CANADIAN SEDIMENT QUALITY GUIDELINES FOR CHROMIUM

Supporting Document

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ABSTRACT

The following document provides a comprehensive review of the available scientific literature on chromium in aquatic sediments. The physical and chemical properties of chromium, its production and uses, and the sources, fate, behaviour and toxicity of chromium in both freshwater and marine (including estuarine) are discussed. A thorough review of the concentrations of chromium found in Canadian sediments (including natural background concentrations) is provided.

Using the weight of evidence of available toxicological information, the modified National Status and Trends Program Approach was used to derive interim sediment quality guidelines for total chromium of 37.3 $mg \cdot kg^{-1}$ (dry weight) and 52.3 $mg \cdot kg^{-1}$ (dry weight) for the protection of freshwater and marine (including estuarine) life, respectively. In addition, the Probable Effect Levels (PELs), above which adverse biological effects are expected to occur frequently, were calculated to be 90.0 $mg \cdot kg^{-1}$ (dry weight) and 160 $mg \cdot kg^{-1}$ (dry weight) for freshwater and marine (including estuarine) sediments, respectively. The interim sediment quality guidelines and PELs are compared to those that have been developed for other jurisdictions in the northern hemisphere. In addition their relevance with respect to the available literature on the bioavailability, bioaccumulation, biomagnification and toxicity of chromium in sediments is addressed.

GLOSSARY OF ACRONYMS AND ABBREVIATIONS

AAS	Atomic Absorption Spectrophotometry
AET	Apparent Effects Threshold
AETA	Apparent Effects Threshold Approach
AOSERP	Alberta Oil Sands Environmental Research Program
ARCS	Assessment and Remediation of Contaminated Sediments
ASTM	American Society for Testing and Materials
ASV	Anodic Stripping Voltammetry
AVS	Acid Volatile Sulphide
BAF	Bioaccumulation Factor
BEDS	Biological Effects Database for Sediments
CCME	Canadian Council of Ministers of the Environment
COA	Co-occurrence Approach
Cr	Chromium
Cr(III)	Trivalent Chromium
Cr(VI)	Hexavalent Chromium
dw	dry weight
E, *	Effect
EC	Effective Concentration
EC_{50}	Median Effective Concentration
EC/MENVIQ	Environnement Canada et Ministère de l'Environnement du Québec
Eh	Electron Potential
EMAP	Environmental Monitoring and Assessment Program
EMB	Embryo
EqP	Equilibrium Partitioning
EqPA	Equilibrium Partitioning Approach
GSC	Geological Survey of Canada
ICP	Inductively Coupled Plasma
ICP-AES	Inductively Coupled Plasma - Atomic Emission Spectrometry
IEPA	Illinois Environmental Protection Agency
IJC	International Joint Commission
ISQG	Interim Sediment Quality Guideline

1	K_d	Partition Coefficient
2	LC	Lethal Concentration
3	LC_{50}	Median Lethal Concentration
4	LOEL	Lowest Observed Effect Level
5	MBI	Macroinvertebrate Biotic Index
6	MISA	Municipal/Industrial Strategy for Abatement
7	NAA	Neutron Activation Analysis
8	NC	No Concordance
9	NE	No Effect
10	NG	No Gradient
11	NGR	National Geochemical Reconnaissance
12	NIVA	Norwegian Institute for Water Research
13	NOAA	National Oceanic and Atmospheric Administration
14	NR	Not Reported
15	NSTP	National Status and Trends Program
16	NSTPA	National Status and Trends Program Approach
17	OES	Optical Emission Spectroscopy
18	OME	Ontario Ministry of the Environment
19	OSPAR	Oslo and Paris Commission
20	PEL	Probable Effects Level
21	PSDDA	Puget Sound Dredged Disposal Analysis
22	RGS	Regional Geochemical Survey
23	SAIC	Science Applications International Corporation
24	SEM	Simultaneously Extracted Metals
25	SG	Small Gradient
26	SSTT	Spiked Sediment Toxicity Test
27	TEL	Threshold Effect Level
28	TFS	Total Free Sulphide
29	TOC	Total Organic Carbon
30	TS	Total Sulphide
31	USEPA	United States Environmental Protection Agency
32	WEA	Weight of Evidence Approach
33	WDE	Washington State Department of Ecology
34	WHO	World Health Organization

CHAPTER 1. INTRODUCTION

The conservation and protection of aquatic and terrestrial ecosystems, and the maintenance, protection and restoration of a high level of environmental quality are priority environmental management goals (CCME 1990). Achievement of these goals requires the availability and use of practical scientific tools to assess the quality of the environment. Canadian environmental quality guidelines are one such scientific tool. The Canadian Council of Ministers of the Environment (CCME) began developing nationally consistent guidelines for environmental quality in Canada (CCREM 1987) in response to a growing public concern that chemical substances entering the environment are placing ecosystem and human health at risk (Gaudet et al. 1995). The development of guidelines for water, tissue, soil, and sediment quality is a key component of Canada's commitment to interjurisdictional cooperation (i.e., federal, provincial and territorial) and is also one of the functions and duties described in the Canadian Environmental Protection Act (CEPA 1988).

Canadian sediment quality guidelines for the protection of aquatic life are developed for individual chemicals for both freshwater and marine (including estuarine) sediments by the Water Quality Guidelines Task Group of the CCME. These guidelines are numerical limits or narrative statements recommended to support and maintain aquatic life associated with bed sediments and are developed from the available scientific information on the biological effects of sediment-associated chemicals. The procedures used in deriving Canadian sediment quality guidelines for the protection of aquatic life are described in an earlier document (CCME 1995).

Sediment quality guidelines are scientific benchmarks, or reference points, that can be used nationally in making decisions regarding the protection, evaluation, and/or enhancement of sediment quality. They can be used to help set targets for sediment quality that will sustain aquatic ecosystem health, they assist in the interpretation of sediment chemistry data (particularly the assessment of sediment quality with respect to the protection of aquatic life), and they play an important role in the development of site-specific sediment quality objectives. Sediment quality objectives are defined as numerical limits or narrative statements that are established to protect and maintain designated uses of the aquatic environment at a particular site. Sediment quality guidelines are intended to be used with other relevant information. For example, complementary

information on background concentrations of natural substances and results of site-specific biological assessments should be considered during the implementation of sediment quality guidelines. CCME (1995) and Chapter 6 of this document briefly discuss the use of sediment quality guidelines along with other relevant information in order to support practical and informed decision-making regarding the management of sediment quality.

This document describes the development of Canadian freshwater and marine (including estuarine) sediment quality guidelines for chromium. In addition to describing the development of sediment quality guidelines for chromium, this document provides a discussion of sources of chromium to Canadian aquatic systems, chromium concentrations in Canadian sediments, the fate and behaviour of chromium in aquatic systems, and the toxicity of sediment-associated chromium to aquatic life associated with bed sediments. Information relevant to both freshwater and marine systems is presented. The information presented in this review is not used directly in the calculation of sediment quality guidelines (i.e., the guidelines are developed using toxicological data only); however it is critical in ensuring that the sediment quality guidelines for chromium are recommended concentrations that are consistent with our current understanding of this substance. In addition, this review provides users of sediment quality guidelines with information (e.g., analytical methods, naturally-occurring concentrations, bioavailability) that may need to be considered during their implementation.

