By 1967, the fisheries habitat in the lower St. Louis River had been degraded to a poor condition (MPCA and WDNR 1992). Thus, it was clear that human activities had created significant and far-reaching impacts to the environment and to people residing within the St. Louis River watershed.

This recognition of pollution problems led to command and control-type management strategies by the MPCA and WDNR to initiate wastewater treatment and to develop permit limits for effluent discharges. Enforcement actions (e.g., fines) were the most common techniques taken against companies that exceeded their permit limits. However, most government agencies have learned that cooperative techniques to educate companies, encourage pollution prevention strategies, incorporate multimedia approaches to the assessment, management, and remediation of contaminated sites, and involvement of stakeholder groups within the watershed yield more positive benefits to the community, the company, and the government agency. Enforcement actions are now taken as a last resort at the MPCA. Therefore, the MPCA has learned to take a more holistic approach to environmental management, in which humans are considered as integral components of the ecosystem. The ecosystem approach provides this progressive perspective by integrating the *egocentric view* that characterized earlier management approaches, with an *ecocentric view* that considers the broader implications of human activities

The primary distinction between the egocentric and ecosystem approaches is whether the system under consideration is external to (in the egocentric approach) or contains (in the ecosystem approach) the human or ecological population under study (Vallentyne and Beeton 1988). The conventional concept of the environment is like that of a *house* - external and detached; in contrast, ecosystem implies *home* - something that we feel part of and see ourselves in, even when we are not there (Christie *et al.* 1986). The change from the egocentric approach to the ecosystem approach necessitates a change in the view of the environment from a political or people-oriented context to an ecosystem-oriented context (Vallentyne and Beeton 1988). The essence of the ecosystem approach is that it relates *wholes* at different levels of integration (i.e., humans and the ecosystems containing humans) rather than the interdependent parts of those systems (i.e., humans and their environment; Christie *et al.* 1986).

The ecosystem approach is not a new concept and it does not hinge on any one program, definition, or course of action. The Canadian Council of Ministers of the Environment defined the ecosystem approach as a geographically comprehensive approach to environmental planning and management that recognizes the interrelated nature of environmental media and that humans are key components of ecological systems (CCME 1996). It places equal emphasis on concerns related to the environment, the economy, and the community. This approach recognizes that it is the human interactions with ecosystems, rather than the ecosystems themselves, that must be managed (Environment Canada 1996; Thomas *et al.* 1988). Adopting an ecosystem approach means viewing the basic components of an ecosystem (i.e., air, water, land, and biota) and its functions in a broad context, which effectively integrates environmental, social, and economic interests into a decision-making framework that embraces the concept of sustainability (Figure 4) (CCME 1996).

This expanded view of the ecosystem helps shape the planning, research, and management decisions that are made by government agencies, industry, and stakeholder groups within and pertaining to the St. Louis River ecosystem. For example, the Northeast Midwest Institute is currently examining economic benefits of sediment remediation in the St. Louis River AOC (Northeast Midwest Institute Web Site: http://www.nemw.org/glvalpln2.htm). Social issues, such as environmental justice and quality of life for people located in neighborhoods adjacent to contaminated sites, are also of concern. These economic and social issues will be incorporated into decisions about cleaning up contaminated sediment sites, particularly in the more populous Duluth-Superior Harbor area.

C.2 BENEFITS OF THE ECOSYSTEM APPROACH

The ecosystem approach is superior to the approaches to environmental management that have been used previously for a number of reasons. First, the ecosystem approach provides a basis for the long-term protection of natural resources, including threatened and endangered species. A number of threatened, special concern, and endangered plants and animals in the St. Louis River AOC are listed in the Stage I RAP (MPCA and WDNR 1992). In the past, management decisions were typically made with specific short-term goals (i.e., within a single political mandate). In contrast, the ecosystem approach necessitates a long-term view of the ecosystem which necessarily considers the welfare of its biotic components. Hence, management decisions are more likely to be consistent with sustainable development goals.

Second, the ecosystem approach provides an effective framework for evaluating the real costs and benefits of new construction projects. Previously, decisions regarding the development of industrial and municipal projects were heavily weighted toward financial benefits and job creation. Neither the long-term impacts of these projects nor the sustainability of the resources upon which they depended were fully considered. In contrast, implementation of the ecosystem approach assures that the long-term effects of developmental activities are incorporated into the assessment process. Therefore, management decisions are less likely to be made based solely on political considerations, such as job creation. For example, businesses and wastewater treatment plants that discharge into the Duluth-Superior Harbor must now comply with the Great Lakes Initiative (GLI). This initiative severely restricts the discharge of critical pollutants, such as mercury, into receiving waters. In order to comply with the GLI, WLSSD has increased its pollution prevention strategies to reduce the amount of mercury in the waste stream, in addition to the incorporation of more stringent clean-up technologies.

The ecosystem approach also enhances the multiple use of natural resources. In the past, governments have often allocated natural resources for the exclusive use of single industrial interests. Implementation of the ecosystem approach ensures that all stakeholders have an opportunity to participate in the establishment of management goals for the ecosystem and that governments do not make political decisions that benefit a single interest group, at the expense of other beneficial uses of natural resources.

Environmental research and monitoring activities are essential elements of the management programs of the MPCA, Minnesota Department of Natural Resources (MDNR), WDNR, and Fond du Lac Band. Each of these groups has staff working on sediment-related projects in the

St. Louis River AOC. The ecosystem approach provides a basis for focusing these activities by establishing very clear management goals for the ecosystem. Therefore, research and monitoring activities are driven by the needs of the program (to determine if the management goals are being met), rather than by the interests of individual scientists or by political factors.

The ecosystem approach also provides a mechanism for bringing diverse stakeholders together to create a shared vision. The Ecosystem Charter for the Great Lakes – St. Lawrence Basin is one such example of how a shared vision has been developed. The Charter incorporates a series of commonly held principles, findings, and related action items to guide ecosystem management in the Great Lakes – St. Lawrence Basin; more than 130 agencies, organizations, and jurisdictions have endorsed the charter (Ecosystem Charter Web Site; http://www.glc.org/ecochart/letter.html).

One of the most important benefits of the ecosystem approach is that it directly involves the public in decision-making processes. Specifically, this approach provides a forum for public input at a non-technical level (i.e., during the establishment of management goals and ecosystem objectives), which can be both effective and non-threatening. The detailed technical issues are then left to those who are charged with the management of these ecosystems. The framework for implementing the approach also assures that these managers can be held accountable for the decisions that they make. The Sediment Contamination Workgroup of the St. Louis River CAC, and the community workgroups for two Superfund sites in the lower estuary, provide a local mechanism for public input on contaminated sediment issues. Public forums have also been held on impairment issues in the St. Louis River AOC. In addition, the MPCA hosted a Governor's Forum in Duluth, MN during May 1999 to better understand citizens' concerns and values about the environment. The highest priority environmental issues in the Duluth area were about water quality degradation and toxins in the environment (MPCA 1999).

Traditionally, environmental impact assessments have not consistently provided reliable information for evaluating the effects of anthropogenic developments on the ecosystem. In the ecosystem approach, however, the functional relationships between human activities, changes to the physical and chemical environment, and alterations in the biological components of the ecosystem are established before making important management decisions. Therefore, management decisions are more likely to be consistent with the long-term goals that have been established. Subsequent monitoring activities can then focus on the ecosystem components that are most likely to be affected.

The ecosystem approach also facilitates the restoration of damaged and degraded natural resources. By explicitly identifying the long-term impacts of degraded ecosystems on designated land and water uses, this approach more clearly delineates the benefits of restoration and remedial measures. Therefore, limited resources can be focused on restoration projects that are likely to yield the greatest benefits to the ecosystem as a whole.

C.3 A FRAMEWORK FOR ECOSYSTEM-BASED MANAGEMENT

Based on the work that has been conducted in a number of ecosystem initiatives, the Canadian Council of Ministers of the Environment has developed a framework for ecosystem-based management (CCME 1996). The framework advocates a four-step process to support informed decision-making, which includes (Environment Canada 1996) (Figure 4):

- *Identify and assess the issues and collate the existing ecosystem knowledge base*, with a focus on ecosystem science, socioeconomic information, historical resource use patterns, and traditional knowledge;
- *Develop and articulate ecosystem health goals and objectives*. In this step, the ecosystem stakeholders use the common knowledge base to formulate broad-based, long-term goals and more specific objectives for the St. Louis River AOC. Societal values are reflected in the goals and objectives through consultation with competing users of ecosystem resources;
- *Develop or select ecosystem health indicators to gauge progress toward ecosystem health goals and objectives*. Once ecosystem objectives have been identified, sets of indicators (including specific *metrics* and *targets*) are developed to track progress toward those objectives; and,
- *Conduct directed research and monitoring*. Information gaps identified in steps one and three can be used to focus research expenditures and monitoring activities.

This framework is intended to support sound management decisions that help to protect, maintain, restore, and enhance ecosystem health. However, for this approach to be effective, the decision-making processes must be accessible to stakeholders, including governments, tribes, industries, communities, and nonprofit organizations. Indeed, full implementation of ecosystem-based management necessitates the development of cooperative decision-making processes, in which stakeholders and the public participate as partners. The development of watershed management plans and/or remedial action plans represent two important mechanisms for sharing authority, responsibility, and accountability for decision-making among participants in these processes. Importantly, these processes provide a basis for clearly identifying the actions that should be taken to achieve the ecosystem goals and objectives.

This framework for ecosystem-based management should be viewed as a dynamic process in which new information (i.e., resulting from research and monitoring programs) continually feeds into an expanding knowledge base. Ecosystem goals, objectives, and indicators are refined as societal values change or new information becomes available. Importantly, the indicators should be re-evaluated following management decisions to help evaluate the efficacy of the actions taken in terms of advancing progress towards the ecosystem objectives.

APPENDIX D

DESIGNATED USES OF AQUATIC RESOURCES IN THE ST. LOUIS RIVER AREA OF CONCERN

APPENDIX D

DESIGNATED USES OF AQUATIC RESOURCES IN THE ST. LOUIS RIVER AREA OF CONCERN

D.1 BACKGROUND

In order to develop indicators of sediment quality conditions, it is important to understand the designated water uses in the St. Louis River AOC. Under Section 303(c) of the Clean Water Act, states are required to develop water quality standards for all surface waters. At a minimum, such standards must include designated water uses, in-stream criteria to protect such uses, and an anti-degradation policy (USEPA 1991). In Minnesota, designated water uses include domestic water supply (i.e., drinking water), fisheries and recreation, agriculture and wildlife, and limited resource value waters (USEPA 1988). In Wisconsin, state water quality standards are intended to protect public water supplies, as well as fish and aquatic life (USEPA 1988). While these water use designations are essential for supporting water quality and quantity management, they were not intended to be directly applicable for managing sediment quality or ecosystem integrity. Sediment quality is briefly discussed in the Minnesota Rule for Total Maximum Daily Loads (TMDLs) for the Lake Superior Basin (Minn. Rules Ch. 7052; http://www.revisor.leg.state.mn.us/arule/7052/).

The St. Louis River watershed is a complex ecosystem that supports a wide range of human and ecological uses. The St. Louis River CAC is currently developing a habitat plan, in conjunction with the MDNR, to delineate the types of habitat found in this ecosystem. Effective management of sediment quality conditions within the St. Louis River AOC requires an understanding of the linkages between sediment quality conditions and the designated uses of the aquatic ecosystem. In the St. Louis River AOC, there are five designated uses of the aquatic ecosystem that have the potential to be adversely affected by sediment contamination, including:

- aquatic life;
- aquatic-dependent wildlife;
- human health;
- recreation and aesthetics; and,
- navigation and shipping.

The ultimate goal of the St. Louis River RAP is to restore impaired uses and to protect those uses that have not been impaired (MPCA and WDNR 1992). To this end, the MPCA and WDNR are responsible for formulating pollution prevention, monitoring, management, and remediation strategies to protect and maintain each of the five designated water uses that are linked to sediment quality conditions in the St. Louis River AOC. These strategies need to take into consideration the periodic or, in some cases, frequent physical disturbances resulting from seiches, ice scour, dredging, ship traffic, and other physical and hydraulic factors in the lower estuary. Such disturbances are likely to influence the productivity of benthic habitats, especially in the shipping channels. In physically disturbed or industrial areas, the MPCA and WDNR may want to consider whether it would be appropriate to permit somewhat higher levels of sediment contamination in these areas and focus limited resources for remediation on those areas that are likely to be more biologically productive. However, care must be taken to minimize the potential offsite migration of toxic and/or, bioaccumulative substances from these areas. In addition, contaminant concentrations can not exceed levels that would preclude the beneficial use of dredged material for habitat enhancement or land application.

D.2 AQUATIC LIFE

Aquatic life represents an important water use in the St. Louis River AOC, as a wide variety of fish and other aquatic organisms utilize habitats within the watershed throughout portions of their life cycle. For example, diverse communities of invertebrates utilize benthic habitats throughout the river system; these invertebrates represent important food organisms for many fish and wildlife species. In addition, walleye, lake sturgeon, and catfish utilize habitats upstream of the USX Superfund site for spawning activities; habitats throughout the AOC are utilized for rearing activities. Northern pike tend to spawn in more quiescent habitats, particularly in the vicinity of Grassy Point, Allouez Bay, and the area upstream of the Bong Bridge. Many non-sport and non-commercial species of fish also utilize habitats within the AOC for all or a portion of their life cycles.

