FRASER RIVER ACTION PLAN

A REVIEW OF ENVIRONMENTAL QUALITY CRITERIA AND GUIDELINES FOR PRIORITY SUBSTANCES IN THE FRASER RIVER BASIN

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A REVIEW OF ENVIRONMENTAL QUALITY CRITERIA AND GUIDELINES FOR PRIORITY SUBSTANCES IN THE FRASER RIVER BASIN

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Chapter 1
Introduction and Background

1.0 Introduction

A wide variety of environmental monitoring initiatives are planned for implementation under the Fraser River Action Plan. Evaluation of the results of these monitoring programs, in the context of the broad water management goals that have been established, necessitates the formulation of indicator levels or environmental quality guidelines for priority substances in the Fraser River Basin (Table 1). In addition to their use as assessment tools, environmental quality guidelines provide a basis for the development of site-specific environmental quality objectives for environmental contaminants. Furthermore, these tools may be used to identify the need for source controls to reduce inputs of toxic and bioaccumulative substances into the aquatic ecosystem.

In Canada, water quality guidelines for the protection of drinking water are derived by Health and Welfare Canada (HWC). Ambient water quality guidelines for other water uses and sediment quality guidelines are developed by Canadian Council of Ministers of the Environment (CCME). Similarly, tissue residue guidelines for the protection of human health and wildlife are formulated by Health and Welfare Canada and the Canadian Council of Minister of the Environment, respectively. While several programs that are currently ongoing in Canada have provided environmental quality guidelines for many of the substances, the requisite guidelines for many priority contaminants in the Fraser River Basin have not yet been established. Nonetheless, the Environmental Quality and Pollution Abatement Task Groups of the Fraser River Action Plan (FRAP) have a need for indicator levels of contaminants to assess environmental survey and sampling results, to establish priority areas for pollution abatement activities, and to assess the effectiveness of any remedial measures that are implemented. For this reason, a comprehensive review of existing environmental quality criteria, guidelines, objectives and standards from jurisdictions throughout the world has been conducted to support environmental management initiatives in the Fraser River Basin. It is understood, however, that any informal environmental quality guidelines selected for use under FRAP will be replaced by the CCME-approved guidelines, as they become available.

1.1 Background

Environmental quality guidelines and criteria for water have historically been concerned with the protection of human health. As such, most of the early water quality guidelines (WQG) focusses on the bacteriological characteristics of surface waters that could be used as raw water supplies. As our understanding of human toxicology expanded, so did our need for water quality guidelines that addressed the chemical attributes of water, such as metals and nitrates levels. Disposal of toxic industrial chemicals and widespread use of pesticides have lead to further concerns over the quality of potable water sources in Canada. As such, the Canadian Council of Ministers of the Environment formed a Task Force on Water Quality Guidelines to initiate the development of a comprehensive suite of guidelines for the protection of human health (CCREM 1987; HWC 1993).

Despite the quality and number of guidelines available to evaluate the quality of drinking water sources in Canada, these guidelines may not necessarily be appropriate for adoption as water quality objectives (WQO) in the Fraser River basin. While the withdrawal of raw water for drinking water supplies is an important water-use it is frequently not the most sensitive water use. Numerous evaluations have demonstrated that freshwater aquatic organisms generally have the most demanding requirements in terms of water quality (see CCREM 1987). Conservation and protection of designated water uses in the Fraser River is, at least in part, dependent on the availability of water quality guidelines for priority substances for the protection of sensitive freshwater and estuarine species. In addition, other important water uses, such as livestock watering, irrigation, recreation and aesthetics, and industrial water supplies, must be addressed in the management of water resources in the Fraser River Basin.

Contaminated sediment has been identified as a significant environmental concern in the Fraser River basin. Elevated levels of environmental contaminants may occur due to an array of natural processes; however, contamination of riverine and estuarine sediments is more likely to occur due to the activities of man. Pollutants from industrial, municipal, and non-point sources can accumulate in suspended or bed sediments and, in so doing, result in adverse effects on benthic organisms. Desorption of toxic chemicals from suspended or deposited sediments can result in the degradation of the quality of the overlying water. Sediment quality guidelines provide a means of assessing the quality of riverine and estuarine sediments and, thereby, a means of making informed management decisions regarding developmental activities.

In addition to the direct toxicological effects of water-borne and sediment-associated environmental pollutants from industrial, municipal, and non-point sources, many of these contaminants have the potential to accumulate in the tissues of aquatic biota. Bioaccumulation in aquatic biota may have significant implications to the species affected, and to the aquatic and terrestrial species, including humans, that consume aquatic biota for food. Numerical tissue-residue guidelines provide a basis for assessing the hazards that tissue-borne contaminants pose to human health and wildlife and, hence, a basis for regulating contaminant inputs into the environment.

Many areas of the Fraser River Basin are characterized by significant and expanding industrial developments. Point and non-point source effluents from these facilities, municipalities, agricultural operations, and other anthropogenic developments have the potential to degrade receiving water systems. Water quality, sediment quality, and tissue residue guidelines are required to support the evaluation of environmental quality conditions and the development of focused water management strategies in the Fraser River basin.

1.2 Definitions

Environmental quality guidelines have been developed by a number of jurisdictions in Canada, with the Canadian Council of Ministers of the Environment (previously known as the Canadian Council of Resource and Environment Ministers; CCREM) taking a lead role in these activities. In addition, various other agencies in Canada and elsewhere around the world have well established environmental quality guidelines development programs. However, many of these jurisdictions have adopted different terms for tools that they use in their water management processes. To avoid confusion in the application of these terms, the following definitions are recommended for use in the Fraser River basin:

(i) Environmental Quality Criteria and Guidelines - These are numerical concentrations or narrative statements recommended to support and maintain designated water uses. For example: "no harmful effects on mysid shrimp will result if the maximum concentration of zinc remains below 0.166 mg/L" (Suter and Rosen 1988).

(ii) Environmental Quality Objectives - These are numerical concentrations or narrative statements which have been established to support and protect the designated uses of water at a specified site. For example: "the average concentration of zinc in the Fraser River Estuary should not exceed 0.050 mg/L. This objective is predicted to protect sensitive fish and aquatic life species, and in so doing should ensure that other beneficial uses of water will not be impaired" (Swain and Holms 1985). Establishment of a water quality objective requires agreement between all of the agencies responsible for the management of water quality in the basin under consideration.

(iii) Environmental Quality Standards - These are environmental quality objectives that are recognized in enforceable environmental control laws of a level of government. For example: "the average concentration of zinc in the Columbia River basin shall not exceed 0.086 mg/L" (Oregon Department of Environmental Quality 1989).

In this document, the terms guidelines and criteria are used interchangeably. However, it should be noted that the CCME has established a unique definition for criteria (see CCREM 1987).

1.3 Study Approach

The primary objective of this study was to prepare a comprehensive database which contains the fresh and estuarine environmental quality criteria, guidelines, objectives, and standards which have been published by regulatory agencies around the world. However, the time and resources available for completion of this study were insufficient to conduct a comprehensive survey of the regulatory agencies in Canada, the United States, and around the world. Instead, a number of key agencies in Canada and the United States were contacted to obtain the most recent freshwater and estuarine guidelines that have been established within their jurisdictions. In addition, the information that had been obtained previously by MacDonald Environmental Sciences Ltd. (MESL) was utilized in this study and comprises the bulk of the project database.

This study was implemented in several steps to assure the ultimate development of a relevant and high quality product. First, the environmental quality assessment values (guidelines) and their supporting documentation were obtained from appropriate agencies or the MESL library. This

information was subsequently screened to identify the guidelines that applied specifically to freshwater and estuarine ecosystems. Next, an electronic database (in Paradox™ format) was created to facilitate tabulation and retrieval of these data (see Figure 1 for an example of a database record). Once entered, all records in the database were independently verified to assure overall data quality. The verified database was then sorted by media type and water use to develop tables summarizing the available guidelines. Finally, a report was prepared that listed the jurisdictions for which guidelines were available (Table 2) and discussed the approaches that were used to develop the guidelines for water (Chapter 2), sediment (Chapter 3), and biota (Chapter 4). Descriptions of the various approaches to derivation of environmental quality guidelines are primarily from MacDonald and Smith (1990), MacDonald and Sobolewski (1993), and Walker and MacDonald (1993).

The summaries of the environmental quality guidelines compiled during this study are presented in Volume 2 of this report (Supporting Documentation). No attempt has been made to identify which guidelines which would be the most appropriate for use in the Fraser River Basin.

Chapter 2
A Review of the Major Approaches to the Development of Water Quality Criteria and Guidelines

2.0 Introduction

Canada is renowned for the myriad lakes, rivers, streams, and wetland areas that lie within its borders, many of which are relatively pristine. In certain regions of the country, vast quantities of high quality groundwater also exist and represent important resources for local area residents. While water management policies in Canada have generally stressed non-degradation (CCREM 1987), human activities have resulted in significant degradation of surface water and groundwater resources in many locations (Allan and Ball 1990; Hiebsch 1988). At some of these locations, the concentrations of contaminants in water exceed the levels that have been recommended for the protection of human health and the environment (CCREM 1987; CCME 1991a; HWC 1989), indicating that further regulation may be required to protect certain water uses. However, Canadian water quality guidelines are not available for many contaminants. It is therefore, difficult to make science-based regulatory decisions that are protective of the designated water uses.

In spite of the absence of CCME-approved water quality guidelines, relevant assessment tools have been developed in other jurisdictions that may be appropriate for use, on an interim basis. The identification of use-protection goals is an essential element of the guidelines derivation process. Water quality criteria (WQC) and guidelines are typically formulated for the protection of these five major water uses (CCREM 1987):

(i) Raw water for drinking water supply;
(ii) Recreational water quality and aesthetics;
(iii) Freshwater, estuarine, and marine aquatic life;
(iv) Agricultural water uses (irrigation and livestock watering); and,
(v) Industrial water supplies.