Physical and Chemical Properties

Elemental chromium (CAS Reg. Number 7440-47-3) is a steel-grey, lustrous, brittle metal (Gauglhofer and Bianchi 1991) that is a member of group VIb of the periodic table of elements. It has an atomic number of 24, a specific gravity of 7.18 or 7.19 at 20°C (Hartford 1979; WHO 1988), a melting point of 1 850 to 1 900°C (Hartford 1979; WHO 1988; Gauglhofer and Bianchi 1991; Environment Canada 1993) and is sparingly soluble in water (Murray et al. 1983). Chromium has four naturally occurring isotopes, ^{50}Cr, ^{52}Cr, ^{53}Cr and ^{54}Cr, with relative abundances of 4.31% , 83.76%, 9.55% and 2.38%, respectively (Hartford 1979; Katz and Salem 1994). Thus, the atomic weight of chromium is 51.996 atomic mass units. The most stable of the man-made chromium

isotopes is ^{51}Cr with a half-life of 27.8 days (Environment Canada 1993; Katz and Salem 1994). A summary of the physical and chemical properties of chromium and selected chromium compounds is given in Table 1.

Production and Uses

Chromium is the twenty-first most abundant element in the earth's crust (Hartford 1979; Gauglhofer and Bianchi 1991; Miller-Ihli 1992) with an average concentration in rocks varying from less than 7 $mg \cdot kg^{-1}$ in granitic igneous and quartz rocks to 2 000 $mg \cdot kg^{-1}$ in ultramarine/basic and serpentine rocks (WHO 1988; Miller-Ihli 1992; Katz and Salem 1994). Average soil concentrations are ca. 100 $mg \cdot kg^{-1}$ (Miller-Ihli 1992). Although approximately 40 chromium-containing minerals are known, the only one of economic importance is chromite, which has the ideal composition of $FeCr_2O_4$ (i.e., 68% Cr_2O_3 and 32% FeO or ca. 46% chromium by weight) (Hartford 1979; WHO 1988; Miller-Ihli 1992; Perron 1992). The chromium to iron ratio varies greatly in nature, leading to chromium ores (e.g., $(Fe, Mg)O(Cr,Fe,Al)_2O_3$) being classified according to their elemental composition and industrial use (Table 2).

In Canada, there are more than 250 documented occurrences of chromite deposits, with ore resources estimated to be about 20 million tonnes (Environment Canada 1993). The principal deposits occur in Québec, Ontario, British Columbia, and Newfoundland. Although domestic chromium ores have been mined in the past, Canada has not exploited its chromium deposits since 1945 (Environment Canada 1993). However, in view of recent developments in plasma smelting, uncertainties over foreign supplies, the potential for low cost production due to low energy costs, and increases in price, there has been some effort directed toward developing chromite deposits in parts of Canada (Perron 1992).

Imports of chromium from all foreign sources (e.g., the United States, Philippines, South Africa, Finland) amounted to 54 372 tonnes, 78 247 tonnes and 86 205 tonnes in 1992, 1993, and 1994, respectively (Statistics Canada 1992, 1993, 1994). Chromium-containing materials fall into three categories according to their industrial uses (see Table 2): 1) metallurgical, 2) chemical, and

Table 1. Physical and chemical properties of chromium and some selected chromium compounds (NA = not available) (from WHO 1988).

Name	Chemical Symbol	Relative molecular mass	Specific gravity* $g \cdot cm^{-3}$	Melting point °C	Boiling point °C	Solubility in water weight %	CAS registry number
Chromium	Cr.	51.996	7.19	1857	2672	insoluble	7440-47-3
Chromium(III)-chloride	$CrCl_3$	158.36	NA	NA	NA	NA	10025-73-7
Chromium(III)-oxide	Cr_2O_3	151.99	5.21	2266	4000	insoluble	1308-38-9
Chromium(VI)-oxide	CrO_3	99.99	2.70	196	NA	62.41	1333-82-0
Potassium-chromate(VI)	K_2CrO_4	194.20	2.732	968.3	NA	39.96	7789-00-6
Potassium-dichromate(VI)	$K_2Cr_2O_7$	294.19	2.676	398	NA	11.7	7778-50-9
Calcium-chromate(VI) dihydrate	$CaCrO_4$ $2H_2O$	192.09	NA	1025	NA	3.5	13765-19-0
Calcium-chromium(III) oxide	$CaCr_2O_4$	208.07	4.8	2090	NA	insoluble	NA

* for vapour pressure @ 20°C

Table 2. Composition of chromite ores (NA = not available) (from Hartford 1979 and Miller-Ihli 1992).

Grade	Composition	Ratio Cr:Fe
metallurgical, high Cr.	46% Cr_2O_3 minimum	> 2.1
chemical, high Fe	40 - 46% Cr_2O_3	1.5 - 2.1
refractory, high Al	22 -34% Al_2O_3 33 -38% Cr_2O_3 > 60% $Al_2O_3 + Cr_2O_3$	NA

3) refractory. Metallurgical-grade chromium is used in the production of ferroalloys. The principal chromium-containing chemicals used in Canada include chromium oxide, chromium chloride, and chromium sulphate, which are manufactured from pure sodium dichromate obtained from chemical grade chromite. In addition, small quantities of chromium are used in a diverse array of products including drilling muds, toners for copying machines, cosmetics, magnetic tapes, fertilizers, rubber products, plastics, soaps, and cleaning products (Gauglhofer and Bianchi 1991; Environment Canada 1993; Katz and Salem 1994). Refractory products are made from chromite ores and concentrates to make products such as bricks, mortars, ramming-gun mixtures and high temperature furnace repair material (Moore and Ramamoorthy 1984).

The industrial uses of chromium that are of particular interest and concern with respect to the aquatic environment are: 1) chromite ore processing, 2) cooling towers, 3) metal plating, 4) chromium pigments, 5) metal finishing, 6) leather tanning, 7) wood preservatives, and 8) ferrochromium production (Environment Canada 1993).

Sources to the Aquatic Environment

Chromium is emitted into the atmosphere in approximately equal amounts by natural and anthropogenic sources. For example, Nriagu (1989) estimated that in 1983, about 60% of global atmospheric emissions of chromium were from natural sources whereas about 40% were from anthropogenic sources. The major natural source of chromium to the atmosphere is geologic emissions, such as volcanic dusts and gases (Salem 1989). Other natural sources of chromium, such as airborne sea salt, smoke from forest fires, and biogenic emissions from vegetation, are not important sources of chromium (Miller-Ihli 1992). Anthropogenic sources of chromium to the atmosphere include, but are not limited to, fossil fuel (i.e., coal and oil) combustion, chemical manufacturing, steel production, cooling towers, municipal and sewage sludge incineration, and catalytic control emission devices (Nriagu and Pacyna 1988; Katz and Salem 1994). In Canada, Cormier (1991) has estimated that total discharges of chromium into the atmosphere are 34 tonnes$\cdot yr^{-1}$.