Instream water uses are probably the most sensitive to the effects of sediment-associated contaminants. Aquatic organisms can be affected by contaminated sediments in several ways, including direct exposure of sediment-dwelling organisms to contaminated sediments (both infaunal and epibenthic species), exposure of pelagic species to degraded water quality as a result of desorption of contaminants from the sediments, and accumulation of toxic substances in the food web. Sediment chemistry, sediment toxicity testing, benthic invertebrate community assessments, bioaccumulation assessments, and SQTs provide the most direct means of determining the extent to which this designated water use is being protected in the St. Louis River AOC (see Appendix B for additional information about these tools).

D.3 AQUATIC-DEPENDENT WILDLIFE

There are a variety of aquatic-dependent wildlife species that occur within the St. Louis River AOC, including a variety of shorebirds (e.g., sandpipers), waterfowl (e.g., ducks), wading birds (e.g., herons), colonial nesting birds (e.g., terns), raptors (e.g., ospreys), muskrats, mink, beavers, and river otters. As these organisms represent integral components of the aquatic ecosystem, their protection should be identified as a high priority goal. While some wildlife species can also be exposed directly to contaminated sediments, concerns relative to the protection of aquatic-dependent wildlife species are primarily focused on dietary exposure to bioaccumulative contaminants (i.e., through the consumption of contaminated fish, other aquatic organisms, and piscivorus wildlife). Reduction in the availability of food organisms is also a concern with respect to the viability of wildlife populations. Numerical SQTs (i.e., bioaccumulation-based SQTs) and numerical tissue residue guidelines provide relevant tools for determining if wildlife are being adversely affected by contaminated sediments in the St. Louis River AOC (see Appendix B for additional information about these tools).

D.4 HUMAN HEALTH

Protection of human health has typically been a major focus during the development of water quality standards. With respect to sediment quality conditions, human health can be adversely affected by direct dermal exposure to contaminated sediments (e.g., swimming or wading) and through the consumption of contaminated fish, crayfish, snapping turtles, and waterfowl tissues. Long-term exposure to sediment-associated contaminants can result in both carcinogenic and non-carcinogenic effects in humans (Crane 1996). Numerical SQTs (bioaccumulation-based SQTs) and numerical tissue residue guidelines can be used to assess the potential effects of contaminated sediments on human health in the St. Louis River AOC (see Appendix B for additional information about these tools).

D.5 RECREATION AND AESTHETICS

Recreation and aesthetics are emerging water uses, which are likely to become even more important in the future as tourism increases in this area (MDNR and NOAA 1997). Recreational water uses include both contact recreation, such as swimming and wading, and non-contact recreation, such as boating and fishing. Recreational activities that involve direct contact with water and sediments can be impaired when sediment contaminant concentrations reach levels that cause skin irritation, respiratory problems, or other significant non-carcinogenic and/or carcinogenic effects. In some cases, these effects may necessitate beach closings. In contrast, non-contact recreation can be impaired when fish populations are degraded, when fish consumption advisories are issued, when fish have an increased incidence of tumors and other deformities, or when environmental conditions adversely affect the boating experience (e.g., through noxious odors or visual impairments, such as oil sheens). In addition to the influence of environmental conditions, aesthetic water uses can be impaired through the loss of fish and wildlife habitats or through degradation of wildlife populations (e.g., reduction in opportunities for wildlife viewing). The effects of contaminated sediments on recreational and aesthetic water uses are difficult to evaluate directly; however, information on benthic invertebrate community status, tumor incidence, and fish populations is useful for conducting indirect assessments in the St. Louis River AOC (see Appendix B for additional information about these tools).

D.6 NAVIGATION AND SHIPPING

As one of the major ports in the Great Lakes, shipping is an important water use in the Duluth-Superior Harbor. Preservation of this water use within the St. Louis River AOC is dependent upon maintaining appropriate depths in shipping channels by carrying out maintenance dredging. On an annual basis, an average of 150,000 cubic yards of sediments is dredged from the Duluth-Superior Harbor (USACOE 1997). This water use can be adversely affected when the concentrations of sediment-associated contaminants exceed the levels that are specified for beneficial use of dredged material (e.g., beach nourishment). In such cases, the dredged material must be transported to the Erie Pier confined disposal facility (CDF) for physical sorting to facilitate disposal of the fine material and beneficial use of the sandy material. Approximately 25% of the dredged material are classified as coarse sand (USACOE 1997). As the Erie Pier CDF nears its current capacity, increased costs for disposing of dredged material and/or significant decreases in dredging activity are likely to occur. Both of these options will have adverse effects on navigation and shipping. Sediment chemistry, numerical SQTs, toxicity testing, benthic invertebrate community assessments, and bioaccumulation assessments represent the most important tools for assessing the likelihood that dredging can occur or dredge material can be utilized for beneficial uses (see Appendix B for additional information about these tools).

APPENDIX E

SCREENING CRITERIA FOR BEDS/SEDTOX CO-OCCURRENCE DATA

APPENDIX E

SCREENING CRITERIA FOR BEDS/SEDTOX CO-OCCURRENCE DATA

***Must be Present**

Reference:____ Reference Number:_______________

1*. Does data set contain matching sediment chemistry and biological effects? (i.e., biological and chemical data collected from the same location at the same time)
NO ____ UNACCEPTABLE
YES ____ Page Reference(s):

2. What is the location of sampling site(s)? Collection Date? Page ref. for site description?

3. Freshwater ___ Estuarine ___ Marine ___ Salinity ___

I Sediment Chemistry

4*. Is there at least 1 non-toxic sample?
NO ____ UNACCEPTABLE
YES ____ Number of non-toxic: _____ Number of toxic: _____

5. Was bioassay conducted on unique ___ or composite ___ samples.
Number of replicates? _____ Size of composite area? ________

6. What chemistry data has been collected? (i.e., metals, PAHs, pesticides, pH, DO, TOC...)
Metals ___ PCBs ___ pH ___ TOC____
PAHs ___ Pesticides ___ DO ___ AVS____

7. Are detection limits below the respective ERLs or TELs?
NO ___ UNACCEPTABLE YES ___

8. Are total metal concentrations measured?
NO ___ SEM metals may not be included.
YES ___ SEM metals may be included.

9. Collection instrument __________________; Sediment depth ________________.

10*. What type of sediment was used:

Bulk Sediment ___	Porewater ___	Organic Extract ___	Elutriate ____
(Sediment with porewater)	(Extract porewater from sediments and expose water column species)	(Sediment extracted with organic solvent and expose liquid form) OTHER TOXICITY DATA ALSO NEEDED	(Sediments with water, mixed, settled and exposed water column species) UNACCEPTABLE

11*. What type of toxicity test was conducted? Length of test? ________________
Static ___ (Water, sed-no change)
Static Renewal ___ (Water, sed-some water change)
Flow-Through ___ (Water, sed-water flowing through)

12*. Are appropriate analytical procedures used to determine total concentrations of the analytes in bulk sediment samples? What method(s) were used?
(Metals: partial digestion, analysis of elutriates or extracts are unacceptable.)

13. Is a dilution series used?
NO ___ YES ___ UNACCEPTABLE

14*. Are measured dry weight contaminant concentrations reported? Conversion from wet weight to dry weight concentration may occur ONLY if data on moisture or TOC are provided. Nominal concentrations are unacceptable.
NO ___ UNACCEPTABLE YES ___ Page reference(s):

II BIOEFFECTS

15*.

a Do toxicity tests employ appropriate laboratory procedures? (ASTM: E1367, E1611, E1706)
NO ___ UNACCEPTABLE YES ____

b Have the following been recorded during testing?
Temperature _____; pH _____; Hardness _____; Conductivity ______;
Salinity ______; DO ______; Alkalinity _____; Ammonia ______.

c Does DO Remain above
60% ___ Needed for Marine
40% ___ Needed for Fresh Water
NO ___ UNACCEPTABLE

d Temperature_______

e Is Temperature within natural range, fluctuate less then 3°C, and have a time weighted average within 1°C of selected temp?
YES __ NO __ UNACCEPTABLE Range__________

f Do Hardness, alkalinity, pH, or ammonia vary more than 50% (for freshwater samples)?
NO __ YES __ UNACCEPTABLE
Range: DO ___________; Alk _________; pH ___________; NH_3________

g Have Salinity levels in porewater been adjusted (for marine samples)?
NO __ YES __ UNACCEPTABLE Range________

h List procedure reference(s) or brief details:

____

16*. Were biological responses compared to the control __ or reference __ sites?
List the Control and Reference sites? Positive Control __________.
reference = uncontaminated site within the same waterbody or watershed; control = uncontaminated site outside the tested water body

____

17*.
a Have sediment samples used for biological testing been frozen?
NO __ YES __ If yes both biological and chemical testing must be performed after thawing sediments.

b Have sediment samples been stored for more than eight (8) weeks prior to biological testing?
NO __ YES __ UNACCEPTABLE
What was the holding time? _____ .

c Are appropriate procedures used for collecting, handling, and storage of sediments?
NO ___YES __ List procedures reference(s) or brief details:

____

18. Identify species used in toxicity testing. Identify organism sources.

____

19. What life stage were the test species at the start of the test?
(*Hyalella azteca* 7-14 day old; *Chironomus tentans* third-instar larvae; *Chironomus riparius* second instar or younger; *Daphnia magna* 5 days old; *Ceriodaphnia dubia* <24h old; *Hexagenia* spp. 3-4 months old; *Tubifex tubifex* adult; *Diporeia* spp. juveniles)

____

20. Organism acclimation time ______________.

---

21. What percentage of the control survived?
Mean range _______
70% for (*Chironomus riparius*, *Chironomus tentans*)
80% for (*Hexagenia* spp., *Daphnia magna*, *Ceriodaphnia dubia*, *Hyalella azteca*)
90% for (*Diporeia* spp., *Tubifex tubifex*, *Polychaetous* annelids, marine amphipods, others)
NO ___ UNACCEPTABLE

---

22. Reference Samples
Survival _______%
Conc. Less than TEL and ERLs? YES ___ NO ___
Grain size ____; % sand ____; % silt ____; % clay_____

---

23. BENTHIC COMMUNITY ANALYSIS
a Is there a benthic community abundance analysis?
NO ___ YES ___ List taxa (e.g., amphipod, sponges,...) Upon which the analysis focuses:

---

b* Do each of the sites within a sampling area have the same general characteristics (i.e., same depth of overlying water, same salinity in overlying water, etc.)?
NO ___ UNACCEPTABLE YES ___ Briefly list details:

III STATISTICAL ANALYSIS

24. Are appropriate statistical procedures reported?
NO ___ YES ___ List procedure reference(s):

Additional Notes/Comments:

APPENDIX F

SCREENING CRITERIA FOR BEDS/SEDTOX SPIKED SEDIMENT BIOASSAY DATA

APPENDIX F

SCREENING CRITERIA FOR BEDS/SEDTOX SPIKED SEDIMENT BIOASSAY DATA

***Must be Present**

Reference:____ Reference Number:___________________

1*. Does data set contain matching sediment chemistry and biological effects (i.e., biological and chemical data collected from the same location at the same time).
NO ____ UNACCEPTABLE
YES ____ Page Reference(s):

____

2. What is the location of sampling site(s)? Collection Date?
Page Ref. for site description?

____

3. Freshwater ____ Estuarine ___ Marine ___ Salinity____

____

I Sediment Chemistry

4*. Is there at least 1 non-toxic sample?
NO ____
YES ___ Number of non-toxic: ____ Number of toxic:___

____

5. Was bioassay conducted on unique ___ or composite ___ samples.
Number of replicates?___ Size of composite area?

____

6. What chemistry data has been collected? (i.e., metals, PAHs, pesticides, pH, DO, TOC...)
Metals ___ PCBs ___ pH ___ TOC __
PAHs ___ Pesticides ___ DO ___ AVS __

____

7. Are detection limits below the respective ERLs or TELs?
NO ___ UNACCEPTABLE YES __

____

8. Are total metal concentrations measured?
NO ___ SEM metals may not be included.
YES ___ SEM metals may be included.

9. What are the conditions in the bioassay chamber? (i.e., TOC, AVS, Grain size, NH_3, and H_2S).
TOC ___; AVS ___; Grain size ___; NH_3 ___; H_2S ___; Salinity ___

10. Collection instrument ____________________; Sediment depth ____________;

11*. What type of sediment was used:

Bulk Sediment ___	Porewater ___	Organic Extract ___	Elutriate ___
(Sediment with porewater)	(Extract porewater from sediments)	(Sediment extracted with organic solvent and expose liquid form) OTHER TOXICITY DATA ALSO NEEDED	(Sediments with water, mixed, settled and exposed water species UNACCEPTABLE

12. What type of toxicity test was conducted? Length of test? _______
Static ___ (Water, sed-no change)
Static Renewal ___ (Water, sed-some water change)
Flow-Through ___ (Water, sed-water flowing through)

13*. Are appropriate analytical procedures used to determine total concentrations of the analytes in bulk sediment samples? What method(s) were used?
(Metals: partial digestion, analysis of elutriates or extracts are unacceptable.)