While a wide variety of procedures have been used to derive water quality criteria and guidelines, the majority of these have been developed using some variation of the theoretical toxicological approach, which is an effects-based approach that relies on published toxicity data from the

scientific literature (see MacDonald 1989 for a more complete description of this methodology). Individual jurisdictions have established unique procedures which can respond to their unique requirements for assessment tools. It would be virtually impossible to review all of the approaches that were encountered in this study; therefore, six of the most relevant approaches were identified and summarized in this document, including:

(i) Canadian Council of Ministers of the Environment Approach;
(ii) Health and Welfare Canada Approach.
(iii) Water Background Approach;
(iv) Environment Ontario Approach;
(v) British Columbia Ministry of the Environment Approach; and,
(vi) United States Environmental Protection Agency Approach;
(vii) The Wisconsin Department of Natural Resources Approach

2.1 Canadian Council of Ministers of the Environment Approach

The Canadian Council of Ministers of the Environment (CCME) has developed several protocols to support the derivation of numerical water quality criteria and guidelines for the protection of various water uses, including aquatic life, livestock watering, and irrigation water. Each of these approaches has been summarized to provide the reader with background information for interpreting the resultant guidelines and criteria.

2.1.1 Aquatic Life

The CCME (1991b) has developed a protocol for the derivation of water quality guidelines for the protection of aquatic life based on both aquatic toxicity and environmental fate information. Using this approach, guidelines or interim guidelines are derived from the lowest observed effect level (LOEL) for the most sensitive life stage of the most sensitive fish, invertebrate or plant species investigated. Guidelines may be derived from the results of either acute or chronic studies, however, chronic studies with non-lethal endpoints are used preferentially. The key study or studies used to derive the guidelines must have demonstrated a clear dose/response relationship and the LOEL must be statistically significant. Full guidelines are developed when the minimum toxicological and

environmental fate data requirements are satisfied, while interim guidelines are derived when the less rigorous data requirements are met.

Using the CCME approach, guidelines are developed from a chronic study by multiplying the most sensitive LOEL by a safety factor of 0.1. This safety factor is designed to account for differences in sensitivity to a chemical due to differences in species, in laboratory vs. field conditions, and in test endpoints. When available, acute to chronic ratios (ACRs) may be used to estimate a long-term no effect concentration from the median lethal results of a short-term study. In situations where no data exist on ACRs, guidelines are derived directly from the results of acute toxicity studies. In this case, application factors are used in conjunction with the test results, depending on the environmental fate of the substance. For persistent (half-life in water > 8 weeks) and non-persistent (half-life in water < 8 weeks) chemicals, application factors of 0.01 and 0.05 are recommended, respectively.

2.1.2 Livestock Watering

Successful livestock production is dependent on the availability of ample supplies of good quality water (Ayers *et al.* 1985). Water of inferior quality may cause adverse effects on the health and growth of domestic animals and, hence, economic losses to livestock producers (Rowe and Hymas 1954). Consideration of these receptors is particularly important because livestock watering is often the most sensitive use of surface water and groundwater supplies (due to the high water consumption rates of these species). Therefore, the CCME has developed a formal protocol to support the derivation of water quality guidelines for the protection of livestock watering (CCME 1993).

The derivation of numerical guidelines using the CCME approach is supported by a comprehensive review of information on the substance under consideration. This includes i.e., physical/chemical properties, environmental fate and behaviour, and toxicity to mammalian and avian receptors. To ensure consistency in the resultant guidelines, minimum data requirements have been established which specify the quality and type of information needed to support the procedure. Depending on the availability and quality of the necessary information, either full guidelines or interim guidelines may be developed.

Using this protocol, guidelines are derived from the results of acute or, preferably, chronic exposure (via the oral route) studies that consider sensitive life stages and endpoints. These data are used to

calculate tolerable daily intakes (TDI) for both mammalian and avian receptors by dividing the geometric mean of the lowest and no observed effect doses by a safety factor. These TDI are then used, in conjunction with livestock body weights, daily water intake rates, and the recommended drinking water contribution to the total daily exposure (i.e., 20%; USEPA 1988) to derive the final guideline value.

The CCME approach is the most (and probably the only) appropriate procedure for deriving water quality guidelines for the protection of livestock water. The methodology used is similar to that used to derive guidelines for the protection of human health and is scientifically-defensible. Unlike many other jurisdictions (which adopt the human health guidelines as surrogates for livestock), the CCME approach considers the specific exposure scenarios and sensitivities of livestock receptors. This receptor specificity produces a higher level of confidence in the resultant generic guidelines.

2.1.3 Irrigation

In many regions of Canada, irrigation of agricultural crops is required to maintain the high growth rates and yields that are necessary to support contemporary farming operations. Contaminated groundwater and surface water supplies may pose significant hazards to agricultural activities through acute and/or chronic effects on agricultural plant species. For this reason, the CCME has developed a formalized protocol for deriving water quality guidelines for the protection of sensitive crop species. As in the case with other CCME protocols (e.g., CCME 1991b), detailed information on both the toxicity and environmental fate of the substance under consideration is required to support the derivation of numerical water quality guidelines. The quality and type of data needed to derive these guidelines are explicitly identified in the minimum data requirements that have been established (CCME 1993).

Using the CCME approach, water quality guidelines are derived from dose-response data for sensitive crop species grown in Canada. These data (LOEC and NOEC) are used, in conjunction with an appropriate safety factor, to calculate an acceptable soil concentration (ASC) for the substance under consideration. This acceptable soil concentration is then used to calculate the water quality guideline by considering information on bulk soil density (BSD), bulk soil volume (BSV), and irrigation rates (IR), using the equation:

$$\mathbf{WQG = ASC \times BSD \times BSV \div IR \times 10^{9}}$$

where:

WQG	=	Water quality guideline (in $\mu g \cdot L^{-1}$);
ASC	=	Acceptable soil concentration (in $mg \cdot kg^{-1}$ soil);
BSD	=	Bulk soil density (in $kg \cdot m^{-3}$);
BSV	=	Bulk soil volume (in m^3);
IR	=	Irrigation rate (in $L \cdot ha^{-1}$); and,
10^9	=	Conversion factor from kg to µg.

In calculating the generic guidelines, conservative estimates of each of these factors are used. For substances that are intentionally applied to agricultural soils (e.g., pesticides), the generic guideline is derived from an acceptable application rate (which is calculated from the no and lowest effect application rates, in conjunction with a safety factor) and the appropriate irrigation rate. While data from irrigation studies are generally not available, a procedure for deriving guidelines from such data is also provided in the protocol (CCME 1993).

2.2 Health and Welfare Canada Approach

Water quality guidelines for potable water supplies have been developed by health agencies in many countries around the world. In Canada, water quality guidelines for the protection of human health are developed by Health and Welfare Canada using a two-step procedure (Meek *et al.* 1988). In the first step of this process (risk assessment), preliminary recommendations are developed based on an assessment of risks to human health posed by the substance. In the second phase of this process (risk management), the preliminary recommendations are evaluated in terms of the practicality, costs, and potential benefits (relative to other health protection priorities) to establish a maximum acceptable concentration (MAC) in drinking water. This MAC is then adopted as the guideline for drinking water quality.

Two procedures have been identified for conducting the risk assessment component of the guideline development process (Meek *et al.* 1988). For substances which are not considered to be carcinogenic to humans, an acceptable daily intake (ADI) is calculated by dividing a no (NOEL) or lowest (LOEL) observed effect level, from an acceptable study, by an uncertainty factor. This uncertainty factor is determined on a substance by substance basis and typically ranges from 100 to 5000, depending on the quality and quantity of the toxicological data available. The ADI is then

used, in conjunction with an average adult body weight (70 kg), an average daily water intake rate (1.5 L), and an apportionment factor (20%; which accounts for the proportion of the total intake that is associated with water consumption), to derive the numerical MAC. In some cases, the MAC may be derived using the body weights and daily water intake rates of the most sensitive subpopulation (e.g., pregnant women; HWC 1989).

The procedures used to derive MAC for human carcinogens are less clearly defined than those used for non-carcinogenic substances. In general, MACs for this class of compound are set as close to zero as reasonably practicable, primarily because carcinogenesis is considered to be a non-threshold phenomenon (i.e., there is no safe level; HWC 1989). To support this assessment, lifetime cancer risks associated with exposure to individual contaminants are estimated considering information on the most relevant tumour types. Typically, the MAC is set at the concentration corresponding to a 10^{-5} or 10^{-6} lifetime cancer risk level (Meek *et al.* 1988).

No formal procedures have been identified for conducting the risk management component of the guidelines derivation process. Meek *et al.* (1988) provided a list of factors to be considered in this phase of the process. First, the proposed MAC must be achievable, at reasonable cost, by available water treatment methods. Second, the MAC must be reliably measurable by available analytical methods. While it is likely that other factors (e.g., professional judgement) are also considered during the risk management process, insufficient information is available to determine what these factors are in practice.

2.3 Water Background Concentration Approach

In the water background concentration approach, the natural background concentrations of a contaminant are determined and these levels are used to define acceptable water quality conditions at the site under consideration. Its use is based on the premise that surface water systems with superior water quality should not be degraded in any way (CCREM 1987). This approach has been used most commonly to define water quality objectives (WQOs) for relatively pristine waterbodies (MacDonald and Smith 1990), including several river systems in Canada (Dunn 1989). It has also been used to establish WQOs when natural background levels of a substance exceed the CCME guideline levels. It is particularly appropriate for groundwater systems because, once contaminated, sub-surface water supplies are very difficult and costly to remediate (Perlin 1989).