Most of the airborne chromium is associated with particulate matter, indicating that dry deposition is an important source (Bergbäck et al. 1989). Rates of dry deposition of chromium from the atmosphere have been estimated to be 0.001 to 0.03 mg$\cdot m^{-2} \cdot yr^{-1}$ for remote areas to 0.2 to 3.0 mg$\cdot m^{-2} \cdot yr^{-1}$ for rural areas (Bergbäck et al. 1989). Although, chromium is not normally a major

contaminant of urban precipitation, the wet deposition rate in some urban areas can be very high, for example, 48 $mg \cdot m^{-2} \cdot yr^{-1}$ in New York City during the period 1972-74 (Nriagu et al. 1988). Nriagu and Pacyna (1988) calculated that atmospheric fallout of chromium could contribute up to 16×10^6 $kg \cdot yr^{-1}$ to aquatic systems worldwide (assuming that 30% of the chromium emitted into the atmosphere was deposited directly onto the water surface). However, this estimate represents less than 7% of the total chromium inputs of $45 - 239 \times 10^6$ $kg \cdot yr^{-1}$. Scudlark et al. (1994) estimated that the total atmospheric inputs (i.e., wet + dry deposition) of chromium to the Maryland portion of Chesapeake Bay, from June 1990 to July 1991, were 1 500 $kg \cdot yr^{-1}$ (i.e., 0.09 $mg \cdot m^{-2} \cdot yr^{-1}$), which is about an order of magnitude lower than those estimated from fluvial loading into the bay. Similarly, Johnson and Nicholls (1988) found that atmospheric deposition was a negligible source of chromium (i.e., 1% of the total chromium) to Lake Simcoe, Ontario, when compared to dispersion of point source (anthropogenic) inputs in Kempenfelt Bay. Likewise, Johnson (1991) reported that the loading of chromium in runoff from the Ontario rivers in his study was 68.6 times that in precipitation.

By far, the major sources of chromium to aquatic ecosystems are from domestic and industrial wastewater effluents. Coastal marine systems are dominated by inputs of chromium from rivers and, to a lesser extent, by dredging sludges and industrial waste discharges (Nriagu and Pacyna 1988; Moore 1991). In Canada, chromium-containing effluents are generated from a multitude of sources, such as tanneries, cooling towers, steel and non-ferrous foundries, metal finishing and plating operations, flat glass and asbestos producing plants, wood treatment facilities, paint and chemical works, oil drilling, recovery rigs, pulp and paper mills, cement and fertilizer plants, textile mills, power plants, chlor-alkali and asbestos plants, and petrochemical industries (Environment Canada 1993). Total discharges to water and land are estimated at 274 $tonnes \cdot yr^{-1}$ and 4 469 $tonnes \cdot yr^{-1}$, respectively (Cormier 1991).

The ratio of anthropogenic to total input of chromium to aquatic systems, as well as the total flux of chromium to the sediments, are variable. Kemp et al. (1978) estimated that anthropogenic inputs of chromium in the early 1970's averaged less than 13% of the total input of chromium in sediment cores collected from lakes Superior and Huron, including Georgian Bay. Substantially higher anthropogenic contributions of chromium have been measured in Canadian aquatic ecosystems influenced by industrialization. For example, the percentage of chromium from anthropogenic sources was 73% of the total input in Lake Simcoe, Ontario (Johnson and Nicholls 1988), 58% and 63% in Lakes Erie and Ontario, respectively (Kemp and Thomas 1976), 59% in Hamilton Harbour, Lake Ontario (Nriagu et al. 1983), and 36% in Lake St. Clair (Rossmann 1988).

Estimated total fluxes of chromium to sediments worldwide include values of 230 $mg \cdot m^{-2} \cdot yr^{-1}$ and 320 $mg \cdot m^{-2} \cdot yr^{-1}$ in Osaka and Tokyo Bays, Japan, respectively (Goldberg et al. 1976), 161 to 303 $mg \cdot m^{-2} \cdot yr^{-1}$ in a Danish firth, the Netherlands (Jorgensen 1990), 1 to 100 $mg \cdot m^{-2} \cdot yr^{-1}$ in Swedish lakes (Bergbäck et al. 1989), and 81 $mg \cdot m^{-2} \cdot yr^{-1}$ in basins in southern California, U.S.A., (Bruland et al. 1974). In the Great Lakes area, Canada, total fluxes of chromium include values of 80 $mg \cdot m^{-2} \cdot yr^{-1}$ (Kemp and Thomas 1976) to 212 $mg \cdot m^{-2} \cdot yr^{-1}$ (Nriagu et al. 1983) in Lake Ontario, 194 $mg \cdot m^{-2} \cdot yr^{-1}$ in Lake Erie (Kemp and Thomas 1976), and 10 $mg \cdot m^{-2} \cdot yr^{-1}$ and 17.3 $mg \cdot m^{-2} \cdot yr^{-1}$ in Lakes Huron and Superior, respectively (Kemp et al. 1978).

Analytical Methods for Sediments

The determination of total chromium in solid samples (including sediments, soils, sludges, and biota) generally requires an acid digestion of the sample prior to. Methods for dissolving solid samples fall into three main categories: 1) open vessel, 2) bomb digestion, and 3) fusion techniques (Achilli et al. 1989). The heating of sediments in open vessel containers is probably the most common approach for digesting sediments. For analyses requiring the measurement of total chromium, there are several possible acid combinations for dissolving sediments. They include nitric-perchloric (HNO_3-$HClO_4$) (e.g., Ajayi and Mombeshora 1989; Weis et al. 1993), hydrofluoric-nitric-perchloric (HF-HNO_3-$HClO_4$) (e.g., Brumbaugh et al. 1994), hydrofluoric-nitric-hydrochloric (HF-HNO_3-HCl) (Moore 1980), hydrofluoric (HF) (Cantillo and O'Connor 1992), nitric (HNO_3)(Mayer and Fink 1980; Elsokkary and Müller 1989; 1990; Cheung and Wong 1992; Balci and Türkoglu 1993; Coakley and Poulton 1993; Gagnon et al. 1993), nitric-hydrofluoric (HNO_3-HF) (Kurita and Pfeiffer 1991) and also *aqua regia* (i.e., a 3:1 mixture of HCl and HNO_3) (e.g., Mudroch and Capobianco 1979; Nair et al. 1990; Samant et al. 1990; Biney and Beeko 1991; Blais and Kalff 1993; Rowan and Kalff 1993). Generally it is agreed that digestions involving only HCl and/or HNO_3 remove only surface adsorbed or precipitated metals, sulphides, oxides, and carbonates (Oliver 1973; Berrow 1991; Ramelow et al. 1992; Gagnon et al. 1993; Rowan and Kalff 1993) and thus have little effect on chromium bound within the sediment lattice. However, Balci and Türkoglu (1993) reported that a HNO_3 digestion using Teflon flasks resulted in 97% recovery of reference sediment material obtained from the International Atomic Energy Agency laboratory in Monaco. Achilli et al. (1989) compared 5 open vessel digestion methods: 1) HNO_3-HF-$HClO_4$-HCl-H_3BO_3, 2) HCl-HNO_3-HF-$HClO_4$, 3) HNO_3-HCl, 4) HNO_3-HF-H_2SO_4-H_3BO_3 and 5) HNO_3-H_2SO_4 for dissolving marine, estuarine and river reference sediments. They found that for chromium, method 5) (i.e., HNO_3-H_2SO_4) gave the best recovery in marine and estuarine sediments. However, both

methods 2) and 3) were suitable for river sediments provided chromium was present in very high concentrations (i.e., about 30 $mg \cdot g^{-1}$), otherwise only about 40% recovery was obtained. The efficiency of various methods used to recover chromium from the sediments is expected to be a function of the relative distribution of chromium within the sediments.