14*. Are measured dry weight contaminant concentrations reported? Conversion from wet weight to dry weight concentration may occur ONLY if data on moisture or TOC are provided. Nominal concentrations are unacceptable.
NO ___ UNACCEPTABLE YES ___ Page reference(s):

15*. Is the equilibrium adjustment period (i.e., time between spiking and initiation of the biological test) reported (24-h min. for metals; ~1-wk min. for organics)?
NO ___ UNACCEPTABLE YES ___ List details:

II BIOEFFECTS

16*.

a Do toxicity tests employ appropriate laboratory procedures? (ASTM: E1367, E1611, E1706).
NO ___ UNACCEPTABLE YES____

b Have the following been recorded during testing?
Temperature ____; pH ____; Hardness ____; Conductivity ____; Salinity ____; DO __

c Does DO Remain above 60% __ Needed for Marine
40% __ Needed for Freshwater
NO __ UNACCEPTABLE

d Temperature ___

e Is temperature within natural range, fluctuate less then 3°C, and have a time weighted average within 1°C of selected temp?
YES ___ NO ___ UNACCEPTABLE Range ____

f Do hardness, alkalinity, pH, or ammonia vary more than 50% (for freshwater samples)?
NO ___ YES ___ UNACCEPTABLE
Range: DO ___________; Alk ___________; pH ___________; NH_3 __________

g Is Salinity within species tolerance (for marine samples)?
YES ___ NO ___ UNACCEPTABLE Range ____

h List procedure reference(s) or brief details:

17*. Were biological responses compared to the control ___ or reference ___ sites?
List the Control and Reference sites? Positive Control _____
reference = uncontaminated site within the same waterbody or watershed; control = uncontaminated site outside the tested water body

18*.

a Have sediment samples used for biological testing been frozen?
NO ___ YES ___ If yes both biological and chemical testing must be performed after thawing sediments.

b Are appropriate procedures used for collecting, handling, and storage of sediments?
NO ___ YES ___ List procedures reference(s) or brief details:

19. Identify species used in toxicity testing. Source of Species?

20. What life stage were the test species at the start of the test?
(*Hyalella azteca* 7-14 day old; *Chironomus tentans* third-instar larvae; *Chironomus riparius* second instar or younger; *Daphnia magna* 5 days old; *Ceriodaphnia dubia* <24h old; *Hexagenia* spp. 3-4 months old; *Tubifex tubifex* Adult; *Diporeia* spp. juveniles)

____

21. Organism acclimation time ________________________.

____

22*. What percentage of the control survived?
Mean range _____ 70% for (*Chironomus riparius*, *Chironomus tentans*)
80% for (*Hexagenia* spp., *Daphnia magna*, *Ceriodaphnia dubia*, *Hyalella azteca*)
90% for (*Diporeia* spp., *Tubifex tubifex*, *Polychaetous* annelids, marine amphipods, others)
NO ___ UNACCEPTABLE

____

23*. Reference samples
Survival ____ %
Conc. less than TEL and ERLs? YES ___ NO __
Grain size __________; % sand ___; % silt ___; % clay __

____

24. BENTHIC COMMUNITY ANALYSIS
a Is there a benthic community abundance analysis?
NO ___ YES ___ List taxa (e.g., amphipod, sponges...) upon which the analysis focuses:

____

b* Do each of the sites within a sampling area have the same general characteristics (i.e., same depth of overlying water, same salinity in overlying water, etc.)?
NO ___ UNACCEPTABLE YES ___ Briefly list details:

____

25. Are end points (e.g., effects on embryonic development, early survival growth, reproduction, adult survival, biomass, density, diversity, avoidance, lesions...) reported?
NO __
YES ___ List endpoint(s):

____

III STATISTICAL ANALYSIS

26. Are appropriate statistical procedures reported?
NO ___ YES ___ List procedure reference(s):

____

Additional Notes/Comments:

APPENDIX G

SUMMARY TABLES OF THE AVAILABLE DATA ON BIOLOGICAL EFFECTS ASSOCIATED WITH SEDIMENT-SORBED CHEMICALS IN THE ST. LOUIS RIVER AREA OF CONCERN

APPENDIX G

SUMMARY TABLES OF THE AVAILABLE DATA ON BIOLOGICAL EFFECTS ASSOCIATED WITH SEDIMENT-SORBED CHEMICALS IN THE ST. LOUIS RIVER AREA OF CONCERN

Twenty-one tables have been assembled to provide a summary of the available data on the biological effects associated with the following sediment-sorbed contaminants in the St. Louis River AOC:

- arsenic
- cadmium
- chromium
- lead
- copper
- mercury
- zinc
- nickel
- total PCBs
- total PAHs
- benz(a)anthracene
- benzo(a)pyrene
- chrysene
- fluoranthene
- pyrene
- anthracene
- fluorene
- phenanthrene
- naphthalene

In addition, tables on the biological effects associated with sediment-sorbed toxic equivalents (TEQs) for wildlife and human health, in the St. Louis River AOC, are also included. Due to the large size of these tables (approximately 250 pages), they will only be provided in PDF format on the MPCA's Contaminated Sediments Web page (under the listing for this project) at: http://www.pca.state.mn.us/water/sediments/studies-stlouis.html#assessment.

Please note that for the Crane *et al.* (1997) study, the TOC data are missing from the summary tables. The TOC data have recently been added to the database. In addition, seven samples from the Interlake/Duluth Tar Superfund site (i.e., Sites C-1-102, C-2-101, C-2-102, C-2-104, SB-65, SB-66, and C-11; IT Corp. 1997) were eliminated from the database shortly before this report was finalized; the appropriate report tables and text were revised to reflect these changes. A member of the MPCA's Interlake/Duluth Tar Superfund site team determined that sediment samples from the aforementioned sites were not collected at the same time for sediment chemistry and sediment toxicity tests. These changes were not made in the following tables due to time constraints with re-creating all twenty-one tables.

The MPCA has obtained a new GLNPO grant (effective October 1, 2000) to expand the matching sediment chemistry/toxicity database for the St. Louis River AOC. This new GIS-based database will include all sediment chemistry and bioeffects data for selected studies. Contact Judy Crane (MPCA) for additional information.

APPENDIX H

CANDIDATE APPROACHES TO THE DEVELOPMENT OF NUMERICAL SEDIMENT QUALITY TARGETS FOR POTENTIAL CHEMICALS OF CONCERN IN THE ST. LOUIS RIVER AREA OF CONCERN

APPENDIX H

CANDIDATE APPROACHES TO THE DEVELOPMENT OF NUMERICAL SEDIMENT QUALITY TARGETS FOR POTENTIAL CHEMICALS OF CONCERN IN THE ST. LOUIS RIVER AREA OF CONCERN

Numerical sediment quality guidelines (including sediment quality guidelines, sediment quality criteria, sediment quality objectives, and sediment quality standards) have been developed by various jurisdictions in North America for both freshwater and marine ecosystems. The SQGs that are currently being used in North America have been developed using a variety of empirical and theoretical approaches, including:

- Screening Level Concentration (SLC) Approach;
- Effects Range (ER) Approach;
- Effects Level (EL) Approach;
- Apparent Effects Threshold (AET) Approach;
- Equilibrium Partitioning (EqP) Approach;
- Logistic Regression Modeling (LRM) Approach; and,
- Consensus Approach.

The aforementioned approaches were considered to support the derivation of numerical SQTs for the protection of sediment-dwelling organisms in the St. Louis River AOC. The tissue residue approach was considered to be the primary method for deriving numerical SQTs for the protection of wildlife and human health (i.e., for substances that bioaccumulate in the food web). The following sections provide brief descriptions of each of these approaches. Table H-1 provides a summary of these approaches.

H.1 SCREENING LEVEL CONCENTRATION APPROACH

The screening level concentration (SLC) approach is a biological effects-based approach that is applicable to the development of SQGs for the protection of benthic organisms. This approach utilizes matching biological and chemistry data collected in field surveys to calculate a screening level concentration (SLC) (Neff *et al.* 1986). 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 concentrations of specific contaminants in sediments and on the co-occurrence of benthic organisms with varying contaminant levels. 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 then compiled as a frequency distribution to determine the concentration that can be tolerated by a specific proportion of the species (i.e., the 5th percentile of the distribution would provide a SLC that should be tolerated by 95% of the species; see Figure 1 in Persaud *et al.* 1993 for a graphic example of this frequency distribution). This concentration is termed the screening level concentration of the contaminant.

A number of jurisdictions have used the SLC approach to derive numerical SQGs. In the St. Lawrence River, two types of SQGs were developed for five groups of PCBs using the SLC approach, including a minimal effect threshold (MET) and a toxic effect threshold (TET) (EC and MENVIQ 1992). The MET was calculated as the 15th percentile of the SSLCs, while the TET was calculated as the 90th percentile of the SSLC distribution for each substance. Therefore, the MET and TET are considered to provide protection for 85% and 10% of the species represented in the database, respectively. Similarly, the Ontario Ministry of Environment and Energy has developed a lowest effect level (LEL) and severe effect level (SEL) for each of five groups of PCBs, hepoxide, and γ-BHC using this approach (Persaud *et al.* 1993). Neff *et al.* (1986) also developed a screening level concentration (SLC) for total PCBs primarily using data from the Great Lakes.

H.2 EFFECTS RANGE APPROACH

The effects range (ER) approach to the derivation of SQGs was developed to provide an informal tool for assessing the potential of various contaminants tested in the National Status and Trends Program (NSTP) to be associated with adverse effects on sediment-dwelling organisms (Long and Morgan 1991). As a first step, a database was compiled which contained information on the effects of sediment-associated contaminants, including data from spiked-sediment toxicity tests, matching sediment chemistry and biological effects data from field studies in the United States, and SQGs that were derived using various approaches. All of the information in the database was weighted equally, regardless of the method that was used to develop it. The objective of this initiative was to identify informal guidelines which could be used to evaluate sediment chemistry data collected nationwide under the NSTP.

Candidate data sets from field studies were evaluated to determine their applicability for incorporation into the database (MacDonald *et al.* 1996). This evaluation was designed to determine the overall applicability of the data set, the methods that were used, the end-points that were measured, and the degree of concordance between the chemical and biological data. The data which met the evaluation criteria were incorporated into the database.

The database that was compiled included several types of information from each study. Individual entries consisted of the concentration of the contaminant, the location of the study, the species tested and endpoint measured, 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 in association with elevated sediment chemistry). Data from nontoxic or unaffected samples were assumed to represent background conditions. Data which showed no concordance between chemical and biological variables were included in the database, but were not used to calculate the SQGs. The data for which a biological effect was observed in association with elevated chemical concentrations (i.e., hits) were sorted in ascending order of concentration and the 10^{th} and 50^{th} percentile concentrations for each compound were determined. The effects range-low (ERL; 10^{th} percentile value) was considered to represent a lower threshold value, below which adverse effects on sensitive life stages and/or species occurred infrequently. The effects range-median (ERM; 50^{th} percentile value) was considered to represent a second threshold value, above which adverse effects were frequently observed. These two parameters, ERL and ERM, were then used as informal SQGs (Long and Morgan 1991; Long *et al.* 1995). These SQGs were subsequently evaluated to determine their ability to correctly classify sediments as toxic and non-toxic (Long *et al.* 1998a,b). The U.S. EPA (1996) used a similar approach to derive ERLs (15^{th} percentile of the effects data set) and ERMs (50^{th} percentile of the effects data set) for assessing sediments from various freshwater locations. Similarly, MacDonald (1997b) applied the ER approach to regionally collected field data to derive site-specific sediment effect concentrations (SECs) for PCBs and DDTs in the Southern California Bight.

H.3 EFFECTS LEVEL APPROACH

The effects level (EL) approach is closely related to the effects range approach. However, the EL approach is supported by an expanded version of the database that was used to derive the effects levels (Long and Morgan 1991). The expanded database contains matching sediment chemistry and biological effects data from spiked-sediment toxicity tests and from field studies conducted throughout North America (including both effects and no effects data). The expanded database also contains SQGs that were derived using various approaches. The information contained in the expanded database was evaluated and classified in the same manner that was used to compile the original NSTP database.

In the EL approach, the underlying information in the database was used to derive two types of SQGs, including threshold effect levels (TELs) and probable effect levels (PELs). The TEL, which is calculated as the geometric mean of the 15th percentile of the effects data set and the 50th percentile of the no effects data set, represents the chemical concentration below which adverse effects occurred infrequently. The PEL is calculated as the geometric mean of the 50th percentile of the effects data set and the 85th percentile of the no effects data set. The PEL represents a threshold value above which adverse effects were frequently observed. These arithmetic procedures have been applied to the expanded database to derive numerical SQGs (i.e., TELs and PELs) for Florida coastal waters (MacDonald *et al.* 1996), U.S. freshwater systems (Ingersoll *et al.* 1996), and Canadian freshwater and marine systems (Smith *et al.* 1996).