Water quality guidelines, developed using the water background concentration approach, may be established in at least two ways. First, a specified statistical definition of the upper limit of background may be established as the WQO. For example, Dunn (1989) defined the upper limit of background of 15 water quality variables as their mean value plus two standard deviations, while Breidt *et al*. (1991) used the 90th percentile value to establish these limits. Other statistical procedures have also been recommended for analyzing data on background water quality conditions (Warn 1982; Van Hassel and Gaulke 1986). Second, the WQO may be set at a level which is slightly above the background level. For example, Singleton (1985) suggested that fish and aquatic life in stream systems would be protected if suspended sediment concentrations were elevated by no more than 10% above background levels. For flow- or temperature-dependent water quality variables, the WQO may be set at the upper 95% prediction limit of the regression equation of the dependent and independent variables (i.e., either temperature or flow; Schropp *et al.* 1990). This would result in a WQO that would vary over time and space, in contrast to the single values that are normally derived.

2.4 Ontario Ministry of Environment and Energy Approach

Ontario Ministry of Environment and Energy uses an effects-based approach to the formulation of water quality guidelines (WQG) for the protection of aquatic life. This process begins with a search of the world literature pertaining to the substance under consideration. Information on aquatic toxicity, bioaccumulation, taste, odour, fish flesh tainting, genotoxicity, consumption limits, impacts on wildlife, and other related topics is collected, collated, and analyzed to support the formulation of assessment values (OMOE 1992). Supplementary data on environmental fate, sources, and water quality criteria and guidelines from other agencies are also obtained and considered in this process. The resultant data are then critically assessed for adequacy and admissibility using a well defined set of criteria. Derivation of numerical water quality guidelines is only considered if the specified minimum amount of primary information exists.

In Ontario, water quality guidelines are derived from the lowest observed effect level (LOEL) for the most sensitive life stage of the most sensitive fish, invertebrate or plant species represented in the toxicological data set (OMOE 1992). Guidelines may be derived from the results of either acute or chronic studies; however, Ontario favours the use of chronic studies with non-lethal endpoints. The key study or studies used to derive the guideline must have demonstrated a clear dose-response

relationship and the LOEL must be statistically significant. If the minimum data requirements are met, the initial or preliminary water quality guideline is set by multiplying the LOEL by a standard uncertainty factor (0.1). If fewer toxicological data are available, a preliminary WQG is derived by applying a more conservative uncertainty factor (which is determined based on the quality and quantity of data available) to the LOEL.

The preliminary water quality guideline forms the basis for the subsequent stages of the guidelines development process. For substances that are known or thought to accumulate significantly in the tissues of aquatic biota [i.e., if the chemical has a bioconcentration factor (BCF) of > 1000, a log K_{ow} of > 4, or is non-polar], a separate assessment of bioaccumulation is undertaken. Where consumption limits for the protection of human health or wildlife apply (i.e., concentrations of contaminant residues in fish), these levels are divided by an established BCF to obtain an estimate of the concentration of the substance in water that would result in the acceptable tissue concentration (OMOE 1992). If this value is lower than the preliminary guideline, then it is adjusted downward accordingly. Mutagenicity of the substance and its potential impacts on the taste and odour of water and fish flesh, on wildlife species, on recreational water uses, and on sediment quality are also evaluated. The results of any of these assessments may be used to adjust the preliminary WQG in an appropriate manner.

The Ontario Ministry of Environment and Energy has acknowledged the need to modify the generic guidelines under certain circumstances (OMOE 1984). According to Ontario's water management policy document (OMOE 1984), deviations from the generic guidelines may be considered when:

(i) Atypical background water quality is present;
(ii) Irreversible human-induced conditions are present;
(iii) Social and/or economic costs of attaining WQG would be prohibitive; or,
(iv) Suitable treatment techniques are not available.

However, well-defined procedures for deriving numerical, site-specific water quality objectives have not yet been proposed (OMOE 1992).

2.5 British Columbia Ministry of the Environment Approach

The British Columbia Ministry of the Environment (BCMOE) is one of the few agencies in Canada that have actively pursued the development of site-specific water quality objectives for the protection of aquatic life (Environment Canada and the Prairie Provinces Water Board have also been active in this area). While general information on this process is provided in the *Principles for Preparing Water Quality Objectives in British Columbia* (BCMOE 1986), it was necessary to interview agency staff to obtain more detailed guidance on these procedures (L. Pommen. Water Quality Branch. BCMOE. Victoria, B.C. pers. comm.).

Formulation of water quality objectives for a specific body of water in British Columbia is a multi-stage process. The first step is the derivation of generic water quality criteria that are considered to be generally applicable to water bodies throughout the province. These criteria are developed for the protection of the five major uses of water in British Columbia: public water supply and food processing, aquatic life and wildlife, agriculture (including livestock watering and irrigation), recreation and aesthetics, and industry (BCMOE 1986). These criteria are either derived by the province (using a procedure that is more or less consistent with the CCME approach) or adopted directly from the Canadian water quality guidelines (CCREM 1987). These values then form a scientific basis for the site-specific water quality objectives that are recommended for specific bodies of water.

The development of site-specific water quality objectives requires detailed information on the waterbody under investigation. A comprehensive assessment of the ambient environmental conditions in the system is conducted to support the objectives development process. This regional basin assessment includes the collection of information on point and non-point source discharges, on water uses, on ambient water quality and quantity, and on the biological characteristics of the system. Evaluation of these data facilitates identification of the water uses that are to be protected in the waterbody and the priority water quality variables for which objectives are needed. For each of these substances, the most sensitive water use is identified and the corresponding water quality criterion is evaluated to determine its applicability to the site.

In most cases, the generic criterion for the most sensitive water use is adopted as the water quality objective. However, under certain circumstances the generic criteria may be modified to reflect local conditions. For example, the toxicity of several metals to aquatic organisms is known to be dependent on water hardness (CCREM 1987) and, hence, the criteria for these substances are

frequently expressed in terms of water hardness. For lead, the 30-day average criterion is 4 $\mu g \cdot L^{-1}$ at water hardness of 20 to 40 $mg \cdot L^{-1}$ $CaCO_3$ (Nagpal 1987). The criteria values increase with increasing water hardness, reflecting this relationship between toxicity and water hardness. Therefore, the site-specific objective for lead at any given site would be determined based on the water hardness at that site. Likewise, the provincial criteria for total ammonia are expressed in terms of temperature, pH, and salinity. Therefore, the site-specific objective for total ammonia would be calculated using the equations relating ammonia toxicity to water temperature, pH, and salinity.

The British Columbia approach also recognizes that high levels of naturally-occurring substances may occur in certain areas. Where high background levels of a substance occur, the generic criterion would also be modified to account for the natural range and variability in the concentrations. In this case, the water quality objective may be expressed in terms of a statistical interpretation of the existing water quality data (i.e., the 90th percentile concentration) or as an allowable increase over background levels (such as for suspended sediments; Singleton 1985). Less frequently, the generic criteria have been modified when atypical biological communities occurred in the system (e.g., when only warmwater fish species are present). In these latter situations, the generic criteria would be re-calculated using only the toxicity data on, for example, warm water fish species. Together, these procedures provide a reasonable means of adapting the generic criteria to account for local conditions and, thereby, achieving an increased level of site-specificity.

2.6 United States Environmental Protection Agency Approach

The United States Environmental Protection Agency (USEPA) has developed a formal protocol for deriving generic, numerical water quality criteria for the protection of aquatic life and their uses (Stephan *et al.* 1985). Using this approach, information is compiled on the physical and chemical properties of the substance under consideration, on its toxicity to aquatic plants and animals, on its bioaccumulation in aquatic organisms, and on its potential effects on consumers of aquatic biota. The formalized protocol includes specific procedures for calculating final acute (FAV), final chronic (FCV), final plant (FPV), and final residue values (FRV) from the available data, provided that the minimum data requirements have been met. For example, derivation of a final acute value for marine and estuarine waters requires acute toxicity data on at least eight families of marine organisms, including at least two families of chordates, five families of invertebrates, and one other

family (e.g., plant). The short-term maximum concentration (criterion maximum concentration; CMC) of the substance is then calculated by applying a safety factor (0.5) to the FAV. The lowest of the FCV, FPV, and the FRV is used directly to establish the long-term mean concentration (criterion continuous concentration; CCC). The criteria are then subjected to critical review to evaluate the completeness of the data and the appropriateness of the results.

In addition to the protocol for formulating generic national criteria, the USEPA has also recommended three distinct procedures for deriving site-specific water quality criteria (USEPA 1983), including:

(i) Recalculation procedure;
(ii) Indicator species procedure; and,
(iii) Resident species procedure.

2.6.1 Recalculation Procedure

The recalculation procedure is a method for deriving site-specific water quality criteria that account for any real difference between the sensitivity range of the species of aquatic organisms represented in the national data set and that of the species that occur (or have historically occurred) at a site under consideration (USEPA 1983). Using this procedure, data on species that are not resident at the site under consideration are eliminated from the data set used to formulate the national criterion. Then, a site-specific FAV is calculated using the same methodology employed to derive the national FAV. The site-specific CMC is then calculated by applying a safety factor to the site-specific FAV. Subsequently, the site-specific CCC is derived by dividing the site-specific FAV by the acute to chronic ratio established during the development of the national criterion. This technique also provides a basis for determining a site-specific FRV using information specific to the site.

In general, the recalculation procedure may be used to derive site-specific water quality criteria only if the minimum data requirements established for formulating national criteria are met. These minimum data requirements may be waived at sites that have fewer than eight families of aquatic organisms. In these situations, the most sensitive resident family mean acute value (which is calculated as the geometric mean of the mean acute values established for each species in the family) would be adopted as the site-specific final acute value. Otherwise, additional toxicity testing on

resident species in laboratory water is needed to generate the information necessary to derive the site-specific water quality criteria.

2.6.2 Indicator Species Procedure

The indicator species procedure is a powerful tool for modifying generic water quality criteria to account for the unique characteristics of the site under investigation. This procedure is based on the assumption that the physical and/or chemical characteristics of water can vary among sites and may influence the bioavailability and, hence, toxicity of environmental contaminants. In many cases, the factors that influence the toxicity of xenobiotic substances have been identified. For example, relationships between water hardness and acute toxicity to fish have been established for several metals (e.g., cadmium, copper, lead, and nickel; CCREM 1987). Likewise, the toxicity of ammonia to fish is known to be a function of pH and temperature (MacDonald *et al.* 1987). The presence of other contaminants in water and other factors (such as suspended particulate matter) at a site could affect the bioavailability of the substance under consideration. Consideration of the factors influencing the toxicity and/or bioavailability of a substance at a site is likely to improve the utility of the water quality criterion.