Common problems with open vessel digestions are sample contamination and loss of volatile components (Bello et al. 1994). These problems can be avoided by conducting all digestions and analyses in ultra-clean laboratories. Alternatively, closed digestion systems such as Teflon bombs (Lacerda et al. 1990; Glahder 1992; Bello et al. 1994; Chovanec et al. 1994) or high pressure microwave decomposition vessels (e.g., Müller et al. 1989; Ramelow et al. 1992; Ankley et al. 1994) can be employed. These systems use acid mixtures similar to those used for open vessel digestions.

Fusion techniques are the third category of sample preparation techniques. Total sediment decomposition can be achieved by fusing the sample with a fluxing agent such as sodium carbonate or lithium metaborate (OME 1991). Fluxing agents are not only capable of dissolving almost all minerals, they also work rapidly and provide extracts that are stable for extended periods of time (Miller-Ihli 1992)

In addition to determining the total amount of chromium in sediments, methods have been suggested for fractionating sediments either physically and/or chemically. For example, sediments can be physically separated according to grain size. The individual analysis of various sediment size fractions for total chromium and other metals is a common practice. Typically, sediments are wet sieved through a 60 or 63 µm mesh with the smaller grain sizes subsequently analysed for trace metals (Damiani et al. 1987; Pestana et al. 1989; Lacerda et al. 1990; Leland and Scudder 1990; Gonçalves et al. 1992; Weis et al. 1993; Fernandes et al. 1994). For estuarine sediment, Förstner (1987) suggests that the analysis of the <20 um grain size is preferable. Alternatively, sediments can be sieved subsequent to drying (Chovanec et al. 1994). For example, the Ontario Ministry of Environment sieves all dry sediments through a 2 mm sieve (ASTM 10 mesh) with the larger size fraction being discarded (OME 1991). Others studies have analyzed a variety of grain sizes and have determined the metal distribution in each fraction (Sager et al. 1990; Deely et al. 1992; Mantei and Sappington 1994; Chovanec et al. 1994). While normalization of total metal concentrations to fine-grained sediments (such as < 63 µm) has been attempted (Benoliel et al. 1988; Cantillo and O'Connor 1992), this procedure may only reduce, not eliminate, the effect of chemically inert, coarse-grained sand (Förstner 1987).

Numerous techniques are also available for the chemical fractionation of chromium and other metals in sediments (see reviews by Förstner 1987 and Campbell and Tessier 1989). It is often useful to know the chemical forms of chromium and other metals in the sediments in order to gain insight into the source and potential fate of metals in the environment. For example, chromium that is associated primarily with the lithogenic sediment fraction would have originated from natural (geologic) sources and would be expected to be relatively unavailable to biota. Alternatively, in theory, chromium recovered in the more easily extractable sediment phases (e.g., iron and manganese oxides), might be more bioavailable. This type of information is lost if only analyses for total chromium are performed.

The chemical fractionation of sediments is defined operationally according to the nature of the reagents used. One of the most widely used sequential fractionation schemes, proposed by Tessier et al. (1979), consists of the analysis of five operationally-defined fractions (Ajayi and VanLoon 1989; Coetzee 1993). A second popular fractionation scheme, developed by Salomons and Förstner (1980), consists of only three operationally defined fractions (Angelidis and Grimanis 1989, Steinberg and Högel 1990). A combination of these two procedures has been proposed by the Commission of European Communities Bureau of Reference and was recently evaluated by Coetzee et al. (1995) and Davidson et al. (1994). This procedure uses three operationally defined fractions: 1) the exchangeable, water and acid soluble phase, extracted with acetic acid (HOAc), 2) the reducible phase (i.e., iron and manganese oxides), extracted using ammoniumhydroxyl-hydrochloride ($NH_2OH{\cdot}HCl$) at pH 2, and 3) the oxidizable phase (e.g., organic matter and sulphides), obtained using peroxide (H_2O_2) followed by ammonium acetate adjusted to pH 2 with acetic acid. Total metal is determined independently using a nitric-perchloric-hydrofluoric acid (HNO_3+$HClO_4$+HF) digestion, with the residual calculated as the difference between total metal - the sum of metal in fractions 1+2+3.

Despite the advantages of using chemical fractionation procedures, the various extraction steps are not always as selective as sometimes hoped (Förstner 1987). Campbell and Tessier (1989) state that some extraction procedures may be compromised by the tendency of metal-containing sulphide phases to be leached progressively, rather than selectively, from the sediment. In addition, fractionation procedures may result in readsorption or redistribution of metals during the different extraction steps (see Campbell and Tessier 1989; Ajayi and VanLoon 1989; Coetzee et al. 1995).

Although sediment solids can be analyzed directly using certain analytical techniques (e.g., X-ray fluorescence), once the sediment is in solution several analytical techniques are available for the determination of chromium. By far the most popular technique is flame atomic absorption spectrophotometry (AAS) as it is fairly fast, straight forward, and affordable (Arafat and Nriagu 1986; Andreev and Simeonov 1988; Achilli et al. 1989; Nair et al. 1990; Miller-Ihli 1992; Balci and Türkoglu 1993; Weis et al. 1993; Bello et al. 1994). Flame AAS is typically used to measure chromium concentrations of 0.1 $\mu g \cdot mL^{-1}$ or greater (Miller-Ihli 1992 - note that a 1 g sediment sample having a chromium concentration of 25 $mg \cdot kg^{-1}$ and diluted to 25 mL would have a chromium concentration of 1 $\mu g \cdot mL^{-1}$). However, in some cases, flame AAS may not be sensitive enough for low level chromium analyses in sediments. Under these circimstances, electrothermal AAS (Ure et al. 1993; Davidson et al. 1994) or graphite furnace AAS (Deely et al. 1992) are the prefered analytical instrument. Graphite furnace AAS offers detection limits of < 1 $ng \cdot mL^{-1}$ (Miller-Ihli 1992) and is the single most widely used method for the determination of low levels of chromium in all sample types (Ruiz et al. 1991; Brumbaugh et al. 1994; Giani et al. 1994). Other techniques for sediment analyses include inductively coupled plasma-atomic emission spectrometry (ICP AES) (Moore 1980; Verbrugge et al. 1991; Coakley and Poulton 1993; Coetzee 1993; Chovanec et al. 1994; Coetzee et al. 1995), X-ray fluorescence (Ajayi and Mombeshora 1989; Rukavina et al. 1990), neutron activation analysis (NAA) (Angelidis and Grimanis 1989) and cold vapour-hydride generation AAS (Benoleil et al. 1988; Lacerda et al. 1990). The Ontario Ministry of Environment (OME 1991) employs ICP-optical emission spectroscopy for their sediment analyses.