H.4 APPARENT EFFECTS THRESHOLD APPROACH

The Apparent Effects Threshold (AET) approach to the development of SQGs was developed for use in the Puget Sound area of Washington State (Tetra Tech Inc. 1986). The AET approach is based on empirically defined relationships between measured concentrations of a contaminant in sediments and observed biological effects. This approach is intended 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 tests), changes in the abundance of various benthic species, and changes in benthic community structure. In Puget Sound, for example, four marine AET values have been generated, including AETs for Microtox®, oyster larvae, benthic communities, and amphipods. The AET values are based on dry weight-normalized contaminant concentrations for metals and either dry weight- or total organic carbon-normalized concentrations for organic substances (Barrick *et al*. 1988; Washington Department of Ecology 1990). The state of Washington has used the various AET values to establish sediment quality standards and minimum clean-up levels for contaminants of concern in the state.

Cubbage *et al.* (1997) refined this approach to support the development of probable AETs (PAETs) using matching sediment chemistry and toxicity data for freshwater sediments from the state of Washington. Ingersoll *et al.* (1996; USEPA 1996) utilized a similar approach to develop freshwater AETs (termed no effect concentrations or NECs in that study) using data from various freshwater locations.

H.5 EQUILIBRIUM PARTITIONING APPROACH

The water-sediment equilibrium partitioning (EqP) approach has been one of the most studied and evaluated approaches for developing SQGs for non-polar organic chemicals and metal mixtures (Pavlou and Weston 1983; Bolton *et al*. 1985; Kadeg *et al*. 1986; Pavlou 1987; Di Toro *et al.* 1991; Hansen *et al.* 1996). This approach is based on the premise that the distribution of contaminants is predictable among different compartments in the sediment matrix (i.e., sediment solids and interstitial water) based on their physical and chemical properties, assuming that continuous equilibrium exchange between sediment and interstitial water occurs. This approach has been supported by the results of spiked-sediment toxicity tests, which indicate that positive correlations exist between the biological effects observed and the concentrations of contaminants measured in the interstitial water (Di Toro *et al.* 1991; Berry *et al.* 1996; Hansen *et al.* 1996).

In the EqP approach, water quality criteria developed for the protection of freshwater or marine organisms are used to support the SQG 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). The SQGs are calculated using the appropriate water quality criteria [usually the final chronic values (FCVs) or equivalent values; USEPA 1997b] in conjunction with the sediment/water partition coefficients (K_p) for the specific contaminants. The FCV is derived from the species mean chronic value that has been calculated from published toxicity data and is intended to protect 95% of aquatic species. The calculation procedure for nonionic organic contaminants is as follows:

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

where:

SQG = Sediment quality guideline (in µg/kg);

K_p = Partition coefficient for the chemical (in L/kg); and,
FCV = Final chronic value (in µg/L).

The K_p is a function of the partition coefficient for sediment organic carbon (K_{oc}) of the substance under consideration and the amount of organic carbon in the sediment (f_{oc}) under investigation (where $K_p = K_{oc} \bullet f_{oc}$; Di Toro *et al.* 1991). The K_{oc} for non-ionic substances can be calculated from its octanol-water partition coefficient (K_{ow}) (Di Toro *et al.* 1991).

The U.S. EPA has prepared draft equilibrium partitioning sediment guidelines (ESGs) for endrin dieldrin, and metal mixtures (cadmium, copper, lead, nickel, silver, and zinc) (Scott Ireland, U.S. EPA, personal communication, 2000). The U.S. EPA is also preparing a draft ESG for PAH mixtures. Draft U.S. EPA documents which provide guidance on the technical basis for deriving ESGs for nonionic organics for the protection of benthic organisms, as well as methods for deriving site-specific ESGs, have also been prepared (Scott Ireland, U.S. EPA, personal communication, 2000). Additional information about the aforementioned reports is available at: http://www.epa.gov/waterscience/pc/csnews/issue25/.

H.6 LOGISTIC REGRESSION MODELING APPROACH

In the logistic regression modeling (LRM) approach, numerical SQGs are derived from the results of field studies of sediment quality conditions. The first step in this process involves the collection, evaluation, and compilation of matching sediment chemistry and toxicity data from a wide variety of sites in North America. Next, the information that is compiled in the database is retrieved on a substance-by-substance basis, with the data from individual sediment samples sorted in order of ascending concentration. For each sediment sample, the ascending data table provides information on the concentration of the contaminant under consideration (on either a dry weight- or organic carbon-normalized basis) and the toxicity test results (i.e., toxic or not toxic) for each toxicity test endpoint (e.g., 10-day survival of amphipods).

In the next step of the process, the data contained in the ascending data tables are screened to minimize the inclusion of samples in which the selected contaminant did not contribute substantially to the observed toxicity. In this analysis, the chemical concentration in each toxic sample is compared to the mean concentration in the non-toxic samples from the same study and geographic area. The toxic samples with concentrations of the selected contaminant that were less than or equal to the average concentration of that chemical in the non-toxic samples were not used in further analyses of the data (i.e., it was highly unlikely that the contaminant substantially contributed to sediment toxicity in such samples).

In the final step of the analysis, the screened data are used to develop logistic regression models, which express the relationship between the concentration of the selected contaminant and the probability of observing toxicity. In its simplest form, the logistic model can be described using the following equation:

$$p = e^{B0 + B1(x)} \div (1 + e^{B0 + B1(x)})$$

where: p = probability of observing a toxic effect;
B0 = intercept parameter;
B1 = slope parameter; and,
x = concentration or log concentration of the chemical.

Using a preliminary database consisting of the results of 10-day marine amphipod toxicity tests, Field *et al.* (1999) derived logistic regression models for seven chemical substances to illustrate the methodology. More specifically, these investigators calculated T10, T50, and T90 values for four metals, two PAHs, and total PCBs. These values represent the chemical concentrations that correspond to a 10%, 50%, and 90% probability of observing sediment toxicity. In addition to supporting the derivation of specific T-values, this method can be used to determine the concentration of a contaminant that corresponds to any probability of observing toxicity. Therefore, a sediment manager can identify an acceptable probability of observing sediment toxicity at a site (e.g., 25%) and determine the corresponding chemical concentrations (e.g., T25 values). The calculated value can then be used as the SQG for the site. The LRM approach is data intensive and has primarily been applied to marine data sets. Limited freshwater data make this approach difficult to develop at this time for freshwater sediments.

H.7 CONSENSUS APPROACH

In the consensus approach, consensus-based SQGs are derived from the existing SQGs that have been published for the protection of sediment-dwelling organisms. Derivation of numerical SQGs using the consensus approach involves a four-step process. First, the SQGs that have been derived by various investigators for assessing the quality of freshwater sediments are collected and collated. Next, the SQGs obtained from all sources are evaluated to determine their applicability to the derivation of consensus-based SQGs. The selection criteria that are applied are intended to evaluate the understandability of the derivation methods, the degree to which the SQGs are effects-based, and the uniqueness of the SQGs.

The effects-based SQGs that meet these selection criteria are then grouped to facilitate the derivation of consensus-based SQGs (Swartz 1999). Specifically, the SQGs for the protection of sediment-dwelling organisms are grouped into two categories according to their original narrative intent, including threshold effect concentrations (TECs) and probable effect concentrations (PECs) (Table H-1). The TECs are those SQGs intended to identify contaminant concentrations below which harmful effects on sediment-dwelling organisms were unlikely to be observed. Examples of TECs include threshold effect levels (TELs) (Smith *et al.* 1996; USEPA 1996), effect range low values (ERLs) (Long and Morgan 1991; USEPA 1996), and lowest effect levels (LELs) (Persaud *et al.* 1993) (Table H-2). The PECs are those SQGs intended to identify contaminant concentrations above which harmful effects on sediment-dwelling organisms were likely to be frequently or always observed (MacDonald *et al.* 1996; Swartz 1999). Examples of PECs include probable effect levels (PELs) (Smith *et al.* 1996; USEPA 1996), effect range median values (ERMs) (Long and Morgan 1991; USEPA 1996); and severe effect levels (SELs) (Persaud *et al.* 1993) (Table H-3).

Following classification of the published SQGs, consensus-based TECs are calculated by determining the geometric mean of the SQGs that are included in this category. Likewise, consensus-based PECs are calculated by determining the geometric mean of the PEC-type values. The geometric mean, rather than the arithmetic mean, is calculated because it provides an estimate of central tendency that is not unduly affected by outliers and because the distributions of the SQGs were not known. Consensus-based TECs or PECs are calculated only if three of more published SQGs are available for a chemical substance or group of substances.

The consensus approach has been used to derive numerical SQGs for a variety of chemical substances and media types. For example, Swartz (1999) derived consensus-based SQGs for PAHs in marine ecosystems. More recently, MacDonald *et al.* (2000b) derived SQGs for total PCBs in freshwater and marine sediments. Ingersoll and MacDonald (1999) and MacDonald *et al.* (2000a) have also developed consensus-based SQGs for metals, PAHs, PCBs, and several pesticides in freshwater sediments. As the term implies, consensus-based SQGs are intended to reflect the agreement among the various SQGs by providing an estimate of their central tendency. Consensus-based SQGs are, therefore, considered to provide a unifying synthesis of the existing SQGs and to account for the effects of contaminant mixtures in sediment (Swartz 1999; MacDonald *et al.* 2000a,b). In addition, the consensus-based PECs are generally comparable to the EqP-based SQGs (MacDonald *et al.* 2000a). Therefore, the consensus-based PECs also define concentrations of sediment-associated contaminants that are sufficient to cause or substantially contribute to sediment toxicity (MacDonald *et al.* 2000a).

H.8 TISSUE RESIDUE APPROACH

The tissue residue (TR) approach (which is also known as the biota-water-sediment equilibrium partitioning approach) is premised on the fact that sediments represent important sources of bioaccumulative contaminants in aquatic food webs. For this reason, it is necessary to assure that the concentrations of sediment-associated contaminants remain below the levels that are associated with the bioaccumulation of such contaminants to harmful levels in the food web. Therefore, application of the tissue residue approach involves the establishment of safe sediment concentrations for individual chemicals or classes of chemicals by determining the chemical concentrations in sediments that are predicted to result in acceptable tissue residues.

Derivation of numerical SQGs using the TR approach involves several steps. As a first step, the contaminants for which SQGs are to be derived are selected based on their potential to accumulate in aquatic food webs. Next, numerical tissue residue guidelines (TRGs) are identified for these contaminants. While most of the available TRGs are intended to provide protection for human health, it is also important to obtain TRGs that are explicitly designed to protect piscivorus wildlife species. Following the selection of TRGs, biota-to-sediment accumulation factors (BSAFs) are determined for each of the substances of concern. Such BSAFs can be determined from the results of bioaccumulation assessments, from matching sediment chemistry and tissue residue data, or from the results of bioaccumulation models. Numerical SQGs are subsequently derived using the equation:

$$\mathbf{SQG} = \mathbf{TRG} \div \mathbf{BSAF}$$

This approach has been used on several occasions to develop SQGs for the protection of human health (most notably for DDT, Hg, and PCBs) (CCME 1999). In addition, sediment contamination limits for 2,3,7,8 tetrachlorodibenzo-p-dioxin (TCDD) have been established for Lake Ontario on the basis of fish tissue residues (Cook *et al*. 1989; Endicott *et al*. 1989). The applicability of this approach to the derivation of SQGs is supported by data which demonstrate that declines in DDT residues in fish and birds (since its use was banned) are strongly correlated with declining concentrations of this substance in surficial sediments in the Great Lakes and

Table H-1. Descriptions of the Published Freshwater SQGs that have been Developed Using Various Approaches

Type of SQG	Acronym	Approach	Description	Reference
Threshold Effect Concentration – SQGs				
Lowest Effect Level	LEL	SLC	Sediments are considered to be clean to marginally polluted. No effects on the majority of sediment-dwelling organisms are expected below this concentration.	Persaud *et al.* (1993)
Threshold Effect Level	TEL	WEA	Represents the concentration below which adverse effects are expected to occur only rarely.	Smith *et al*. (1996)
Effects Range - Low	ERL	WEA	Represents the chemical concentration below which adverse effects would be rarely observed.	Long and Morgan (1991)
Threshold Effect Level for *Hyalella azteca* in 28-day tests	TEL-HA28	WEA	Represents the concentration below which adverse effects on survival or growth of the amphipod, Hyalella azteca, are expected to occur only rarely (in 28-day tests).	USEPA (1996); Ingersoll *et al.* (1996)
Minimal Effect Threshold	MET	SLC	Sediments are considered to be clean to marginally polluted. No effects on the majority of sediment-dwelling organisms are expected below this concentration.	EC and MENVIQ (1992)
Chronic Equilibrium Partitioning Threshold	SQAL	EqP	Represents the concentration in sediments that is predicted to be associated with concentrations in the interstitial water below a chronic water quality criterion. Adverse effects on sediment-dwelling organisms are predicted to occur only rarely below this concentration	Bolton *et al*. (1985); Zarba (1992); USEPA (1997a)

Table H-1. Continued

Type of SQG	Acronym	Approach	Description	Reference
Probable Effect Concentration – SQGs				
Severe Effect Level	SEL	SLC	Sediments are considered to be heavily polluted. Adverse effects on the majority of sediment-dwelling organisms are expected when this concentration is exceeded.	Persaud *et al.* (1993)
Probable Effects Level	PEL	WEA	Represents the concentration above which adverse effects are expected to occur frequently.	Smith *et al*. (1996)
Effects Range - Median	ERM	WEA	Represents the chemical concentration above which adverse effects would frequently occur.	Long and Morgan (1991)
Probable Effects Level for *Hyalella azteca* in 28-day tests	PEL-HA28	WEA	Represents the concentration above which adverse effects on survival or growth of the amphipod, Hyalella azteca, are expected to occur frequently (in 28-day tests).	USEPA (1996); Ingersoll *et al*. (1996)
Toxic Effect Threshold	TET	SLC	Sediments are considered to be heavily polluted. Adverse effects on sediment-dwelling organisms are expected when this concentration is exceeded.	EC and MENVIQ (1992)

SQGs = sediment quality guidelines; SLC = screening level concentration; WEA = weight of evidence approach; EqP = equilibrium partitioning.