The indicator species procedure provides a means of accounting for the effects of site characteristics on bioavailability without the need for detailed information on the nature of these relationships. Using this procedure, acute toxicity tests are conducted with indicator species (i.e., acceptable non-resident species that are used as surrogates for resident species) using both site water and laboratory water (i.e., standard reconstituted laboratory water). Typically, rainbow trout (*Oncorhynchus mykiss*), fathead minnows (*Pimephales promelas*), water fleas, (*Ceriodaphnia dubia and Daphnia magna*), and the alga, *Selenastrum capricornutum* are used as indicator species because they are easy to culture, widely available, and generate consistent data (Willingham 1988; MacDonald *et al.* 1989). Information generated in these investigations is used to determine the ratio of the toxicity of the substance in water from the site to its toxicity in laboratory water, which is known as the *water effect ratio*. The calculated water effect ratio is then used to convert the national criterion maximum concentration to a site-specific CMC. For example, if a substance is twice as toxic in site water as it is in laboratory water, then the national CMC would be divided by a factor of two to obtain the site-specific value. A site-specific final acute value is then calculated from the site-specific CMC (i.e., by multiplying the CMC by a safety factor), which is subsequently used to derive the site-specific criterion continuous concentration (i.e., CCC = FAV ÷ acute:chronic ratio).

A site-specific final chronic value may be derived in one of three ways using the indicator species procedure. First, the site-specific FAV may be divided by the acute to chronic ratio for the chemical, if that ratio has been determined. Second, acute and chronic toxicity tests on at least one fish and one invertebrate species (either resident or indicator species may be used) may be performed in site water. Using the results of these bioassays, acute to chronic ratios are calculated for each species and site-specific FAV is divided by the geometric mean of these ratios to derive the site-specific FCV. Third, a chronic toxicity water effect ratio may be calculated from the results of chronic toxicity tests conducted in both site water and laboratory water. Data on at least one fish and one invertebrate species are required to calculate the geometric mean chronic water effect ratio, which is then used to modify the national FCV.

2.6.3 Resident Species Procedure

The resident species procedure is designed to account for both of the major factors affecting the derivation of site-specific criteria: the sensitivity of the resident species and, the influence of site-water characteristics on toxicity (USEPA 1983). This procedure involves the generation of a complete data set on the acute toxicity of the substance under consideration using site water and resident species (i.e., that satisfies the minimum data requirements for deriving criteria). At least eight families of aquatic organisms that are resident at the site must be represented in this data set, unless fewer than eight families occur at the site. The site-specific acute toxicity data are used to calculate a final acute value for the site. The site-specific FCV may then be determined using one of the three methods described for the indicator species procedure (Section 2.6.2). The final criteria values for the substance at the site (i.e., CMC and CCC) are subsequently derived using procedures outlined for the national criteria (Stephan *et al.* 1985).

2.7 The Wisconsin Department of Natural Resources Approach

The Wisconsin Department of Natural Resources (WDNR) has developed a procedure for deriving water quality criteria for the protection of wildlife that considers the potential for bioaccumulation of persistent toxic substances. WDNR defines a wildlife criterion as the concentration of a substance in water which, if not exceeded, protects wildlife populations from adverse effects resulting from ingestion of surface waters and consumption of aquatic organisms taken from these

surface waters (Goodman 1990). Wildlife criteria provide a means of accounting for the exposure of wildlife to toxic substances as a result of their accumulation in the tissues of aquatic biota.

WDNR recommends that wildlife criteria be developed, preferentially, from the results of bioassays conducted on wildlife species. Criteria may also be developed from data on non-wildlife mammalian or avian species if data on wildlife species are not available. Acceptable tests include long-term studies (90 days for mammals and 28 days for birds) in which subchronic and reproductive or developmental effects have been investigated using the oral, dermal, or inhalation exposure routes. When insufficient information is available to develop a wildlife criterion, a wildlife advisory may be calculated using those data which are available (Goodman 1990).

Wildlife criteria are derived in a one step process, in accordance with the following:

$$WC = \frac{NOAEL \cdot BW \cdot SSF}{WIR + (FIR \cdot BAF)}$$

where:

WC = Wildlife Criterion (in $mg \cdot L^{-1}$);

NOAEL = No Observed Adverse Effect Level (in $mg \cdot kg^{-1}$ $BW.day^{-1}$);

BW = Body Weight of species selected (in kg);

SSF = Species Sensitivity Factor;

WIR = Water Intake Rate (in $L \cdot day^{-1}$);

FIR = Food Intake Rate (in $kg \cdot day^{-1}$); and,

BAF = Bioaccumulation Factor (in $L \cdot kg^{-1}$).

The safety factor is used to account for differences in the sensitivity of test species and may range from 0.01 to 1 (Goodman 1990).

Chapter 3
A Review of the Major Approaches to the Development of Sediment Quality Criteria and Guidelines

3.0 Introduction

Concerns relative to the management of aquatic ecosystems have traditionally focused on water quality. The importance of sediments in determining the fate and effects of a wide variety of contaminants has become more apparent in recent years (Long and Morgan 1990). Sediment quality is particularly important because many toxic contaminants found in only trace amounts in water may accumulate to elevated levels in sediments. Sediments serve both as reservoirs and as sources of contaminants to the water column. Sediments also tend to integrate contaminant concentrations over time and sediment-sorbed contaminants have the potential to affect benthic and other sediment-associated organisms directly (Chapman 1989). Furthermore, many sediment-associated contaminants have the potential to bioaccumulate to significant levels in the food web, thereby representing potential hazards to the organisms (including humans) that consume aquatic biota.

Contaminated sediments have been found in freshwater, estuarine, and marine ecosystems throughout the world. In Canada, the most severely contaminated sediments have been observed in the vicinity of heavy industrial operations, pulp and paper mills, and urban developments. Contaminated sediments have been observed throughout the Fraser River Basin (e.g., Mah *et al.* 1989; Dwernychuk *et al.* 1991), yet Canadian sediment quality guidelines (SQG) are not yet available to evaluate sediment chemistry data. While CCME-approved sediment quality guidelines are forthcoming (S. Smith. Evaluation and Interpretation Division. Environment Canada. pers. comm.), the immediate need for sediment quality assessment tools in the Fraser River Basin will necessitate that use of guidelines that have been derived in other jurisdictions.

A variety of approaches have been devised to formulate regional or national sediment quality criteria (SQC) and guidelines in North America and elsewhere in the world. These approaches have been reviewed and summarized by Chapman (1989), Persaud *et al*. (1990), Beak (1987; 1988), United States Environmental Protection Agency (USEPA 1989a; 1989b; 1992), Sediment Criteria Subcommittee (1989; 1990), MacDonald *et al*. (1992) and MacDonald (1993a). Many of these approaches are not directly relevant to the establishment of interim sediment quality guidelines in

the Fraser River Basin. Therefore, only five of the major approaches have been summarized in this document:

(i) Canadian Council of Ministers of the Environment Approach;
(ii) Sediment Background Approach;
(iii) Equilibrium Partitioning Approach;
(iv) Tissue Residue Approach; and,
(v) Apparent Effects Threshold Approach.

3.1 Canadian Council of Ministers of the Environment Approach

The Canadian Council of Ministers of the Environment approach to the derivation of SQG was originally developed to provide informal tools for assessing the potential for biological effects of sediment-sorbed contaminants tested in the National Status and Trends Program (NSTP; Long and Morgan 1990). This approach has been modified in recent years to accommodate the needs of the CCME (Smith and MacDonald 1993) and Florida Department of Environmental Protection for numerical SQC (MacDonald 1993a).

Using this approach (also known as the weight-of-evidence approach; WEA), matching sediment chemistry and biological effects data are collected, evaluated, and incorporated into a common database (termed the Biological Effects Database for Sediments; BEDS). Data sets obtained, are screened to evaluate their overall applicability to the guidelines derivation process (i.e., considering existence of matching sediment chemistry and biological effects data, sampling, analytical and testing methods, the type and magnitude of the end-point measured, and the degree of concordance between the chemical and biological data). Data which passed the screening tests are incorporated into the database on a substance by substance basis. Individual entries consist of the contaminant concentration, the type of biological response measured (usually specifying the location of the test), and an indication of whether or not there was concordance between the effect observed and the concentrations of a specific chemical (i.e., no effect, no or small gradient, no concordance, or a hit, which indicates that an effect was measured and correlated with sediment chemistry).

For those analytes for which sufficient data exist (20 effects and 20 no effects data entries), data tables are prepared and sorted according to ascending chemical concentrations. Three ranges of

chemical concentrations are identified using a method that considers both the concentrations associated with biological effects (the "effects" data) and those associated with the no observed effects (the "no-effects" data). A threshold effects level (TEL) is calculated as the square root of the product of the lower 15th-percentile concentration associated with observations of biological effects (the ERL) and the 50th-percentile concentration of the no-observed-effects data (the NER-M). A safety factor of 0.5 is applied to the TEL to define a no-observable-effects level (NOEL). A probable-effects level (PEL) is calculated as the square root of the product of the 50th-percentile concentration of the effects data (the ERM) and the 85th-percentile concentration of the no effects data (the NER-M; Long and MacDonald 1993). The lower value is used to define the upper limit of the no effects range and the higher value is used to define the lower limit of the probable effects range. The concentrations between the two values are defined as the possible effects range. In Canada, the NOEL is used as the generic SQG (Smith and MacDonald 1993).