A thorough discussion of sampling methods for sediments and interstitial water is beyond the scope of this report and therefore will not be addressed here. However, it is worth mentioning that the terms 'surface' and 'top' sediments are arbitrary and are clearly dependent on the method used for collecting sediment. The surface of sediment cores usually refers to the top one or two cm (Ajayi and Mombeshora 1989; Dave 1992; Rowan and Kalff 1993), whereas the surface of sediment grabs (e.g., Ponar, Ekman, Van Veen, Shipek) can range from the top 2-3 cm (Angelidis and Grimanis 1989; Metcalfe-Smith et al. 1992; Poulton 1992; Deely et al. 1992), 2-5 cm (Coakley and Poulton 1993), 5 cm (Nair and Balchand 1993), or even to the top 10 cm (Kurita and Pfeiffer 1991). Sometimes even greater depths are sampled and homogenized (Mantei and Sappington 1994). Thus when comparing the results of studies conducted on sediments it is critical to ascertain how the sample was collected and to what depth. Environment Canada (1994) has produced a report concerning guidelines for the collection, handling, transport, storage, and manipulation of sediments for chemical characterization and toxicity testing.

CHAPTER 2. CONCENTRATIONS IN CANADIAN SEDIMENTS

Chromium enters the aquatic environment as a result of both natural processes and human activities. Consequently, chromium concentrations in Canadian sediments are dependent on the location of the sediment relative to the source (natural or anthropogenic) of chromium. Concentrations may be higher when the sediment is located close to point source releases of chromium, such as industrial effluents from iron and steel plants, petroleum refineries, pulp and paper mills, and leather tanning industries, as well as discharges from urban stormwater runoff, thermal generating stations, and sewage treatment plants. In addition, chromium concentrations in sediments may be naturally elevated, for example, when sediment overlies chromium-rich rock, such as serpentine. In contrast, sediments in remote areas or those overlying chromium-poor geological formations, may contain relatively low chromium concentrations. At these locations, atmospheric deposition of chromium (e.g., windblown dusts, volcanic emissions, seasalt aerosols, or fossil fuel consumption) directly onto the water surface is likely to be the major source of chromium to the sediments.

The purpose of this chapter is to provide a summary of reported concentrations of chromium in Canadian freshwater and marine (including estuarine) sediments. An effort is made to differentiate between concentrations resulting from natural processes and anthropogenic activities. The identification of natural versus anthropogenically-enriched concentrations of chromium is important in examining the variation in chromium concentrations across Canada and also in evaluating the potential impact of sediment-associated chromium on aquatic biota at specific sites. Organisms living in or on sediments with naturally high chromium levels are likely to be adapted to these levels. Therefore, the site-specific identification of natural versus anthropogenically-enriched concentrations of chromium and other toxic trace elements in sediments is important in ensuring that limited resources are used effectively in assessing and managing sediment quality. Sediment management activities generally focus on areas that have trace element concentrations in sediments exceeding Canadian sediment quality guidelines (or other biological-effects-based assessment tools such as Probable Effect Levels) and that result primarily from human activities as opposed to natural processes.

The sediment-associated chromium concentrations summarized in this chapter are categorized as 'background', 'ambient' or 'contaminated' and are defined as follows:

background: estimates of naturally-occurring concentrations (i.e., no discernable human influence on measured concentrations)

ambient: concentrations that are generally typical of (or represent "baseline" conditions for) a given area and may be the result of human activities, natural processes, or both (i.e., this evaluation could not conclusively identify these data as either "background" or "contaminated" but they likely have at least some anthropogenic component)

contaminated: concentrations that are known to be higher than background concentrations because of human activities

Background concentrations of trace metals, such as chromium, usually are estimated from the analysis of deep sediment layers (i.e., that pre-date European settlement) of sediment cores (Kemp et al. 1978; Nriagu et al. 1983; Johnson et al. 1986; Mudroch et al. 1988; Johnson and Nicholls 1988; Rossmann 1988). Whereas trace metal concentrations in surficial sediment samples may be derived from both natural and/or anthropogenic sources, concentrations in the deep layers of sediment cores generally are thought to be the result of natural processes only. The use of sediment core data for estimating background concentrations is based on the assumption that annual deposits build up to form static vertical sediment profiles and that there is no post-depositional movement of metals. Note, however, that redistributional processes, such as physical and biological mixing, or changes in physicochemical conditions, may result in the migration of trace metals upward or downward through the sediment profile. For example, as discussed in Chapter 3, changes in pH and redox potential at the sediment-water interface can result potentially in the oxidation of Cr(III) to Cr(VI) and its subsequent remobilization and diffusion into the water column (Schroeder and Lee 1975; Nakayama et al. 1981a; Van der Weijden and Reith 1982; Balistrieri et al. 1992; Johnson and Sigg 1992). Although processes such as these can affect the interpretation of sediment core data, generally it is not possible to quantify them because supporting data, such as chromium mass balance budgets or chromium speciation measurements, are not available.

Ambient concentrations of chromium are estimated from surface sediment samples that have been collected with either a sediment corer (e.g., Benthos, Livingstone corer) or a grab sampler (e.g., Ponar grab, Ekman dredge) from remote locations where direct point-source inputs of chromium from anthropogenic activities have not been verified. Ambient concentrations therefore reflect atmospheric sources of chromium (e.g., combustion of fossil fuel), inputs from runoff and upstream discharge from non-point source anthropogenic activities, as well as inputs from natural sources.

Thus, concentrations referred to as ambient in this chapter may include a substantial but unidentifiable anthropogenic component.

Chromium concentrations in surface sediments were identified as being contaminated when: 1) measured chromium concentrations in the surficial sediments were clearly elevated relative to background or ambient concentrations for that location, and 2) the sediment sampling location(s) was in close proximity to a known point source input of chromium. In most cases, a link between the concentration of chromium in the surficial sediments and known anthropogenic inputs at the sampling sites was indicated by the author(s) of the particular study. It should be emphasized that while elevated chromium concentrations in surficial sediments do occur as a result of inputs from point sources associated with human activities, upward movement of chromium through the sediment profile (described above), the presence of chromium-rich geological formations and inputs from atmospheric deposition or other non-point sources can also contribute to increased levels. Howver, these contributions are considered to be small in comparison to those from human activities.