Table H-2. Sediment Quality Guidelines for Freshwater Ecosystems that Represent Threshold Effect Concentrations (i.e., below which harmful effects are unlikely to be observed) (MacDonald *et al*. 2000a)

	Threshold Effect Concentrations						
Substance	**TEL**	**LEL**	**MET**	**ERL**	**TEL-HA28**	**SQAL**	**Consensus-Based TEC**
Metals (in mg/kg DW)							
Arsenic	5.9	6	7	33	11	NG	9.79
Cadmium	0.596	0.6	0.9	5	0.58	NG	0.99
Chromium	37.3	26	55	80	36	NG	43.4
Copper	35.7	16	28	70	28	NG	31.6
Lead	35	31	42	35	37	NG	35.8
Mercury	0.174	0.2	0.2	0.15	NG	NG	0.18
Nickel	18	16	35	30	20	NG	22.7
Zinc	123	120	150	120	98	NG	121
PAHs (in µg/kg DW)							
Anthracene	NG	220	NG	85	10	NG	57.2
Fluorene	NG	190	NG	35	10	540	77.4
Naphthalene	NG	NG	400	340	15	470	176
Phenanthrene	41.9	560	400	225	19	1800	204
Benz[a]anthracene	31.7	320	400	230	16	NG	108
Benzo(a)pyrene	31.9	370	500	400	32	NG	150
Chrysene	57.1	340	600	400	27	NG	166
Dibenz[a,h]anthracene	NG	60	NG	60	10	NG	33.0
Fluoranthene	111	750	600	600	31	6200	423
Pyrene	53	490	700	350	44	NG	195
Total PAHs	NG	4000	NG	4000	260	NG	1610

Table H-2. Continued

Substance	*Threshold Effect Concentrations* TEL	LEL	MET	ERL	TEL-HA28	SQAL	Consensus-Based TEC
PCBs (in µg/kg DW)							
Total PCBs	34.1	70	200	50	32	NG	59.8
Organochlorine Pesticides (in µg/kg DW)							
Chlordane	4.5	7	7	0.5	NG	NG	3.24
Dieldrin	2.85	2	2	0.02	NG	110	1.90
Sum DDD	3.54	8	10	2	NG	NG	4.88
Sum DDE	1.42	5	7	2	NG	NG	3.16
Sum DDT	NG	8	9	1	NG	NG	4.16
Total DDTs	7	7	NG	3	NG	NG	5.28
Endrin	2.67	3	8	0.02	NG	42	2.22
Heptachlor epoxide	0.6	5	5	NG	NG	NG	2.47
Lindane (gamma-BHC)	0.94	3	3	NG	NG	3.7	2.37

TEL = threshold effect level, dry weight (Smith *et al.* 1996); LEL = lowest effect level, dry weight (Persaud *et al.* 1993); MET = minimal effect threshold, dry weight (EC and MENVIQ 1992); ERL = effects range low, dry weight (Long and Morgan 1991); TEL-HA28 = threshold effect level for *Hyalella azteca*, 28-day test, dry weight (USEPA 1996); SQAL = sediment quality advisory levels, dry weight at 1% OC (USEPA 1997a); NG = no guideline.

Table H-3. Sediment Quality Guidelines for Freshwater Ecosystems that Represent Probable Effect Concentrations (i.e., above which harmful effects are likely to be observed) (MacDonald *et al.* 2000a)

Substance	*Probable Effect Concentrations*					
	PEL	**SEL**	**TET**	**ERM**	**PEL-HA28**	**Consensus-Based PEC**
Metals (in mg/kg DW)						
Arsenic	17	33	17	85	48	33.0
Cadmium	3.53	10	3	9	3.2	4.98
Chromium	90	110	100	145	120	111
Copper	197	110	86	390	100	149
Lead	91.3	250	170	110	82	128
Mercury	0.486	2	1	1.3	NG	1.06
Nickel	36	75	61	50	33	48.6
Zinc	315	820	540	270	540	459
PAHs (in µg/kg DW)						
Anthracene	NG	3700	NG	960	170	845
Fluorene	NG	1600	NG	640	150	536
Naphthalene	NG	NG	600	2100	140	561
Phenanthrene	515	9500	800	1380	410	1170
Benz[a]anthracene	385	14800	500	1600	280	1050
Benzo(a)pyrene	782	14400	700	2500	320	1450
Chrysene	862	4600	800	2800	410	1290
Fluoranthene	2355	10200	2000	3600	320	2230
Pyrene	875	8500	1000	2200	490	1520
Total PAHs	NG	100000	NG	35000	3400	22800

Table H-3. Continued

Substance	*Probable Effect Concentrations*					
	PEL	**SEL**	**TET**	**ERM**	**PEL-HA28**	**Consensus-Based PEC**
PCBs (in µg/kg DW)						
Total PCBs	277	5300	1000	400	240	676
Organochlorine Pesticides (in µg/kg DW)						
Chlordane	8.9	60	30	6	NG	17.6
Dieldrin	6.67	910	300	8	NG	61.8
Sum DDD	8.51	60	60	20	NG	28.0
Sum DDE	6.75	190	50	15	NG	31.3
Sum DDT	NG	710	50	7	NG	62.9
Total DDTs	4450	120	NG	350	NG	572
Endrin	62.4	1300	500	45	NG	207
Heptachlor Epoxide	2.74	50	30	NG	NG	16.0
Lindane (gamma-BHC)	1.38	10	9	NG	NG	4.99

PEL = probable effect level, dry weight (Smith *et al.* 1996); SEL = severe effect level, dry weight (Persaud *et al.* 1993); TET = toxic effect threshold, dry weight (EC and MENVIQ 1992); ERM = effects range median, dry weight (Long and Morgan 1991); PEL-HA28 = probable effect level for *Hyalella azteca*, 28-day test, dry weight (USEPA 1996); NG = no guideline.

APPENDIX I

PREDICTIVE ABILITY TABLES

Table I-1. Predictive Ability of the Consensus-based SQGs in Freshwater Sediments for Amphipod Tests from the St. Louis River AOC, Great Lakes Region, and non-Great Lakes Sites in North America (USEPA 2000a)

	Incidence of Toxicity: 10-14 day Amphipod Tests			
Mean PEC-Q Range	**St. Louis River AOC**	**Other Great Lakes Sites**	**All Great Lakes Sites**	**Non-Great Lakes Sites**
≤0.1	6.8% (3 of 44 samples)	50% (3 of 6 samples)	12% (6 of 50 samples)	22% (20 of 90 samples)
>0.1 to ≤0.5	11% (9 of 80 samples)	25% (14 of 56 samples)	17% (23 of 136 samples)	16% (23 of 142 samples)
>0.5 to ≤1.0	30% (3 of 10 samples)	52% (14 of 27 samples)	46% (17 of 37 samples)	26% (9 of 35 samples)
>1.0 to ≤5.0	27% (3 of 11 samples)	68% (25 of 37 samples)	58% (28 of 48 samples)	19% (8 of 42 samples)
>5.0	75% (3 of 4 samples)	77% (23 of 30 samples)	76% (26 of 34 samples)	64% (21 of 33 samples)
Overall	14% (21 of 149 samples)	51% (79 of 156 samples)	33% (100 of 305 samples)	24% (81 of 342 samples)

SQGs = sediment quality guidelines; PEC-Q = probable effect concentration quotient; AOC = Area of Concern.
For 10-14 day amphipod tests, the survival and growth of *Hyalella azteca* were measured.
Sites 102-TR and 044-TR, from the R-EMAP study (Breneman *et al*. 2000), were removed from the incidence of toxicity calculations due to incomplete sediment chemistry data (i.e., PAHs, PCBs) for these known contaminated areas.

Table I-2. Predictive Ability of the Consensus-based SQGs in Freshwater Sediments for Amphipod Tests from the St. Louis River AOC and North America (USEPA 2000a)

	Incidence of Toxicity			
Mean PEC-Q Range	**10-day Amphipod Tests St. Louis River AOC**	**10-14 day Amphipod Tests Other North American Sites**	**10-14 day Amphipod Tests All North American Sites**	**28-42 day Amphipod Tests All North American Sites**
≤0.1	6.8% (3 of 44 samples)	24% (23 of 96 samples)	18% (26 of 140 samples)	10% (6 of 63 samples)
>0.1 to ≤0.5	11% (9 of 80 samples)	19% (37 of 198 samples)	16% (46 of 278 samples)	17% (5 of 30 samples)
>0.5 to ≤1.0	30% (3 of 10 samples)	37% (23 of 62 samples)	36% (26 of 72 samples)	56% (15 of 27 samples)
>1.0 to ≤5.0	27% (3 of 11 samples)	42% (33 of 79 samples)	40% (36 of 90 samples)	96% (24 of 25 samples)
>5.0	75% (3 of 4 samples)	70% (44 of 63 samples)	70% (47 of 67 samples)	100% (6 of 6 samples)
Overall	14% (21 of 149 samples)	32% (160 of 498 samples)	28% (181 of 647 samples)	37% (56 of 151 samples)

SQGs = sediment quality guidelines; PEC-Q = probable effect concentration quotient; AOC = Area of Concern.
For 10-14 day amphipod tests, the survival and growth of *Hyalella azteca* were measured.
For 28-42 day amphipod tests, the survival, growth and reproduction of *Hyalella azteca* were measured.
Sites 102-TR and 044-TR, from the R-EMAP study (Breneman *et al*. 2000), were removed from the incidence of toxicity calculations due to incomplete sediment chemistry data (i.e., PAHs, PCBs) for these known contaminated areas.

Table I-3. Predictive Ability of the Consensus-based SQGs in Freshwater Sediments for Midge Tests from the St. Louis River AOC, Great Lakes Region, and non-Great Lakes Sites in North America (USEPA 2000a)

Mean PEC-Q Range	**Incidence of Toxicity: 10-14 day Midge Tests**			
	St. Louis River AOC	**Other Great Lakes Sites**	**All Great Lakes Sites**	**Non-Great Lakes Sites**
≤0.1	6.5% (3 of 46 samples)	19% (10 of 52 samples)	13% (13 of 98 samples)	50% (10 of 20 samples)
>0.1 to ≤0.5	12% (9 of 74 samples)	23% (31 of 135 samples)	19% (40 of 209 samples)	12% (10 of 87 samples)
>0.5 to ≤1.0	20% (2 of 10 samples)	59% (20 of 34 samples)	50% (22 of 44 samples)	22% (4 of 18 samples)
>1.0 to ≤5.0	36% (4 of 11 samples)	47% (27 of 57 samples)	46% (31 of 68 samples)	28% (5 of 18 samples)
>5.0	100% (5 of 5 samples)	63% (19 of 30 samples)	68% (24 of 35 samples)	50% (3 of 6 samples)
Overall	16% (23 of 146 samples)	35% (107 of 308 samples)	29% (130 of 454 samples)	22% (32 of 149 samples)

SQGs = sediment quality guidelines; PEC-Q = probable effect concentration quotient; AOC = Area of Concern.
For 10-14 day midge tests, the survival and growth of *Chironomus tentans* were measured.
Sites 102-TR and 044-TR, from the R-EMAP study (Breneman *et al*. 2000), were removed from the incidence of toxicity calculations due to incomplete sediment chemistry data (i.e., PAHs, PCBs) for these known contaminated areas.

Table I-4. Predictive Ability of the Consensus-based SQGs in Freshwater Sediments for Midge Tests from the St. Louis River AOC and North America (USEPA 2000a)

Mean PEC-Q Range	Incidence of Toxicity		
	10-day Midge Tests St. Louis River AOC	**10-14 day Midge Tests Other N. American Sites**	**10-14 day Midge Tests All North American Sites**
≤0.1	6.5% (3 of 46 samples)	28% (20 of 72 samples)	19% (23 of 118 samples)
>0.1 to ≤0.5	12% (9 of 74 samples)	18% (41 of 222 samples)	17% (50 of 296 samples)
>0.5 to ≤1.0	20% (2 of 10 samples)	46% (24 of 52 samples)	42% (26 of 62 samples)
>1.0 to ≤5.0	36% (4 of 11 samples)	43% (32 of 75 samples)	42% (36 of 86 samples)
>5.0	100% (5 of 5 samples)	61% (22 of 36 samples)	66% (27 of 41 samples)
Overall	16% (23 of 146 samples)	30% (139 of 457 samples)	27% (162 of 603 samples)

SQGs = sediment quality guidelines; PEC-Q = probable effect concentration quotient; AOC = Area of Concern.
For 10-14 day midge tests, the survival and growth of *Chironomus tentans* were measured.
Sites 102-TR and 044-TR, from the R-EMAP study (Breneman *et al*. 2000), were removed from the incidence of toxicity calculations due to incomplete sediment chemistry data (i.e., PAHs, PCBs) for these known contaminated areas.