3.2 Sediment Background Concentration Approach

The sediment background concentration approach (SBA) has been used primarily as an assessment tool for determining if, and to what extent, bed sediments have been contaminated by trace elements (generally metals and metalloids). This procedure has also been used to develop regional sediment quality criteria in a number of areas (e.g., in the Great Lakes, Texas, Virginia, and Illinois; SAIC 1991). A novel application of this approach has been developed to determine background levels of metals in marine and estuarine sediments in the Gulf of St. Lawrence (Loring 1991) and Florida (Schropp *et al.* 1990).

Background concentrations of trace elements in bed sediments may be determined using one of three methods. First, historical sediment quality data may be available for the site under consideration and this information may be assembled and evaluated to determine historical background levels of priority contaminants (Giesy and Hoke 1990). The analytical methods used to generate these data must be compatible with those that are presently being used at certified analytical laboratories. In this respect, both the digestion techniques and quantification procedures utilized must be evaluated.

If historical data are not available or not appropriate, sediment coring techniques may be used to collect sediment samples from various depths at the site under consideration (Mudroch and MacKnight 1991). When used in conjunction with information on sedimentation rates, data

generated from this type of sampling program may be used to estimate historical background levels of specific contaminants.

Reference sites also provide a basis for establishing background levels of metals and metalloids in bed sediments (Persaud *et al.* 1990). Using this procedure, one or more nearby reference sites, which are considered to represent uncontaminated sediment quality conditions are selected (Giesy and Hoke 1990). Surficial sediment samples are then collected and analyzed to obtain data on the concentrations of various analytes at these sites. Little specific guidance is available on the calculation of background levels of contaminants in sediments. As recommended for water, the mean concentration plus two standard deviations or the 90th percentile value could be used to establish the upper limit of the background of contaminant concentrations (Dunn 1989; Breidt *et al.* 1991).

The Florida Department of Environmental Regulation (Schropp *et al.* 1990) and others (e.g., Loring 1991) has developed a unique approach for determining background concentrations of metals in coastal sediments. This procedure relies on normalization of metal concentrations to a reference element, such as aluminum or lithium. Briefly, data on sediment metal concentrations were collected from roughly 100 sites which were thought to be representative of natural estuarine areas throughout Florida. Concentrations of individual metals were then regressed against concentrations of aluminum (following log transformation). The 95% prediction limits were then calculated and plotted for each metal. Significant correlations with aluminum concentrations were obtained for arsenic, cadmium, chromium, copper, lead, nickel, and zinc. These plots, provided a basis for interpreting data on the concentrations of metals in sediments, such that anthropogenic enrichment of metal levels would be suspected when metals to aluminum ratios exceeded the upper 95% prediction limits calculated from the regression equation (Schropp and Windom 1988). Loring (1991) subsequently demonstrated that normalization to lithium concentrations provided a superior basis for evaluating anthropogenic metals enrichment in high latitude coastal sediments. No comparable technique for evaluating the probable origin of metals in freshwater sediments was found, however.

Procedures for determining background concentrations of organic contaminants in sediments are not well developed and their improvement presents a unique challenge for several reasons. First, hydrocarbons in bed sediments as a result of natural processes (i.e., seepages, etc.); however, these discharges tend to be isolated and infrequent. Therefore, sediment chemistry data from nearby reference sites may not fully reflect these natural contaminant sources. Second, the background

concentrations of substances that are not produced naturally are expected to be non-detectable. However, it is well established that detectable concentrations of many contaminants (e.g., toxaphene) occur in sediments due to the long range transport of atmospheric pollutants (Mudroch *et al.* 1989). Therefore, there is a need to develop procedures of establishing background levels of these organic substances that will support the derivation of realistic sediment quality criteria.

3.3 Equilibrium Partitioning Approach

The water-sediment equilibrium partitioning approach (EqPA) has been one of the most studied and evaluated approaches used to develop sediment quality criteria (primarily for non-polar hydrophobic organic chemicals) in the United States (Pavlou and Weston 1983; Bolton *et al*. 1985; Kadeg *et al*. 1986; Pavlou 1987; Di Toro *et al.* 1991). It has also been used to derive sediment quality criteria for metals and non-polar organics in the Netherlands (Van Der Kooij *et al.* 1991; van de Plassche and de Bruijn 1992). This approach assumes that the distribution of contaminants among different compartments in the sediment matrix (i.e., sediment solids and interstitial water) is predictable based on their physical and chemical properties. It also assumes that a dynamic equilibrium of the contaminant exists between sediment and interstitial water. This approach has been supported by the results of a number of sediment toxicity tests, which indicated positive correlations between the biological effects observed and the concentrations of contaminants measured in the interstitial water (Swartz *et al.* 1990).

Using this approach, water quality criteria derived for the protection of freshwater or marine organisms are used to calculate sediment quality criteria. A basic assumption underlying this approach is that the water quality criteria formulated for the protection of water column species are also applicable to benthic organisms (Di Toro *et al*. 1991). Sediment quality criteria are generally calculated using the final chronic values (FCV), in conjunction with the sediment/water partition coefficients (K_p) for the specific contaminants. The calculation procedure for deriving SQC for non-ionic organic contaminants is as follows:

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

where:

SQC = Sediment quality criterion (in $\mu g \cdot kg^{-1}$);

K_p = Partition coefficient for the chemical (in $L \cdot kg^{-1}$);

K_p = $K_{oc} \cdot f_{oc}$;

FCV = Final chronic value (in $\mu g \cdot L^{-1}$);

K_{oc} = Organic carbon/water partition coefficient (in $L \cdot kg^{-1}_{oc}$); and,

f_{oc} = Fraction of organic carbon in the sediment.

This procedure is considered to be appropriate for deriving sediment quality criteria for non-ionic organic substances (such as polycyclic aromatic hydrocarbons; polychlorinated benzenes, biphenyls, dioxins, and furans; and most pesticides; USEPA 1991). For these substances, total organic carbon (TOC) normalization appears to provide a reliable basis for predicting toxicity to aquatic organisms (Swartz *et al.* 1990). The role of acid volatile sulfide (AVS) in determining the bioavailability of metals is also under investigation (Di Toro *et al*. 1989), and normalization procedures for this class of chemicals are being developed (Di Toro *et al.* 1992). Di Toro *et al.* (1991) have also noted that porewater dissolved organic carbon (DOC) levels may influence the bioavailability of hydrophobic compounds, but, the nature of this relationship has yet to be established.

3.4 Tissue Residue Approach

The tissue residue approach (TRA) to the development of sediment quality criteria (which is also known as the biota-water-sediment equilibrium partitioning approach) is designed to assess sediment contaminant concentrations in terms of their potential for bioaccumulation (Cook *et al.* 1992). The first step in this process is to determine the contaminant concentrations that are considered to be acceptable in the tissues of aquatic biota. Several procedures have been used to establish these tissue residue guidelines, including:

(i) Adopt the existing tolerances or criteria for the protection of human health (Huston 1988);

(ii) Adopt the existing tissue residue criteria for the protection of wildlife consumers of aquatic biota (Walker and MacDonald 1993); and/or,

(iii) Determine dose-response relationships between tissue residue levels in aquatic biota and adverse effects on those organisms (MacDonald 1993b; Cook *et al.* 1992).

Once the maximum acceptable tissue residue level has been established, it is necessary to establish linkages between the chemical residues in specific organisms and the concentrations of that substance in sediments. These linkages are primarily determined using equilibrium partitioning models (Clark *et al.* 1988), however, they may also be estimated using site-specific measurements of sediment-organism partition coefficients (Kuehl *et al.* 1987) and related procedures (Cook *et al.* 1992). Using this latter procedure, numerical SQC are developed from tissue residue criteria, in conjunction with sediment to aquatic biota bioaccumulation factors, as follows:

$$\mathbf{SQC = TRC \div BAF}$$

where:

SQC = Sediment quality criterion ($\mu g \cdot kg^{-1}$);

TRC = Tissue residue criterion ($\mu g \cdot kg^{-1}$); and,

BAF = Sediment to biota bioaccumulation factor.

Using this equation, SQC for the protection of aquatic life, wildlife, and/or human health may be derived.

The potential for bioaccumulation of contaminants from sediments to the tissues of aquatic organisms must be considered in the development of site-specific sediment quality criteria and guidelines. The main application of the tissue residue approach in the derivation of SQC is in the evaluation of effects-based sediment quality criteria. In this context, tissue residue criteria for the most sensitive use (i.e., aquatic life, wildlife, or human health) would be used to derive a bioaccumulation-based sediment quality criterion. If the bioaccumulation-based value is lower than the effects-based value, then the bioaccumulation-based value would be selected as the SQG.

This approach has been used successfully to establish sediment contamination limits for 2,3,7,8 tetrachlorodibenzo-p-dioxin (T_4CDD) for Lake Ontario on the basis of fish tissue residues (Endicott *et al*. 1989; Cook *et al*. 1990). The relevance of this approach to the derivation of SQC is supported by data which demonstrate that declines in DDT residues in fish and birds (since its use was banned) are strongly correlated with declining concentrations of this substance in surficial sediments in the Great Lakes and Southern California Bight. As such, this approach is a logical companion for the effects-based procedures that have been described previously.

3.5 Apparent Effects Threshold Approach

The apparent effects threshold approach (AETA) to the development of sediment quality criteria was developed by Tetra Tech Inc. (1986) for use in the Puget Sound area of Washington State. The AETA is similar in many ways to the sediment quality triad (Long and Chapman 1985; Chapman 1986; Long 1989) and screening level concentration (Neff *et al.* 1986) approaches, since each of these approaches relies on matching biological effects and sediment chemistry data. The practical goal of this procedure is to define the concentration of a contaminant in sediment above which significant (p 0.05) biological effects are ***always*** observed. These biological effects include, but are not limited to, toxicity to benthic and/or water column species (as measured using sediment toxicity bioassays) and changes in the diversity and abundance of benthic organisms.