Unfortunately, it is difficult to ascertain what proportion of the sediment metal concentration is natural and what proportion is anthropogenic. This difficulty exists because anthropogenic inputs and natural sedimentary loads can change through time and can vary greatly depending on the nature, grain size distribution, and source of the metal. Because metals, including chromium, from natural and anthropogenic sources accumulate together, their origin is difficult to separate post deposition. As a result, a number of approaches have been used to compensate for grain size and minerological effects on natural metal variability so that the anthropogenic metal can be quantified. These approaches include: 1) granulometric procedures, where the metal analysis is restricted to a specific grain size such as < 63 µm (Luoma 1990; Luoma et al. 1990) or the data are mathematically normalized, post analysis, to the amount (weight %) of a specific size fraction such as < 63 µm (Long et al. 1990), and 2) geochemical procedures where the data are normalized to a conservative element such as aluminum (Johnson and Nicholls 1988; Coakley and Poulton 1993; Gagnon et al. 1993), iron (Rule 1985; Gibbs 1994), scandium and cesium (Ackermann 1980) or lithium (Loring 1990, 1991). Using the geochemical approach, it is assumed that the relationship between the normalizing element and the metal (e.g., chromium) is linear, that is, the ratio between the concentration of the normalizing element and chromium is constant through time. Geochemical normalization has the advantage over granulometric procedures because it reduces metal variability caused by both grain size differences and also mineralogy. In addition, anomalous metal contributions can be identified.

The data summarized in this chapter were collected for a variety of purposes using a variety of sampling methods. Although an attempt has been made to categorize these data consistently according to the above definitions, all of them may not be comparable. For example, many of the background concentration data for freshwater sediments were obtained from the National Geochemical Reconnaissance (NGR) Program Open Files (see Friske and Hornbrook 1991 for a description of the NGR Program). During this program, lake sediments were collected using a hollow-pipe, bottom-valved sampler and the top few centimetres of sediment were washed out of the core barrel. Background chromium concentrations were measured on the remaining organic-rich material called gyttja. Traditionally, however, background concentrations of metals, including chromium, in freshwater sediments are derived from sectioned sediment cores and the background concentration is estimated from the analysis of sediment layers that pre-date European settlement (Kemp et al. 1978; Nriagu et al. 1983; Johnson et al. 1986; Mudroch et al. 1988; Johnson and Nicholls 1988; Rossmann 1988). This requires that the sediment cores be dated using either marker (e.g., ^{137}Cs, *Ambrosia* horizon) or radiogenic (e.g., ^{210}Pb) techniques. In contrast, estimates of ambient and contaminated chromium concentrations in both freshwater and marine sediments were assumed to be represented by the top 1 to 5 cm of both grab samples (e.g., Ekman, Shipek, Ponar) and core samples (e.g., K-B, triple benthos) unless otherwise noted.

For several studies, the chromium concentrations in the fine-grained sediment fractions (e.g., less than < 63 μm) were reported, while in other studies, chromium concentrations in whole sediment samples (i.e., all size fractions) were given. Because chromium tends to be associated with fine-grained materials such as silts and clays (Oliver 1973; Willey 1976; Loring 1979a; Rossmann 1988; Duzzin et al. 1988; Sager et al. 1990; Anderson 1990; Coakley and Poulton 1993), its concentration will vary inversely with differences in sediment texture and grain size (Willey and Fitzgerald 1980; Loring 1982,1990,1991). Thus, data reported for different size fractions will not be directly comparable unless they refer to standard grain sizes. For the purpose of this summary chapter, chromium concentrations refer to whole sediments, unless otherwise noted.

Analytical methods also varied among the studies, particularly the acid digestion techniques used in the quantification of chromium in the sediments. Although concentration data reported in this chapter were included only if standardized total (e.g., hydrofluoric acid or the sum of the concentrations measured in sequentially-extracted sediment fractions) or near-total digestion methods (e.g., *aqua regia*, hydrochloric acid, nitric acid, boric acid and perchloric acid) were used, it should be emphasized that digestions that include the residual fraction (hydrofluoric acid

digestion) may not be comparable to weak/other acid digestions. Considering that a large fraction of chromium is often associated with the residual fraction (i.e., part of the mineral lattice) (Samant et al. 1980), for this review, an attempt is made to compare only those data that have been obtained using similar analytical or digestion techniques. In addition, all concentrations are reported on a dry weight basis.

Freshwater Sediments

Background Concentrations

A summary of background concentrations for chromium in Canadian freshwater sediments, obtained from the NGR program of the Geological Survey of Canada (GSC), is given in Table 3. The data collected during this program provides background concentrations of chromium for lake and stream sediments in Newfoundland, Labrador, Ontario (excluding the Laurentian Great Lakes and St. Lawrence River), Saskatchewan, British Columbia, and the Yukon. Estimates of background chromium concentrations in sediments from other sources (Kemp et al. 1978; Johnson et al. 1986; Rieberger 1992) and also other parts of the country (North West Territories, Allan 1975; Nova Scotia, Mudroch and Sandilands 1978) and the Laurentian Great Lakes are summarized in Table 4. No background concentrations of chromium were available for New Brunswick, Prince Edward Island, Alberta, or Manitoba.

The NGR program was initiated in 1973, although chromium was not included in the sediment analyses until 1990 (Friske and Hornbrook 1991; Painter et al. 1994). Nonetheless, the data collected during the NGR program (Table 3) provide a consistent and reliable database for background chromium concentrations in freshwater sediments across Canada. During the NGR program, and also in conjunction with provincially-operated surveys conducted using NGR protocol, sediment samples were collected from almost 180 000 freshwater sites, covering approximately 2.1 million km^2, or 25% of the total area of the country (Friske and Hornbrook, 1991). The database includes chromium analyses for 30 528 lake sediment samples and 12 815 stream sediment samples.

It is evident from Tables 3 and 4 that background concentrations of chromium in the sediments of Canadian freshwater lakes and streams vary considerably across the country. Background estimates obtained from the NGR program alone (Table 3) differ by several orders of magnitude,

Table 3. Background chromium concentrations ($mg \cdot kg^{-1}$ dry weight) in Canadian freshwater sediments summarized from data from geochemical reconnaissance programs. National Geochemical Reconnaissance (NGR) data summaries provided by P.W.B. Friske, GSC, Ottawa (1994).