APPENDIX J

AN INTEGRATED FRAMEWORK FOR ASSESSING SEDIMENT QUALITY CONDITIONS

APPENDIX J

AN INTEGRATED FRAMEWORK FOR ASSESSING SEDIMENT QUALITY CONDITIONS

J.1 OVERVIEW

The numerical SQTs recommended in Tables 14 and 15 provide a strong technical basis for assessing sediment quality in the St. Louis River AOC. These SQTs explicitly address the inherent uncertainty associated with the sediment quality assessment process by identifying ranges of contaminant concentrations instead of absolute values. In addition, the probability of observing adverse biological effects within these ranges has been determined to provide further guidance on the use of these management tools. Nonetheless, the possibility of arriving at erroneous conclusions still exists if the SQTs are used in isolation. For this reason, a framework has been developed to assist in the design and implementation of efficient and effective sediment quality assessments.

This framework is intended to provide preliminary guidance on the uses of SQTs and related tools in comprehensive sediment quality assessments. This framework identifies the steps that should be followed in conducting site-specific sediment quality assessment programs and is comprised of the following elements:

- evaluate sediment quality issues and concerns;
- collect, collate, and evaluate the adequacy of existing sediment quality data;
- collect additional sediment quality data, if needed;
- assess sediment chemistry data;
- assess sediment toxicity data;
- assess data on benthic invertebrate community structure;
- assess tissue chemistry data;
- identify actual and probable use impairments;
- develop and implement remediation measures, if necessary; and,
- conduct confirmatory monitoring and assessment.

The recommended framework is to provide general guidance to support the sediment quality assessment process. More detailed guidance on the assessment of contaminated sediments is currently being developed for GLNPO (MacDonald *et al.* In preparation).

J.2 EVALUATION OF SEDIMENT QUALITY ISSUES AND CONCERNS

The first phase of a site-specific sediment quality assessment involves the evaluation of sediment issues and concerns at the site under investigation. As a first step in this process, the pertinent historical information on the site under consideration is collected and reviewed. Information is required on the types of industries and businesses that operate or have operated in the area, on the location of wastewater treatment plants, on land use patterns in upland areas, on stormwater drainage systems, on residential developments, and on other historic, ongoing, and potential activities within the area. Any unique attributes of the site must also be taken into consideration. These data provide a basis for identifying potential sources of contaminants to aquatic ecosystems. Information on the chemical composition of wastewater effluent discharges, types of contaminants likely to be associated with non-point sources, and physical/chemical properties of those substances provides a basis for developing an initial list of PCOCs at the site. By evaluating the probable environmental fate of these PCOCs, it is possible to establish a list of COCs and area of potential concern at the site (Figure J-1).

In addition to information on contaminant sources, information should be collected that helps define environmental management goals at the site (if these have not been defined). Environmental management goals in freshwater systems may be based on protection of the ecosystem as a whole, maintenance of viable populations of sportfish species, protection of human health (e.g., swimmable and fishable), or a variety of other considerations (e.g., regional stormwater management, industrial development). As such, information on existing uses of the site provide a basis for making decisions regarding the nature and extent of the investigations that should be conducted at the site. MacDonald (1989), Baudo and Muntau (1990), and Mudroch and McKnight (1991) provide detailed descriptions of information that should be collected and how these data may be used to assess ambient environmental quality.

J.3 COLLECT AND EVALUATE EXISTING SEDIMENT QUALITY DATA

Collection and evaluation of existing sediment quality data is a critical component of the site-specific sediment quality assessment process. The types of data that should be collected and collated during this stage of the assessment include sediment chemistry data, tissue chemistry data, sediment toxicity data, and data on benthic invertebrate community structure. Any ancillary data on fish tumors, etc. would also be helpful. Because data may have been collected under a variety of programs and for a number of reasons, it is important that these data be evaluated to determine their applicability in the sediment quality assessment process. This evaluation should cover the overall quality of the data set and the degree to which the data are thought to represent current conditions at the site under consideration.

Concerns regarding data quality may be resolved by comparing the quality assurance/quality control (QA/QC) measures that were implemented during the collection, transport, and analysis of sediment samples to the data quality objectives (DQOs) identified in the quality assurance project plan (QAPP). A number of conventions have now been established which provide guidance on the field aspects of sediment sampling programs (ASTM 1990; USEPA and USACOE 1998). While a diversity of analytical procedures have been developed to quantify concentrations of contaminants in sediments, acceptable methods have been reported, for example in the Inland Testing Manual (USEPA and USACOE 1998). Analytical procedures may be evaluated based on the reported accuracy and precision of the technique (i.e., the results of analyses performed on standard reference materials, and split and spiked sediment samples). Analytical detection limits are also relevant to the assessment of potential biological effects at the site. The suitability of the detection limits may be assessed by comparing them to the Level I SQTs for that substance.

In addition to reliable data, assessment of sediment quality also requires information that adequately represents the contemporary environmental conditions at the site under consideration. Therefore, the age of the data is a central question with respect to determining the applicability of the data. Natural degradative processes in the environment can lead to reductions in the concentrations of sediment-associated organic contaminants over time (Mosello and Calderoni 1990). Major events (such as storms) may result in the transport of sediments between sites, and industrial developments and/or regulatory activities may alter the sources and composition of contaminants released into the environment over time. Thus, it is important that assessments of sediment quality be undertaken with the most recent data available.

In addition to temporal variability, the sediment quality is known to vary significantly on a spatial basis. Therefore, any single sample is likely to represent only a small proportion of the geographic area in which it was collected. For this reason, data from a number of stations are required to provide a representative picture of sediment quality conditions at the site, with the actual number of stations required dependent on the size of the area under consideration, the concentrations of sediment-associated contaminants, and the variability of contaminant concentrations.

Another important factor to consider in evaluating the applicability of existing sediment quality data is the list of variables that were analyzed. It is important that the list of analytes reflects the existing and historical contaminant sources from land and water use activities in the area. In harbors, for example, chemicals such as pentachlorophenol (which is used as a preservative for pilings), tributyltin (which is used in antifouling paints for ships), and copper (which is used in antifouling paints for pleasure craft) should be measured. Similarly, highly elevated concentrations of PAHs and lead are often associated with urban stormwater discharges (Ireland, *et al.* 1996). In agricultural areas, persistent pesticides and nutrients should be considered in sediment quality assessments.

If the results of the data evaluation process indicate that the sediment chemistry data are acceptable, it is possible to proceed with the sediment quality assessment. However, if the sediment chemistry data are considered to be of unacceptable quality or are not considered to adequately represent the site, additional data may be required to complete the sediment quality assessment. Similar data quality issues must also be considered for the collection of any bioeffects data.

J.4 COLLECT SUPPLEMENTAL SEDIMENT QUALITY DATA

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

Sampling programs should be designed to delineate temporal (through time-series sampling and/or sediment coring) and spatial variability in sediment quality conditions and explicitly identify the QA/QC measures that will be implemented. Collection, handing, and storage of sediment samples should follow established protocols (e.g., ASTM 1999). Analytical methods and detection limits should be appropriate for the substances under consideration (i.e., one-half the Level I SQT values). Toxicity testing should follow the guidance that has been established by ASTM (1999) and the U.S. EPA (2000b). Bioaccumulation tests should follow the guidance given by the U.S. EPA (2000b). Guidance on conducting benthic community surveys has been developed by the U.S. EPA (1990, 1997c). Implementation of a focused, well-designed monitoring program will ensure that the resultant data will support a defensible sediment quality assessment.

J.5 ASSESS SEDIMENT CHEMISTRY DATA

Sediment chemistry data alone do not provide an adequate basis for assessing the hazards posed by sediment-associated contaminants to aquatic organisms or other receptors. In addition, interpretive tools are useful to determine if sediment-associated chemicals are present at concentrations which could, potentially, impair the designated uses of the aquatic environment. In this respect, the SQTs that have been recommended for the St. Louis River AOC provide a scientifically defensible basis for evaluating the potential effects of sediment-associated contaminants on aquatic organisms.

The assessment of sediment chemistry data consists of three main steps (Figure J-2). First, the concentrations of sediment-associated chemicals that were measured at the site under consideration should be compared to the SQTs for the protection of aquatic life. Sediment samples with mean PEC-Qs of ≤0.1, >0.1 to ≤0.6, and >0.6 should be considered to have low (i.e., <10%), moderate (i.e., <50%), and high (i.e., >50%) probabilities, respectively, of being toxic to sediment-dwelling organisms, when the results of long-term (i.e., 28-42 day) toxicity tests are considered (USEPA 2000a). Next, the levels of contaminants in site sediments should be compared to the bioaccumulation-based SQTs, including those for the protection of wildlife and for the protection of human health. Sediments in which the concentration of one or more substance exceeds these SQTs should be viewed as potentially problematic. Finally, the measured contaminant concentrations at the site should be compared to regional background levels to determine if they are elevated relative to background conditions.

Problematic levels of contamination are indicated when sediment-associated contaminants are present at concentrations above one or more of the various SQTs and above background levels. However, the results of the sediment chemistry assessment should not be viewed in isolation. Instead, these results should be evaluated in conjunction with the results of the other assessments that are conducted at the site (i.e., sediment toxicity, tissue chemistry, and benthic community structure) (Table J-1). The use of a weight-of-evidence approach is especially important when costly remediation decisions are being made.

J.6 ASSESS SEDIMENT TOXICITY DATA

Biological testing is an essential component of the sediment quality assessment process. At present, the nature and extent of available information on the effects of sediment-associated contaminants is such that there is often significant uncertainty associated with predictions of the biological significance of sediment-associated contaminants (i.e., most of the data available do not support the establishment of cause and effect relationships). Therefore, biological testing is required to provide reliable information regarding the toxicity of sediments (generally a suite of biological tests is preferred) and to confirm the results of the sediment chemistry assessment.

Biological testing is required to support three distinct aspects of the sediment quality assessment process. First, biological testing may be needed to assess the toxicity of sediments at sites where the concentrations of one or more contaminants fall between the Level I and Level II SQTs. Second, biological testing is needed to assess the toxicity of sediments likely to contain unmeasured substances. Third, biological data are needed to assess the site-specific applicability of the recommended SQTs. In this respect, additional biological testing is required when the forms of the contaminants that are present are likely to be less biologically available than those at other sites (i.e., the data that were used to support the predictive ability evaluation).

The biological testing program should be designed to assess the toxicity of whole sediments; however, investigation of the potential effects in the water column and/or pore water may also be useful in certain applications. Evaluation of whole-sediment toxicity is a key component of the sediment quality assessment process in both regulatory and management applications. The ASTM presently has developed and approved whole sediment tests for assessing the acute and chronic toxicity of freshwater sediments to a variety of invertebrate species, including amphipods, midges, and mayflies.

In addition to whole sediment toxicity tests, various procedures are available for assessing the potential for adverse effects on aquatic organisms due to the resuspension of sediments or partitioning of contaminants into the water column. Perhaps the most sensitive and frequently used of these is the bacterial luminescence test (Microtox®; Burton and Stemmer 1988; Schiewe *et al.* 1985); however, the environmental relevance of this test is not fully established. Tests using algae, invertebrates, and fish also have been employed to assess the toxicity of the suspended and/or aqueous phases (ASTM 1997a).

The assessment of sediment toxicity data involves two main steps (Figure J-3). First, the results of the toxicity tests should be compared to the negative control data to determine if the sediments are significantly toxic. Next, the toxicity test results should be compared to data from appropriately selected reference sites. In this case, a reference should only be considered to be acceptable if it has been well-characterized and satisfies the criteria for negative controls [i.e., reference sediments should have low concentrations of PCOCs (typically below Level I SQTs) and reference results should not be significantly different from controls]. Sediments that are found to be significantly toxic relative to control or reference sediments should be considered to be problematic. The results of the sediment toxicity assessment should be considered along with the results of the companion assessments that are conducted at the site (Table J-1). Additional information about sediment toxicity tests is provided in Appendix B.

J.7 ASSESS BENTHIC INVERTEBRATE COMMUNITY STRUCTURE DATA

In addition to toxicity testing, there are several other types of biological information that may also be used in the sediment quality assessment process. For example, comparison of biological indicators (such as the diversity and abundance of benthic invertebrate communities) at test sites and appropriate reference sites (i.e., sites with similar depth, particle size distribution, TOC, etc.) provides a means of assessing the relative impacts associated with exposure to site sediments (Figure J-4). Various statistical procedures may be used to help identify contaminants associated with observed biological effects when adequate sediment chemistry data are available (Breneman *et al*. 2000). Canfield *et al.* (1994, 1996, 1998) provide detailed information on the use of information on benthic invertebrate community structure in sediment quality assessments. Additional information on conducting benthic macroinvertebrate community assessments is provided in Appendix B.