Apparent effect threshold values are calculated using the following procedure (Malek 1992). First, a sampling program is conducted to generate paired sediment chemistry and biological effects data. Then, the 'impacted' and 'non-impacted' sites are identified based on the results of investigations conducted using each biological indicator (e.g., using amphipod bioassays, a station would be considered as impacted if the survival of test organisms was significantly different from that in reference sediments). An AET for each chemical constituent may then be established by identifying the highest detectable concentration among the samples that do not exhibit statistically significant effects. The preliminary AET is then checked to verify that adverse biological effects are observed at all sites having higher concentrations of the chemical. In Washington State, AET values are based on dry weight normalized contaminant concentrations for metals and either dry-weight- or TOC- normalized concentrations for organic substances (Barrick *et al*. 1988; Washington Department of Ecology 1990a).

Using the AETA, the Washington Department of Ecology (1990a; 1990b) has established marine sediment management standards for 47 contaminants of concern (with the exception of phenanthrene, which was derived using the EqPA) to support the identification of contaminated sites and the establishment of appropriate cleanup levels. The sediment quality standard for each substance was set at the lowest AET value for the four biological indicators considered (including amphipod mortality, oyster larvae abnormality, Microtox bioluminescence, and benthic macroinvertebrate abundance). A numerical minimum cleanup level for a particular chemical may be designated by selecting one of four alternatives:

(i) The no adverse effects level, which is equal to the sediment quality standard;
(ii) The minor adverse effects level, which is equal to the second lowest AET value;
(iii) The moderate effects level, which is equal to the second highest AET value; and,

(iv) The severe effects level, which is equal to the highest AET value.

By identifying four distinct levels of protection, this procedure allows modification of the regional sediment quality standards that consider technical feasibility, costs, and risks associated with site remediation. In addition, it supports the definition of functional cleanup levels expressed in terms of specific bioassay results.

Chapter 4
A Review of the Major Approaches to the Development of Tissue Residue Criteria and Guidelines

4.0 Introduction

While environmental contaminants which are associated with water and/or sediment may be directly toxic to aquatic life, many of these substances also have the potential to accumulate in the tissues of aquatic biota. Dramatic declines in the populations of predatory birds in the 1960s, which were linked to biomagnification of DDT, provide graphic examples of the potential effects of persistent chemicals in the food chain (Odum 1975). Hence, it is apparent that contaminants in aquatic biota may represent significant hazards when the tissues of these organisms are consumed by fish, other aquatic organisms, and wildlife.

An important step towards the effective management of toxic and bioaccumulative substances is the development of tools which may be used to interpret the significance of contaminants in the tissues of aquatic biota. In Canada, tissue residue guidelines (TRGs) have been proposed as relevant tools for assessing issues and concerns related to the presence of bioaccumulative substances in the environment. Tissue residue guidelines are intended to specify safe levels of contaminants in the tissues of aquatic biota for the protection of human and nonhuman consumers of these organisms. Tissue residue guidelines are particularly important in the Fraser River Basin because many anthropogenic activities are known to release bioaccumulative substances into the ecosystem and because a variety of biomonitoring programs are currently ongoing within the basin.

There are several approaches that have been used by various jurisdictions (Health and Welfare Canada, Environmental Protection Agency/Food and Drug Administration, Wisconsin Department of Natural Resources, New York State Department of Environmental Conservation) to evaluate the significance of contaminant residues in aquatic biota. Most of these approaches are designed to address human health concerns related to the consumption of contaminated food products (Huston 1988; USEPA 1989b). Only one of these approaches directly supports the derivation of numerical tissue residue guidelines for the protection of wildlife species that consume aquatic biota (Newell *et al.* 1987). The CCME has now established a formal protocol for deriving TRGs for the protection

of wildlife (Walker and MacDonald 1993); however, no CCME-approved guidelines have yet been promulgated.

4.1 The Health and Welfare Canada Approach

In Canada, the Health Protection Branch of Health and Welfare Canada (HWC) is responsible for assessing the potential risks to Canadians associated with dietary exposures of potentially toxic substances. When concerns are raised about contaminant residues in specific food products (including aquatic biota), HWC may assess the human health risks associated with ingestion of the contaminated food. A prerequisite for applying the regulatory provisions of the Food and Drug Act is that the food product must be offered for sale. Therefore, fish caught by anglers for their own use are not covered by the provisions of the Act. In practice, however, Health and Welfare Canada generally supports the management decisions taken by other regulatory agencies (i.e., fishery closures promulgated by Fisheries and Oceans Canada and provincial agencies) by conducting human health risk assessments even when the food products are not likely to be offered for sale (Huston 1988).

Assessment of the human health risks associated with exposures to a specific toxic substance in food is a three step process. HWC assembles and evaluates all available dose-response data from acceptable mammalian toxicity studies. These data are then used to calculate a tolerable daily intake (TDI; in $mg \cdot kg^{-1} \cdot day^{-1}$) of the contaminant by dividing the no observed adverse effects level by a safety factor. Typically, the safety factors range from 100 to several thousand, depending on the quality and quantity of toxicity data (Huston 1988). A cancer risk assessment may be conducted if the substance is known or suspected to be carcinogenic.

The second step in the evaluation involves the calculation of a probable daily intake (PDI) of the contaminant by humans. The PDI is estimated by considering the exposure from all potential sources (including air, water, and food) and a number of cultural, socio-economic and geographic factors. These factors are intended to account for differences in the consumption of contaminated food products by specific segments of the population. For example, First Nations groups consume relatively large qualities of fish and shellfish (P. Quaw. Lheit-Lit'en Nation. Prince George, British Columbia. pers. comm.). The TDI is then compared to the PDI to determine if significant risks to human health are likely to be associated with the consumption of the contaminated food. When the

PDI exceeds the TDI, specific risk management options (i.e., fishery closures, consumption advisories, etc.) may be considered by the agency responsible (Huston 1988).

Under certain circumstances, HWC may recommend numerical tolerances or guidelines for chemical contaminants that tend to accumulate in food products. (These tolerances should not be confused with the Maximum Residue Limits which apply to substances that are intentionally added to food products). Tolerances are legally enforceable limits on the concentrations of contaminants in food products, which are promulgated under the Canadian Food and Drug Act. (C. Charbonneau. Health and Welfare Canada. Ottawa, Canada. pers. comm.). These tolerances specify levels of chemicals in food products below which no significant health hazard would be presented to consumers (Huston 1988). When insufficient data are available to support the derivation of formal tolerances, informal guidelines may be developed to provide environmental managers with relevant information to support regulatory actions. These regulatory actions would be designed to limit exposures of consumers to contaminants that may be present in the Canadian food supply. Guidelines are continuously under review and may be revised in light of new toxicological and exposure information. The information used to calculate the TDI, in conjunction with data on body weight and daily intake rates of fish and shellfish, form the basis for recommendation of Canadian tolerances and guidelines for chemical contaminants in food products.

4.2 The United States Environmental Protection Agency/Food and Drug Administration Approach

In the United States, evaluation of information on contaminant residues in the tissues of aquatic organisms is a complex process, that may involve several levels of government and numerous regulatory agencies. The Environmental Protection Agency (USEPA) and the Food and Drug Administration (FDA) share responsibility for the regulation of contaminants in food that move in interstate commerce. However, when seafood products are used entirely within the state of their origin, state agencies may also play a significant role in the evaluation of contamination issues. The following discussion has been abstracted from a guidance manual published by the USEPA (1989b) entitled, *'Assessing Human Health Risks from Chemically Contaminated Fish and Shellfish'*.

Under the Federal Food, Drug, and Cosmetics Act (FFDCA), FDA has the primary responsibility for assuring the safety of the food supply, including fish and shellfish. As such, FDA is responsible

for establishing safe levels of chemical substances (except pesticides) that have the potential to contaminate food products. In fulfilment of its responsibilities under the FFDCA, FDA may develop formal tolerances, which establish legally enforceable limits on the levels of chemical contaminants in foods. However, action levels may also be established to support regulation of contaminants in food products. Action levels are similar to tolerances, except that they are intended for interim periods and can be instituted and changed more quickly than tolerances. With respect to pesticide residues in food, USEPA has the lead role in the establishment of tolerances and action levels.

Tolerances and action levels are established using the toxicological data which is available on the substance under consideration. Evaluation of data on acute, chronic, reproductive, teratogenic, mutagenic, carcinogenic, and no-adverse effects provides a basis for establishing a reference dose (RfD) of the substance. The RfD (in $\mu g \cdot kg^{-1} \cdot day^{-1}$) is an estimate of the daily intake rate causing no adverse effects over a long period of exposure (often the lifetime of the consumer is considered in this calculation). The RfD is derived from an observed threshold dose [a no observed adverse effect level (NOAEL) or lowest observed adverse effect level (LOAEL) if the NOAEL is indeterminate] in a chronic animal bioassay by applying an uncertainty factor. The uncertainty factor may vary from 1 to 1000 or more, depending on the availability of data on human toxicology, on sublethal effects, on sensitive endpoints, and on long-term exposures.

The reference dose forms the basis of the action level development process; however, information is also required on the body weight (BW) of the consumer and their daily consumption of fish and other seafood products (food intake rate; FIR). Since consumers may be exposed to toxic substances by a variety of routes (i.e., air, water, food, and dermal exposure), information is required on the relative contribution of each (i.e., percent of total exposure). An estimate of the percentage of the total exposure attributable to the consumption of contaminated foods (percent food contribution; PFC) is used in the derivation of action levels for contaminant residues, as follows:

$$AL = [RfD \cdot BW \div FIR] \cdot PFC$$

where:

AL = Action Level (in $mg \cdot kg^{-1}$),

RFD = Reference Dose (in $mg \cdot kg^{-1}$ $BW \cdot day^{-1}$),

BW = Body Weight (in kg),

FIR = Food Intake Rate (in $kg \cdot day^{-1}$),

PFC = Percent Food Contribution.