Location	Mean ($mg \cdot kg^{-1} \pm$ S.D.)	Range[1] ($mg \cdot kg^{-1}$)	95th Percentile	Sample Size	Geographic Coverage[2]	Reference
Lakes						
Newfoundland	42.33 ± 159.6	10.0 - 6 120	100	16 569	100%	Nfld Geological Survey
Labrador	59.6 ± 40.8	10.0 - 320	140	2 917	~10%	NGR Open Files
Ontario	48.6 ± 33.5	10.0 - 520	110	7 343	~15%	NGR Open Files
Saskatchewan	30.7 ± 21.0	10.0 - 320	66	3 238	~20% of N. Saskatchewan	NGR Open Files
British Columbia	22.3 ± 20.8	4.0 - 230	51	461	0.4% Nechako Plateau (central B.C.)	2 NGR Open Files[3]
Streams						
Ontario	108.2 ± 70.0	10.0 - 620	240	200	< 1%	NGR Open Files
British Columbia	42.7 ± 36.9	5.0 - 528	102	4 367	~10%	38 NGR/RGS [4] Open Files
Yukon	86.1 ± 71.1	10.0 - 3 000	130	8 248	~30%	20 NGR Open Files

[1] the 10.0 $mg \cdot kg^{-1}$ concentration represents a default value chosen as half the detection limit for chromium
[2] based on a sampling density of 1 sample/~13 km^2
[3] mean and 95th percentile are from 1 Open File only
[4] NGR/RGS = joint federal/provincial (Regional Geochemical Survey) study

despite the standardized collection and analytical procedures used within the program. Background concentrations range from < 20 $mg \cdot kg^{-1}$ (the detection limit for chromium) in all areas of Canada to 6 120 $mg \cdot kg^{-1}$ for lake sediments in Newfoundland and 3 000 $mg \cdot kg^{1}$ for stream sediments in the Yukon. Thus, the NGR program data provide information on naturally-occurring variation in lake and stream sediment chromium concentrations. The high background levels measured in Newfoundland lake sediments is not surprising given that parts of the province are underlain by intrusive gabbro and serpentine rocks of the Cenozoic, Mesozoic and Paleozoic eras (Matthews and Morrow 1995). Furthermore, the chromite ore deposits in this province are among the highest grade (up to 53% Cr_2O_3) found in Canada (Phillips 1989). Lower grade deposits are found elsewhere, primarily in British Columbia, Québec, Manitoba and Ontario (Phillips 1989; Perron 1992), where outcroppings of serpentine, gabbros and other mafic igneous rocks (such as

basalts) also occur. For example, the Interior Plateau region of British Columbia has many ferromagnesian intrusive rocks that are associated generally with chromite and nickel (Rieberger 1992).

Table 4. Background chromium concentrations ($mg \cdot kg^{-1}$ dry weight) in Canadian freshwater lake sediments exclusive of the NGR data reported above.

Location	Range of Means ($mg \cdot kg^{-1}$)	Reference
Ontario	31.8 ± 11.1 to 92	Kemp et al. (1978), Mudroch (1980), Johnson et al. (1986), Mudroch et al. (1988), IJC (1988), Johnson and Nicholls(1988), Snetsinger (1993)
Québec	> 100	Blais and Kalff (1993)
Nova Scotia	~4 to 22	Mudroch and Sandilands (1978)
NWT	> 60	Allan (1975)
Lake Athabasca, Alberta	~ 30	Bourbonniere et al. (1996)

Despite the wide variation in measured background concentrations of the NGR program, average chromium concentrations in both lake and stream sediments are surprisingly similar across the country, ranging between 22.3 ± 20.8 $mg \cdot kg^{-1}$ in British Columbia lakes to 59.6 ± 40.8 $mg \cdot kg^{-1}$ in Labrador lakes, and 42.7 ± 36.9 $mg \cdot kg^{-1}$ in British Columbia streams to 108.2 ± 70.0 $mg \cdot kg^{-1}$ in Ontario streams. The general trend of higher chromium concentrations in the stream sediments relative to lake sediments is characteristic of many of the other trace metals analyzed in the NGR program (e.g., Cd) (P.W.B. Friske 1994, Gelogical Survey of Canada, Ottawa, pers. com.). Part of the reason for this trend may be the closer proximity of streams to source rocks. However, there are two points that should be emphasized concerning the NGR program data. Firstly, the minimum chromium concentrations reported in Table 3 were calculated using a default value which is half the detection limit (i.e., 20 $mg \cdot kg^{-1}$) for chromium. Thus the mean concentrations reported in Table 3 may be inaccurate. Secondly, despite the similarity among the mean background chromium concentrations, the standard deviation is very high, with the coefficient of variation being consistently greater than 65% and, in one case (for Newfoundland lake sediments), almost 380% (i.e., the standard deviation is almost four times the mean concentration). This wide variation in background chromium levels indicates that there are significant inputs of chromium from natural

sources at localized sites in Canada. Such information should be considered when assessing sediments and implementing freshwater sediment quality guidelines in Canada.

In Ontario, data relating to background concentrations of chromium in freshwater sediments are available for several lakes (Johnson et al. 1986; Johnson and Nicholls 1988; Blais and Kalff 1993; Snetsinger 1993), including the Laurentian Great Lakes (Kemp et al. 1978; Nriagu et al. 1983; IJC 1988; Mudroch et al. 1988) and their connecting channels (Fallon and Horvath 1985; Hesselberg and Hamdy 1987; Rossmann 1988; Environment Canada 1988) and the Ottawa and Rideau Rivers (Oliver and Kinrade 1972; Oliver and Agemian 1974) (see Table 4). Background levels of 44 $mg \cdot kg^{-1}$ and 38 $mg \cdot kg^{-1}$ were measured in the Ottawa and Rideau Rivers, respectively, by Oliver and Agemian (1974) and 22 $mg \cdot kg^{-1}$ and 21 $mg \cdot kg^{-1}$, respectively, by Oliver and Kinrade (1972). The slight discrepancy between the two studies may be because the former authors analyzed whole sediment whereas Oliver and Agemian (1974) measured the 80 mesh size fraction.

In the sediments of the Laurentian Great Lakes, Mudroch et al. (1988) summarized information on background concentrations of chromium. However their data are difficult to compare because of different sampling localities and techniques, and the methods used for the sediment analyses. Background levels of 51.4 ± 15.1 $mg \cdot kg^{-1}$, 31.8 ± 11.1 $mg \cdot kg^{-1}$, and 34.3 ± 8.2 $mg \cdot kg^{-1}$ have been reported for Lake Superior, Lake Huron, and Georgian Bay, respectively (Kemp et al. 1978). Similarly, Nriagu et al. (1983) calculated an average chromium concentration of approximately 55 $mg \cdot kg^{-1}$ in the pre-colonial sediments of Toronto and Hamilton Harbours, Lake Ontario. These values are comparable to chromium concentrations at the *Ambrosia* horizon in Lakes Superior and Huron (IJC 1988) and in Big Marsh Creek, Lake Erie (Mudroch 1980).

In the sediments of some smaller Ontario lakes, background concentrations of chromium similar to those observed in the Laurentian Great Lakes have been measured, for example, in the pre-1800 sediments of Lake Simcoe (i.e., 45.8 ± 1.6 $mg \cdot kg^{-1}$; Johnson and Nicholls 1988) and in the Turkey Lakes (i.e., 45.7 ± 3.9 $mg \cdot kg^{-1}$; Johnson et al. 1986). In the Turkey Lakes, background levels were calculated as the average concentration below the first core slice that exceeded the 'running' mean background concentration of deeper slices by two standard deviations. A larger range of background levels (i.e., 23 to 92 $mg \cdot kg^{-1}$ for time periods ranging between 1800 and 1906) has been observed in the Sudbury area (Snetsinger 1993).