J.8 ASSESS TISSUE CHEMISTRY DATA

Information on levels of contaminants in aquatic biota and on bioaccumulation supports determination of the significance of contaminant levels in sediments relative to the protection of human health and the health of wildlife that consume aquatic organisms. At this stage of the process, the measured concentrations of contaminants in biological tissues are compared to tissue residue guidelines and to regional background levels to determine if the tissues contain elevated levels of contaminants (Figure J-5). The results of this assessment should also be considered in conjunction with the other types of assessments that were conducted for the site.

Interpretation of tissue residue data is challenging for a number of reasons. While many aquatic organisms are sedentary (i.e., infaunal invertebrate species), others can be highly migratory (i.e., fish). For migratory species, it can be very difficult to establish where the exposure to bioaccumulative contaminants actually occurred. In addition, the concentrations of tissue-associated contaminants can vary depending on the trophic status, age, tissue sampled, and lipid content of the species under consideration, to name a few of the most important factors. Therefore, it is difficult to fully characterize the risks to wildlife and human health that are associated to the accumulation of contaminants in the food web. Additional information on bioaccumulation assessments is provided in Appendix B.

J.9 IDENTIFY USE IMPAIRMENTS

Sediment quality assessments are typically conducted to determine if sediments have become contaminated as a result of land and/or water use activities. When such contamination is indicated, the results of sediment quality assessments should provide the information needed to evaluate the nature, severity, and areal extent of sediment contamination. In turn, this information can be used to identify the use impairments at the site under consideration.

Each of the four elements of the sediment quality quadrad (i.e., sediment chemistry, sediment toxicity, benthic invertebrate community status, and tissue chemistry) provide important information for assessing sediment quality conditions. However, a more comprehensive interpretation of the information is possible when the results of the four independent assessments are considered together. The contingency table presented in Table J-1 provides a means of interpreting the results of detailed investigations that are conducted to determine if a site is contaminated and which uses are likely to be impaired as a result of that contamination.

J.10 DEVELOP AND IMPLEMENT A SEDIMENT MANAGEMENT STRATEGY

The ultimate objective of the sediment quality assessment process is to provide information that supports the management of environmental quality. The management decisions that are ultimately made will depend on various factors, including the nature and severity of the contamination, the potential for exposure of aquatic organisms, the management goals for the site, the availability of remediation technology, the costs associated with remediation, and public expectations. Integration of information on these factors will enable managers and others to make defensible decisions regarding remediation, abating existing pollution sources, preventing increased contaminant loadings, or simply monitoring trends in environmental contamination.

Development of a sediment management strategy (SMS) is a critical component of the contaminated site management process because it outlines the steps that should be taken to restore beneficial uses at the site. A number of sediment quality management decisions are possible, based on consideration of available information from the environmental assessment (Table J-2). At some sites, evaluation of the available information will indicate that no additional action is warranted. At other sites, monitoring for assessment of trends in sediment quality may be required. At sites that are seriously contaminated, some remedial action may be necessary to achieve environmental management goals. These remedial actions could include removal and treatment of toxic materials, isolation (or capping) of contaminated sediments, implementation of source control measures, or no action at all (i.e., permit natural degradative and sedimentation processes to mitigate contaminant effects). Sediment toxicity identification evaluation procedures and sediment spiking (Appendix B) can be used to identify specific causative agents in the sediments.

J.11 CONDUCT CONFIRMATORY MONITORING AND EVALUATION

Any sediment remediation action should include confirmatory sampling and analysis to determine if the remedial measures have achieved the goals identified in the sediment management strategy and to insure the area has not been recontaminated (Krantzberg *et al.* 1999). The design and implementation of confirmatory monitoring programs should be consistent with those that were utilized in the preliminary and detailed site investigations (DSIs). Additionally, the tools that were used to interpret the data that were collected during these investigations should also be used to evaluate the efficacy of the remedial measures that were implemented. These tools should not be based simply on chemistry and engineering considerations, but should also include biological targets (e.g., toxicity, benthos, tissue residues).

Table J-1. Contingency Table for Assessing Contaminated Sediments Using the Results of a Detailed Site Investigation

	Results of Detailed Site Investigation				Potential for Sediment-Associated Use Impairments		
	Sediment Chemistry	**Sediment Toxicity**	**Benthic Community**	**Tissue Chemistry**	**Aquatic Life**	**Wildlife**	**Human Health**
1	(+)	(+)	(+)	(+)	Highly likely	Likely	Likely
2	(+)	(+)	(+)	(-)	Highly likely	Possible	Unlikely
3	(+)	(-)	(+)	(+)	Likely	Likely	Likely
4	(+)	(+)	(-)	(+)	Likely	Likely	Likely
5	(-)	(+)	(+)	(+)	Likely	Likely	Likely
6	(-)	(+)	(-)	(+)	Likely	Likely	Likely
7	(+)	(-)	(+)	(-)	Likely	Possible	Unlikely
8	(-)	(+)	(+)	(-)	Likely	Possible	Unlikely
9	(+)	(+)	(-)	(-)	Likely	Unlikely	Unlikely
10	(-)	(+)	(-)	(-)	Likely	Unlikely	Unlikely
11	(+)	(-)	(-)	(+)	Unlikely	Likely	Likely
12	(-)	(-)	(+)	(+)	Unlikely	Likely	Likely
13	(-)	(-)	(+)	(-)	Unlikely	Possible	Unlikely
14	(-)	(-)	(-)	(+)	Unlikely	Possible	Possible
15	(+)	(-)	(-)	(-)	Unlikely	Unlikely	Unlikely
16	(-)	(-)	(-)	(-)	Highly unlikely	Highly unlikely	Highly unlikely

+ = Statistical difference between test and control or reference conditions.
- = No statistical difference between test and control or reference conditions.

Table J-2. Interpretation of Contingency Table for Assessing Sediments Using the Results of a Detailed Site Investigation

	Most Probable Interpretation	Conclusion	Probable Actions
1	Contaminant-induced degradation of benthos and bioaccumulation evident	Site contam.	Conduct RA or develop SMS
2	Contaminant-induced degradation of benthos evident; minimal bioaccumulation	Site contam.	Conduct RA or develop SMS
3	Toxicity tests not sensitive enough; bioaccumulation evident	Site contam.	Conduct RA or develop SMS
4	Toxic chemicals stressing benthos; bioaccumulation evident	Site contam.	Conduct RA or develop SMS
5	Unmeasured sediment contaminants contributing to toxicity; bioaccumulation evident	Site likely contam.	Expand DSI to measure more factors
6	Unmeasured factors contributing to toxicity; bioaccumulation evident	Site likely contam.	Expand DSI to measure more factors
7	Toxicity tests not sensitive enough; minimal bioaccumulation	Site contam.	Conduct RA or develop SMS
8	Unmeasured contaminants contributing to toxicity; minimal bioaccumulation	Site likely contam.	Expand DSI to measure more substances
9	Toxic chemicals stressing the system; minimal bioaccumulation	Site contam.	Conduct RA or develop SMS
10	Unmeasured factors contributing to toxicity; minimal bioaccumulation	Uncertain	Expand DSI to measure more factors
11	Contaminants not toxic to benthos; bioaccumulation evident	Site contam.	Conduct RA or develop SMS
12	Benthic effects probably not due to sediment contamination; bioaccumulation evident	Site contam.	Conduct RA or develop SMS
13	Benthic effects not due to sediment contamination; minimal bioaccumulation	Site contam. unlikely	No further action needed
14	No degradation of benthos apparent; tissue residues due to exposure at other sites	Site likely contam.	Expand DSI to identify sources
15	Contaminants unavailable to benthos; minimal bioaccumulation	Site contam. unlikely	No further action needed
16	Contaminant-induced degradation of benthos not evident; minimal bioaccumulation	Site contam. unlikely	No further action needed

DSI = detailed site investigation; RA = risk assessment; contam. = contaminated;
SMS = sediment management strategy.