While action levels and tolerances provide a consistent means of evaluating the risks to human consumers associated with chemical contaminant residues in fish and shellfish, these regulatory tools do not apply to food products that are not involved in interstate commerce. In addition, both tolerances and action levels are developed using information on the average human consumption rates of fish and shellfish, calculated on a nation-wide basis. Therefore, tolerances and action levels may not be directly applicable to the evaluation of contaminant concerns in lakeshore or coastal areas, where consumption rates of fish and other seafood products are known to be much higher than national consumption rates.

To account for the recognized deficiencies of tolerances and action levels, USEPA and a number of states (i.e., California and Washington) have adopted risk assessment approaches to the evaluation of health risks related to the consumption of contaminated fish and shellfish. These risk assessments are similar to those undertaken by Health and Welfare Canada. In the United States, however, consumers are also provided with information that facilitates estimation of the health risks that are associated with the consumption of specific quantities of contaminated seafood products. This ancillary information is provided to allow consumers of aquatic organisms to assess their individual level of risk, based on the quantity of specific seafood products that they consume. The reader is directed to specific publications, such as Pollack *et al.* 1990, USEPA 1989b, and Keenan *et al.* 1990, for more information on the risk assessment procedures used in the United States.

4.3 The New York State Department of Environmental Conservation Approach

The New York State Department of Environmental Conservation has developed a procedure for estimating safe levels of contaminants in the tissues of aquatic biota for the protection of piscivorous wildlife species near the Niagara River (Newell *et al.* 1987). The approach relies on dose-response information from studies on laboratory animals (and wildlife species, when available) and on biological data on the wildlife species being considered for protection.

Newell *et al.* (1987) developed two separate but related procedures for deriving numerical tissue residue guidelines (TRGs). For non-carcinogenic substances, the TRG is based on an estimation of the safe daily dose of the toxicant for wildlife species (termed the wildlife NOEL). The wildlife NOEL may be calculated from acute or chronic LOELs or from the chronic NOELs reported for laboratory animals, in conjunction with appropriate application and uncertainty factors. Safety

and/or uncertainty factors are selected considering the information that is available during the toxicological assessment, and are explicitly defined in the methodology. Only data on mammals are used to extrapolate to mammalian wildlife species. Similarly, only avian data are used to extrapolate to wildlife bird species. Where available, the results of feeding studies on wildlife species are incorporated into the database to provide information on the relative sensitivity of these species.

Tissue residue guidelines for non-carcinogenic substances are derived from the wildlife NOEL by considering the body weights and daily intake rates of target wildlife species. Using this procedure, TRGs are derived as follows:

$$TRG = NOEL \times BW \div FIR$$

where:

TRG = Tissue Residue Guideline (in $mg \cdot kg^{-1}$);
NOEL = No Observed Effects Level for Avian or Mammalian Wildlife Species (in $mg \cdot kg^{-1} \cdot day^{-1}$);
BW = Body Weight (in kg); and,
FIR = Food Intake Rate (in $kg \cdot day^{-1}$).

Newell *et al.* (1987) also developed a procedure for deriving tissue residue guidelines for carcinogenic substances. This procedure relies on quantitative cancer risk assessments that have been developed for mammalian receptors (primarily rodents). First, a one in one hundred increased cancer risk dose ($CRD_{10^{-2}}$) is calculated from the one in one million increased cancer risk dose ($CRD_{10^{-6}}$). This cancer risk dose is then converted to a wildlife dietary guideline by considering the body weights and daily food intake rates of wildlife species, as follows:

$$TRG = CRD_{10^{-2}} \times BW \div FIR$$

where:

TRG = Tissue Residue Guideline ($mg \cdot kg^{-1}$);
$CRD_{10^{-2}}$ = 1 in 100 Cancer Risk Dose ($mg \cdot kg^{-1} \cdot day^{-1}$);
= $CRD_{10^{-6}} \times 10\,000$;
BW = Body Weight (kg); and,
FIR = Food Intake Rate ($kg \cdot day^{-1}$).

The tissue residue guideline for the target species may be selected from the results of either of the two derivation procedures (i.e., wildlife NOEL or the cancer risk dose procedures). A final tissue residue guideline value may, then, be selected from the guidelines calculated for the various target species. Generally, the guideline for the most sensitive species would be adopted as the final tissue residue guideline.

Chapter 5
Summary and Recommendations

5.0 Summary

A review of the world literature was conducted to identify environmental quality criteria, guidelines, objectives, and standards that could be utilized by the Environmental Quality and Pollution Abatement Task Groups of the Fraser River Action Plan. While no attempt have been made to identify the most appropriate environmental quality guidelines for use in the Fraser River Basin, descriptions of the major approaches to the derivation of these assessment tools have been included in this document to facilitate decision-making activities by Task Group members. It is understood that the CCME-approved guidelines will be used when available and will replace the informal guidelines as they are developed. It should also be noted that approaches utilized by British Columbia and Ontario are functionally similar to those that are recommended by the CCME. Therefore, the resultant criteria and guidelines are likely to be similar to those that will, ultimately, be approved by the CCME.

Summaries of the environmental quality criteria, guidelines, objectives, and standards compiled during this study are contained in the supporting documentation for this report (Haines *et al.* 1994). These summaries are presented by media type and water use. Water quality guidelines for the protection of 1) human health, 2) recreation and aesthetics, 3) aquatic life, 4) agriculture, and 5) industrial water uses have been presented in separate tables. Sediment quality guidelines for the protection of 1) aquatic life, 2) wildlife, and 3) human health are also included in the supporting documentation. Lastly, tissue residue guidelines for the protection of human and wildlife consumers of aquatic biota have been summarized in Haines *et al.* (1994).

One of the major products of this study is an electronic database, entitled the Environmental Assessment Information System (EAIS), that contains all of the environmental quality criteria, guidelines, objectives, and standards that were retrieved during this study. Individual records in this database include the chemical name, media type (i.e., water, sediment, or biota), water type (freshwater or estuarine), water use, Chemical Abstracts Service name (CAS name), CAS number, IUPAC name, and any other names that were identified. In addition, each data record included the guideline value, units, approach, jurisdiction, and the reference (bibliographic citation). Lastly, the

application (or rationale) for each guideline and any supporting information provided on the application of the guidelines (notes) are included in each data record. This information system has been created in Paradox™ format, which facilitates inquiries using a broad range of database searching strategies. Brief instructions on the use of this joint Environment Canada and MacDonald Environmental Sciences Ltd. database are provided in the supporting documentation (Haines *et al.* 1994).

The Environmental Assessment Information System provides a relevant tool for evaluating the results of environmental survey and monitoring programs conducted in the Fraser River Basin. This information system could be improved if it included the marine environmental quality criteria and guidelines. In addition, the system will require periodic updating if it is to remain current and viable. Therefore, it is recommended that the system be updated annually to assure that users are provided with access to the most recent information for assessing environmental quality.

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Table 1. List of Priority Substances in the Fraser River Basin.

Chemical	Chemical	Chemical
1,1,1-Trichloroethane	DDAC	Nitric Acid
1,1,2,2-Tetrachloroethane	Di-n-butyl phthalate	Nitrite
1,2-Dichlorobenzene	Di-n-octyl phthalate	Nitrobenzene
1,2-Dichloroethane	Dibenzo(a,e)pyrene	Nitrosamines
1,4-Dichlorobenzene	Dichlorobenzenes	Nonylphenol
2,4-Dimethylphenol	Dichloroethylene	o-Xylene
3-methylphenol	Dichloroethylether	PAHs
Acenaphthylene	Dichloromethane	PCBs
Acetic acid	Diethyl phthalate	Pentachlorobenzene
Acetone	Dimethyl phthalate	Pentachlorophenol
Acrylonitrile	Dimethylnaphthalenes	Phenanthrene
Aluminum	Dimethylphenanthrene	Phenol
Ammonia	Dioxane (1,4)	Phthalate esters
Aniline	Dioxin	Propylene dichloride
Anthracene	Ethylbenzene	Pyrene
Antimony	Ethylene dichloride	Residual chlorine
AOX	Ethylene Oxide	Resin acids
Arsenic	Fluoranthene	Selenium
Barium	Fluorene	Silver
Benz(a)anthracene	Formalydehyde	Sodium hydroxide
Benzene	Furan	Styrene
Benzo(a)flouranthene	Hexachlorobenzene	Sulphate
Benzo(a)pyrene	Hexane	Sulphide
Benzylbutyl phthalate	Hydrochloric Acid	Tetrachlorobenzenes
Beryllium	Hydrogen fluoride	Tetrachloroethylene
Bis(2-ethylhexyl)phthalate	IPBC	Titanium
Boron	Iron	Toluene
Bromoform	Isophorone	Tributyltin
Cadmium	Lead	Trichlorobenzenes
Carbon tetrachloride	m,p-Xylene	Trichloroethylene
Chlorocatechols	Magnesium	Trimethylnaphthalenes
Chloroform	Manganese	Trimethylphenanthrene
Chloroguiacols	Mercury	Xylene
Chlorophenols	Methylchloroform	Zinc
Chromium	Methylnaphthalenes	
Chrysene	Methylphenanthrene	
Cobalt	Molybdenum	
Copper	Naphthalene	
Cyanide	Nickel	

Table 2. List of Jurisdictions for which Environmental Quality Guidelines were Available.

Jurisdiction	Jurisdiction	Jurisdiction
Alaska	Israel	Spain
Alberta	Italy	St. Lawrence River, Canada
Arizona	Japan	Sweden
Arkansas	Kansas	Switzerland
Australia	Kentucky	Tennessee
Brazil	Korea	Thailand
British Columbia	Louisiana	USSR
California	Manitoba	United Kingdom
Canada	Mariana Islands	United States
Chicago, IL	Maryland	Utah
Chile	Michigan	Venezuela
Colorado	Minnesota	Virginia
Delaware	Mississippi	Western Australia
Denmark	Mississippi Sound, MS	Washington
District of Columbia	Missouri	West Virginia
Ecuador	Netherlands	Wisconsin
Europe	Nevada	Zambia
Finland	New York	
Florida	New Zealand	
France	Ohio	
Georgia	Oklahoma	
Germany	Ontario	
Great Lakes	Oregon	
Greece	Phillippines	
Hong Kong	Poland	
Iceland	Puerto Rico	
Illinois	Puget Sound, Washington	
India	Quebec	
Indiana	Rhode Island	
International	Saskatchewan	
Iowa	South Dakota	

Figure 1. Example of a Database Record in the Environmental Assessment Information System

Chemical Name : Methylene Chloride

Media Type : Water Water Type : FW

Use : Human Health CAS Number : 75-09-2

CAS Name : Dichloromethane

Other Names1 : methylene dichloride; methylene bichloride

Other Names2 :

Guideline : 2000 Units : ug/L Approach : TRA

Jurisdiction : Great Lakes Reference : U.S. Environmental Protection Agency 1992

Application : Human Cancer Criterion for nondrinking water

Notes :

Notes2 :

Appendix 1

Overview of the Use of the Environmental Assessment Information System

An Environmental Assessment Information System (EAIS) has been compiled in the database system Paradox For Windows (Version 1.0), which operates within the Microsoft Windows environment on the IBM (or compatible) computer. Paradox for Windows requires a microprocessor of 80386 or higher; a minimum of 4MB RAM; at least 20MB of free disk space; EGA or higher video monitor; Microsoft Windows Version 3.1 or higher. Another 9 MB of disk space are required to load and run the EAIS database. The following information is intended to provide the reader with a brief overview of the EAIS; it was not designed as a user manual. Potential users of the EAIS will require knowledge of the use of databases and spreadsheets in a Windows environment. For advanced operations the user will also require experience in using the Paradox for Windows database.

The EAIS consists of two databases. The first is the EAIS database which holds information on over 400 substances, environmental quality guidelines and their underlying rationale, and jurisdiction (see below for a full listing of fields). The second is the CASNAME database which contains information on each of the substances, their chemical names, chemical abstracts registry numbers, and alternate names. The following files are included:

Database Files:

EAIS.DBF	Environmental Assessment Information System Database
EAIS.VAL	Validity checks and referential integrity for the EAIS table
CASNAME.DBF	Chemical Abstracts Name Database
CASNAME.PX	Primary index of the Chemical Abstracts Database (which sorts the database according to Chemical Name)

Form Files:

EAIS.FSL	Form for viewing the EAIS Database
CASNAME.FSL	Form for viewing the CASNAME Database

Report Files:

EAISREC.RSL	Single record report of the EAIS Database
EAISTBL.RSL	Table report on the EAIS Database
CASNAME.RSL	Single record report of the CASNAME Database

Other Files:

EAISTBL.QBE	Query for genering supporting documentation tables for export to spreadsheet

The Database setups are as follows:

EAIS.DBF

Field Name	Field Type	Field Size
Chemical Name	Alphanumeric	60
Media Type	Alphanumeric	15
Water Type	Alphanumeric	20
Use	Alphanumeric	30
CAS Name	Alphanumeric	235
Other Names1	Alphanumeric	200
Other Names2	Alphanumeric	150
CAS Number	Alphanumeric	20
Guideline	Alphanumeric	30
Units	Alphanumeric	5
Approach	Alphanumeric	6
Application	Alphanumeric	255
Notes	Alphanumeric	255
Notes2	Alphanumeric	255
Jurisdiction	Alphanumeric	25
Reference	Alphanumeric	100

For referential integrity the "Chemical Name" field of the EAIS database is validated against the "Chemical Name" field of the CASNAME database. Therefore a CAS name must exist in the CASNAME database before that chemical and related information can be added to the EAIS database.

CASNAME.DBF

Field Name	Field Type	Field Size
Chemical Name	Alphanumeric	60 *
CAS Name	Alphanumeric	235
Other Names1	Alphanumeric	200
Other Names2	Alphanumeric	150
CAS Number	Alphanumeric	20

*keyed field sorts by Chemical Name

In Paradox, information is stored in tables, which display multiple records at a time. However for the purposes of this database, a form has been created (EAIS.FSL) which allows viewing of full records one at a time.

To open a form you select: File | Open | Form. Then select the EAIS.FSL selection and select OK.

Once the form is open, Paradox displays buttons (icons) at the top of the screen for selecting frequently used commands (such as move, search, cut, copy, etc.). When you place the cursor on the button selection, a prompt appears in the bottom left hand corner of the screen, explaining the function of the button. These selections can also be made using the menu above the button list.

Moving Around the Database in Form View

Tab, Return	Moves one field
Cursor Keys	Travel in the direction of the cursor
Home	Takes you to the top of the form
End	Takes you to the bottom of the form
Ctrl Home	Takes you to the first record
Ctrl End	Takes you to the last record
Page Up	Takes you one record back
Page Down	Takes you one record forward

Locating Records and Values in the Database in Form View

Choose: Record | Locate | Record#, to locate a particular record number in the database.

Choose: Record | Locate | Value; Ctrl Z; or press the Search Value button to locate a particular value in the database.
(You can either position your cursor on the field that you wish to search on or, select the field you wish to search on once the locate value dialog box is open).

When searching for a particular value, Paradox offers the following options:
Match capitalization by choosing the "Case Sensitive" selection
Match exact values by choosing "Exact Match" selection
Broaden your search by selecting "@ and .." and including wildcards in your value field

Wildcards include: "@" which represents any single character; or ".." which represents any value. These wildcards may be place at the end, beginning, middle, beginning and end of the value you are searching.

Editing the data

To **edit data** in Paradox you must be in "edit data" mode. You can start "edit data" mode by pressing the function key F9 or selecting the "edit data" button.

To **edit** a particular **field** you must be in "field view". You can start "field view" by pressing the function key F2 or selecting the "field view" button. While editing a field, cursor keys will move you left, right. The "Home" and "End" keys will locate you to the beginning and end of the field, cursor keys up and down will move one line at a time. If you are not in "field view", typing on a field which already contains data will replace that existing data with the new information being input. Pressing enter will end the Field Edit.

As mentioned above, a link between the "Chemical Name" field of the EAIS and the "Chemical Name" field of the CASNAME databases has been established. This link ensures that a chemical exists and all of the available information on that chemical (i.e., CAS Name, CAS Number, etc.) has been located and input into the CASNAME database before guidelines for that chemical can be entered into the EAIS database. This is useful for consistency in reporting Chemical Names which have multiple common name uses [i.e., dichloromethane and methylene chloride are the same chemical; for the purposes of the EAIS database, the chemical name reported in the Merck Index (Budavari *et al.* 1989) was used as the standard reporting name]. When situated on the "Chemical Name" field and in "edit data" mode, a prompt will appear at the bottom left hand corner of the screen which tells you to press "Ctrl+Space Bar for Lookup Help". Pressing "Ctrl+Space" places you in the CASNAME database temporarily. If the name which is in the "Chemical Name" field of the EAIS database exists in the CASNAME database, Paradox will position the cursor at the matching chemical name. If the name does not exist Paradox will position you at the beginning of the CASNAME database. You can search this lookup table by: typing the first characters of the name you wish to search, at which point Paradox will move according to the characters typed; or you can press "Ctrl Z" and then input your search criteria as explained above in "Locating Records and Values in the Database". Once you have made your selection you are returned to the EAIS database, or if your selection does not exist, choose "Cancel" to return back to the EAIS database. You cannot amend the CASNAME database while in lookup mode.

Contained on the EAIS form is a graphic called a "Push Button", which has been assigned the name "CASNAME". This has been added to the form to aid in adding or searching records in the CASNAME database while working in the EAIS database (this allows you full editing capability which is not available while in lookup mode). Pressing the "CASNAME" push button places you in the CASNAME database. Once in the CASNAME database, you can edit data as described in the "Editing the Data" section above. Once you have completed editing the CASNAME database there is a push button named "Back to Database" which returns control back to the EAIS database.

Querying the database

Querying the database allows you to isolate records that match a particular value or set of values. When you query a database, the results of that query are placed in a Table called Answer. You can select all the fields of the database or simply the fields which are of interest to you. If you have selected all of the fields for the database you can access any of the forms and reports that have been created for EAIS. This can be accomplished by (for example opening a form): File | Open | Form; Choosing the "Change Table" option; Selecting the Answer Table as the new table to use with the existing form.

Options in Queries

Paradox uses check marks to select the fields. Placing a check mark in the database field box (which appears under the tables title in the query image) allows you to select all the fields of the database. You can also select specific fields by placing a check mark in the field check box. A single check mark selects unique values for a particular field. A check plus selects all the values for a particular field. "Examples" may be placed in the query, for example, by typing the word "water" in the Media Type field, only the records for "water" will be selected. You can place "examples" in a combination of fields to get an "and" result or include the word "or" to select multiple values (i.e., "water or sediment"). You can also use Paradox wildcards (i.e., "wat.." to select water.) You can also have multiple lines to specify "or" criteria. When using multiple lines, check marks must be placed in the same fields on the second line that have check marks in the first (i.e., if the "Media Type" field is checked in the first line it must also be checked in the additional lines regardless of whether "Media Type" is part of your additional search strategy).

Printing Reports

Paradox has the ability to create default formats for reports; however, due to the length of the fields and the number of records contained in the EAIS database, this may not be the best way to report on the data. For this reason, a report format (EAISTBL) has been designed to enable printing of the database. This report file groups the data according to "Media Type" and "Use" and sorts the data according to "Chemical Name". The output format is in table form and includes all of the fields contained in the Supporting Documentation. There is also a report called "EAISREC", which reports data one record at a time (see Figure 1). You can report all of the data contained in the database or you can query the database to isolate specific sets of data and then print only those queried data.

To generate the tables for the supporting documentation of this report the EAIS database was queried, and the information on "Media Type" and "Use", was selected. Subsequently, these data were sorted by the "Chemical Name". The data was then exported to Microsoft Excel, where custom formatting and Table numbers were assigned. For flexibility, working with spreadsheets in Excel or another compatible program is the preferred method. For straight output not requiring custom formatting, Paradox reporting is acceptable.