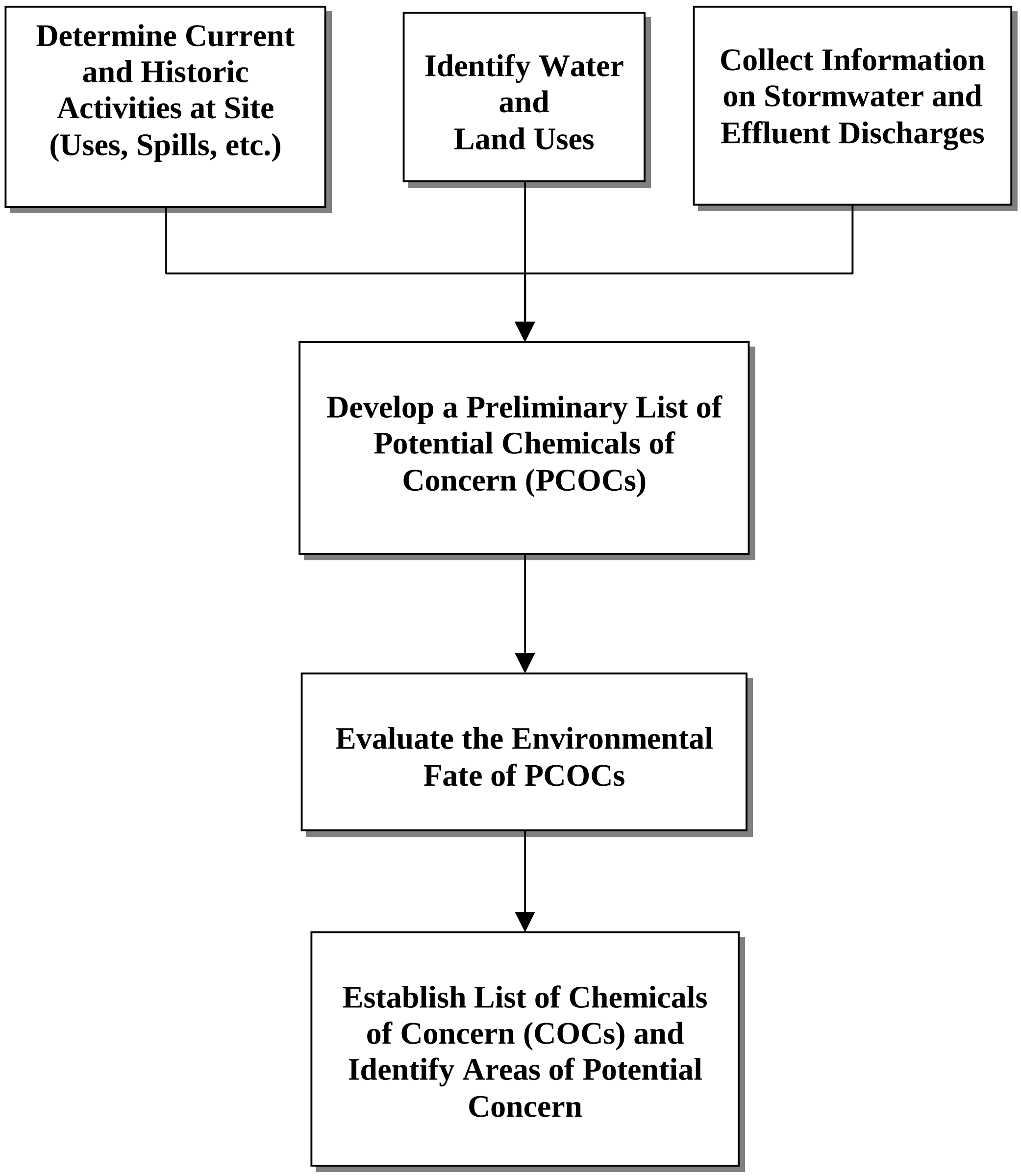
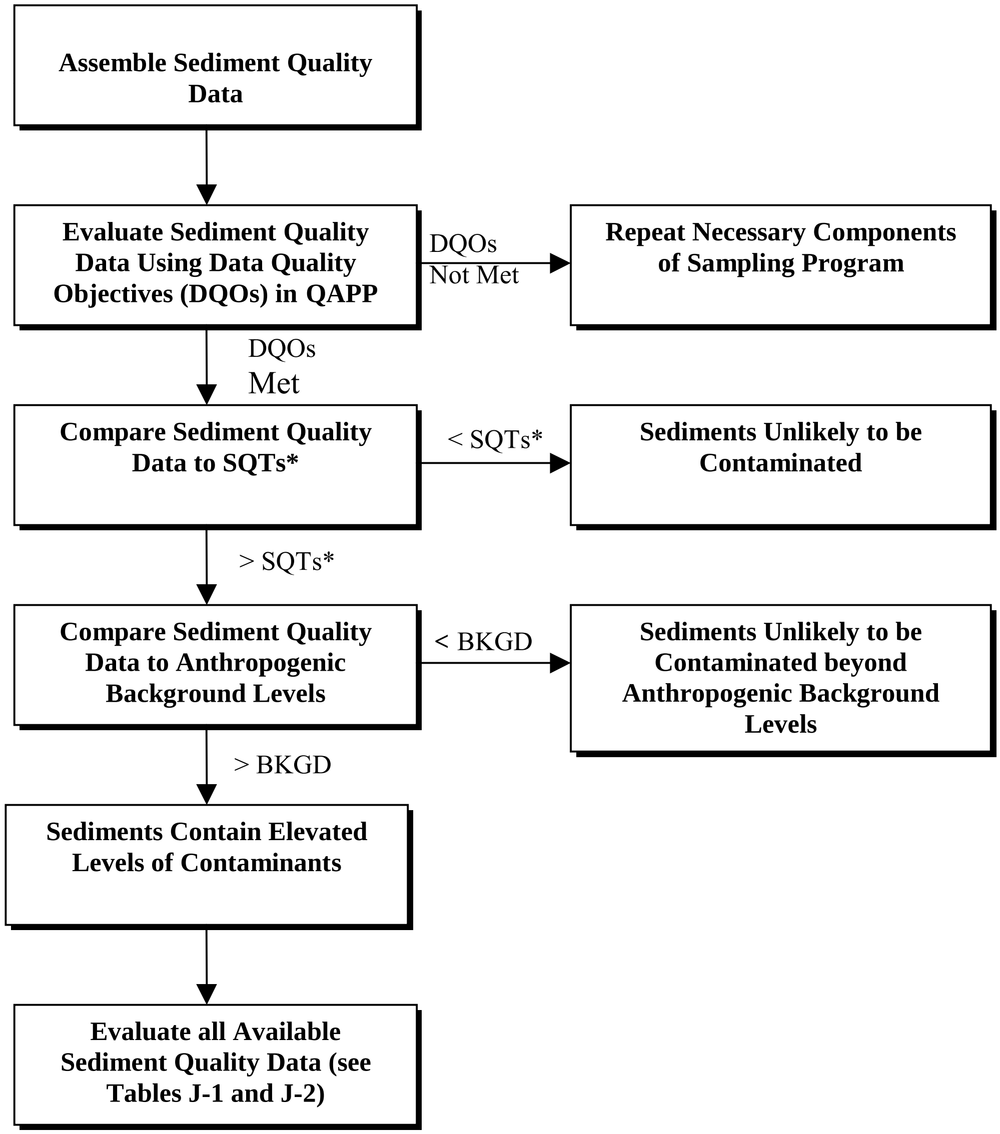
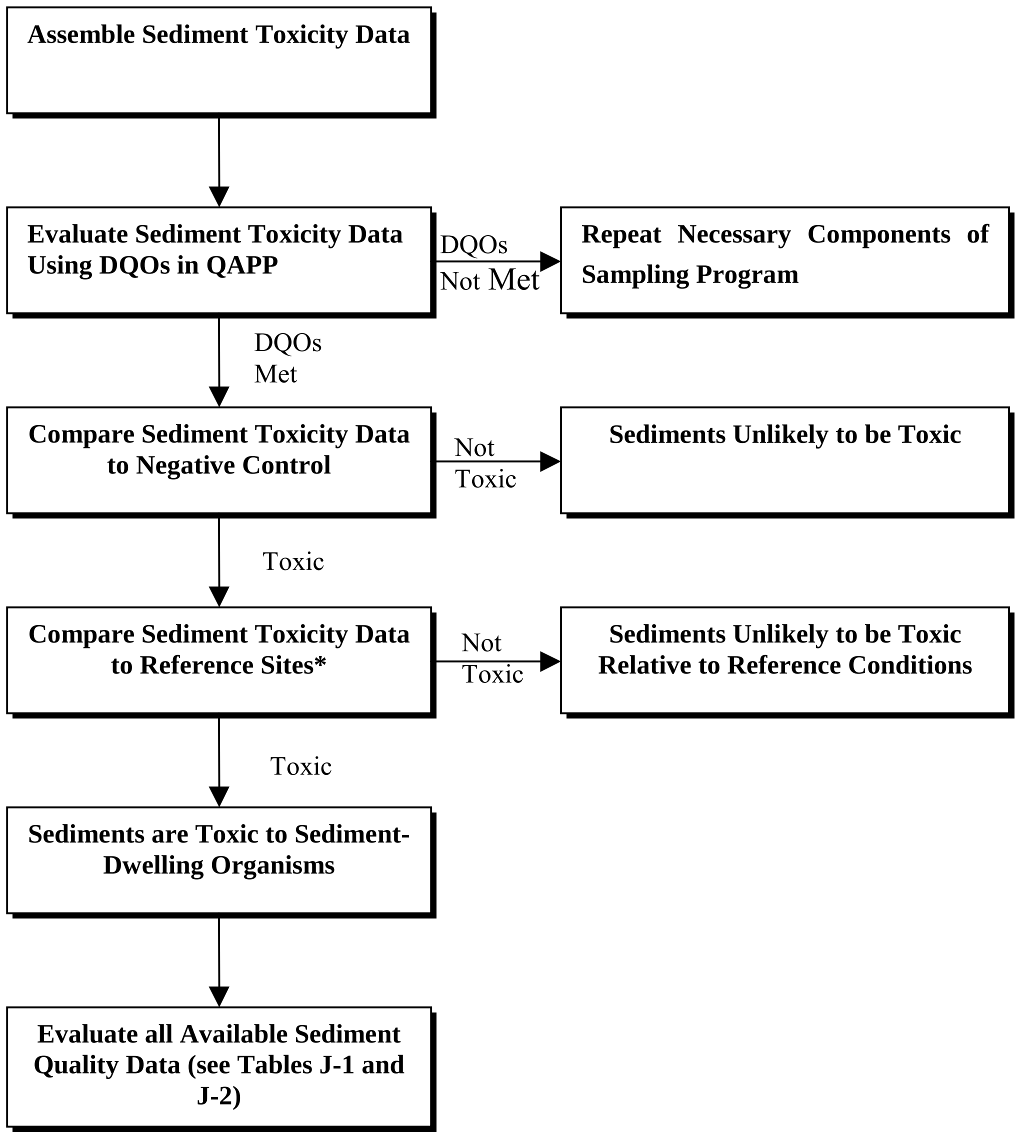


Figure J-1. Evaluation of sediment quality issues and concerns.



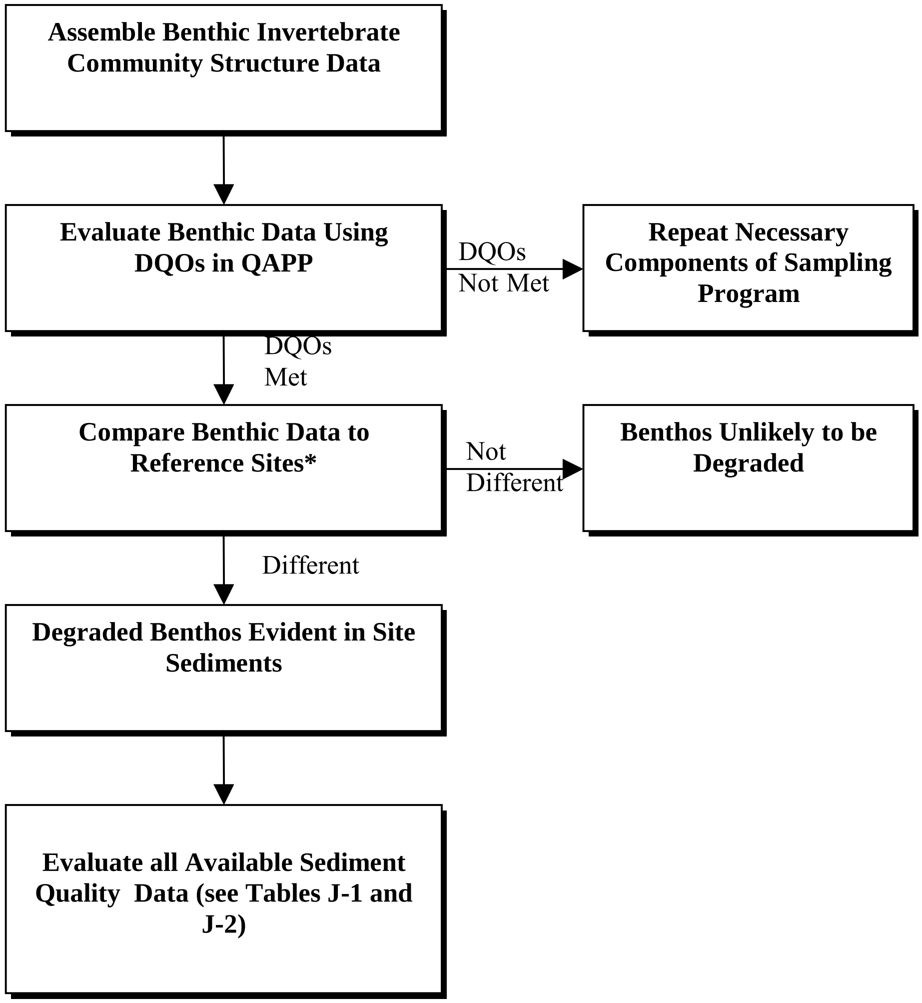
*Level I and Level II SQTs for the protection of aquatic life, wildlife, and human health.

Figure J-2. Recommended procedure for assessing sediment chemistry data.



* Comparison to reference sites is only appropriate if reference sites have been well characterized and satisfy criteria for negative controls [i.e., reference should not be significantly different from negative controls (ASTM 1999)].

Figure J-3. Recommended procedure for assessing sediment toxicity data.



* Comparison to reference sites is only appropriate if reference sites have been well characterized and satisfy the criteria that have been established for reference sites (ASTM 1999).

Figure J-4. Recommended procedure for assessing benthic data.

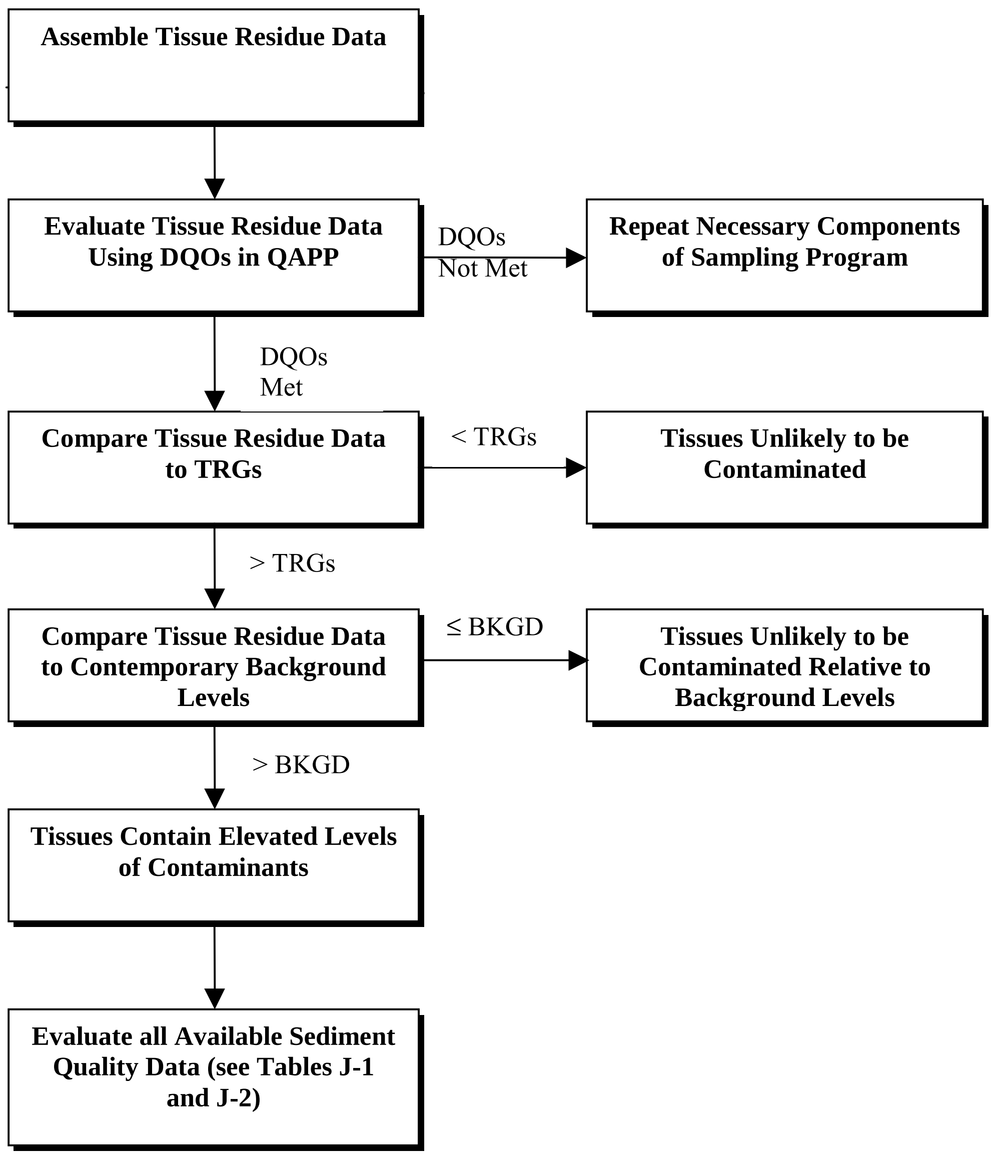


Figure J-5. Recommended procedure for assessing tissue residue data.