Less data relating to background concentrations for chromium other than those in the NGR database are readily available for other areas of the country. In Lake Athabasca, Alberta,

background chromium levels are approximately 30 $mg \cdot kg^{-1}$ (Bourbonniere et al. 1996), whereas in Nova Scotia, Mudroch and Sandilands (1978) found that chromium concentrations below 5 cm depth were between approximately 4 and 22 $mg \cdot kg^{-1}$ in six cores collected from four lakes in the province. Relatively high natural chromium levels of > 60 $mg \cdot kg^{-1}$ were reported by Allan (1975) who 23 summarized data from areas in the northwestern Canadian Precambrian Shield, North West Territories, remote from point-source inputs of chromium. Similarly, background concentrations of > 100 $mg \cdot kg^{-1}$ were reported for many cores collected from lakes located in the Eastern Townships of Québec (Blais and Kalff 1993), an area underlain by undifferentiated rock of the Palaeozoic era. The province of Québec has also completed an extensive, internally consistent, geochemical dataset for the province, with over one million sediment samples collected to date. However, these data are not yet available in published format (M. Beaumier, Ministere d'Energies, Mines et Resources Naturelles, Québec, 1994, pers. com.).

Ambient and Contaminated Concentrations

Sediments contaminated with chromium can be attributed to a number of human activities. The majority of the anthropogenic emissions of total chromium to the atmosphere in Canada result from fossil fuel combustion, with smaller amounts being released from various industrial processes (including iron and steel production, and refractory and chemical processing) and transportation-related activities, such as motor vehicle operation (Environment Canada 1993). Although a complete inventory of chromium discharges to land and water is unavailable, it has been suggested that the annual loadings exceed 5 000 $tonnes \cdot yr^{-1}$ and 27 $tonnes \cdot yr^{-1}$ to the two media, respectively (Environment Canada 1993). Of the more than 5 000 tonnes of chromium wastes discharged annually onto land in Canada, the majority comes from metal-finishing sludges and from slags, sludges, and solids wastes produced by non-ferrous base metal smelters and refineries (Environment Canada 1993). With respect to loadings into water, available data indicate that liquid effluents from Canadian base metal smelters and refineries, as well as iron and steel plants, petroleum refineries, and metal refinishing plants in Ontario release significant quantities of chromium each year (Jaimet 1990; MacLatchy 1992, and data from the Municipal/Industrial Strategy for Abatement program). Other potentially important sources of chromium to the freshwater environment in Canada include discharges from pulp and paper mills, leather tanning industries and thermal generating stations. Many small industries utilize chromium in their operations and as a result substantial amounts can be discharged from urban stormwater runoff and sewage treatment plants. In addition to human activities, elevated chromium concentrations

in sediments may be attributed to long-range transport and deposition of atmospheric chromium (natural or anthropogenic), as well as natural geological sources. In this chapter, observed concentrations are considered to be contaminated if there is a known point source input of chromium to the water or sediment; otherwise, it is classified as ambient.

Most of the data relating to surface (ambient and contaminated) concentrations of chromium in Canadian freshwater sediments have been collected in Ontario. In the Sudbury area, where elevated concentrations of many metals (e.g., nickel, copper, cadmium, and zinc) have been reported (Dillon et al. 1988), concentrations of chromium in the recent sediments (i.e., 1966 to 1985) of 19 lakes ranged between 31 and 98 $mg \cdot kg^{-1}$ (Snetsinger 1993). These levels are relatively similar to the background concentrations of 23 to 92 $mg \cdot kg^{-1}$ measured in the same lakes suggesting that the surface concentrations are generally uncontaminated by human activities. Similar ambient concentrations of 35.1 to 90.6 $mg \cdot kg^{-1}$ were reported for the epilimnetic sediments of 15 lakes located in the Muskoka-Haliburton region of Ontario (Reimer and Duthie 1993), which is an area that is distant from anthropogenic sources of chromium and is dominated by Precambrian Shield bedrock. In the Moira River drainage basin, which discharges into the Bay of Quinte, Lake Ontario, slightly lower ambient concentrations of 12 to 31 $mg \cdot kg^{-1}$ were measured (Mudroch and Capobianco 1979). The Moira River basin also is underlain by Precambrian metamorphic and plutonic (igneous) rocks although these are overlain in the southern part of the basin by Palaeozoic sedimentary and metamorphic rocks. Thus, the reason for the difference in the two areas (Moira River basin versus Muskoka-Haliburton area) may be due to geology, or it may be attributed in part to the weaker digestion method used by Mudroch and Capobianco (1979) compared to Reimer and Duthie (1993).

Merriman (1987) used a 0.5 N hydrochloric acid 'cold' digestion to determine the available chromium concentrations in the top 3 cm of sediments of the Ottawa River (i.e., less than 13 $mg \cdot kg^{-}$). Stronger digestions (i.e., nitric + hydrochloric acids at 70-90°C) were applied by Oliver and Kinrade (1972) and Oliver and Agemian (1974). The latter authors reported elevated concentrations of chromium (114 $mg \cdot kg^{-1}$) in the river sediments at the mouth of Brewery Creek, which receives drainage from 6 sanitary sewers in Hull. In the Rideau River, Oliver and Kinrade (1972) reported a concentration of 128 $mg \cdot kg^{-1}$ at a station adjacent to the sewage treatment plant for Smith Falls. Similarly elevated chromium concentrations (138 $mg \cdot kg^{-1}$) were found at Merrickville (about 12 km downstream of Smith Falls) where in the past, untreated waste from two metal plating industries has been discharged directly into the river (Oliver and Agemian 1974).

Other data for Ontario lake sediments have been reported for Lake Simcoe. In Kempenfelt Bay, point-source inputs of chromium from the town of Barrie have resulted in significant chromium contamination, with an average total concentration of 416.7 ± 155.9 $mg \cdot kg^{-1}$ being found in surface (0 - 1 cm) sediments (Johnson and Nicholls 1988). It is evident that the point-source inputs from Barrie into Kempenfelt Bay have had an impact on ambient chromium concentrations in the other basins of the lake because concentrations decrease as a function of distance from Kempenfelt Bay (i.e., 80.7 ± 6.6 $mg \cdot kg^{-1}$, 75.4 ± 3.9 $mg \cdot kg^{-1}$ and 43.3 ± 6.9 $mg \cdot kg^{-1}$ in Main Bay, Cook Bay, and the outlet of the lake, respectively) (Johnson and Nicholls 1988).

Surficial concentration data for the Laurentian Great Lakes (Thomas et al. 1975; Kemp et al. 1978; Nriagu et al.1983; Mudroch 1980, 1984, 1993; Hart et al. 1986; IJC 1988; OME 1989a; Verbrugge et al. 1991; Jaagumagi et al. 1991b; Poulton 1992; Krantzberg 1994), their tributaries (Fitchko and Hutchinson 1975) and their connecting channels (i.e., St. Clair River, Lake St. Clair, Detroit River, St. Mary's River; see Thornley and Hamdy 1984; Fallon and Horvath 1985; Hamdy and Post 1985; Mudroch and Duncan 1986; Hesselberg and Hamdy 1987; Environment Canada 1988; Edsall et al. 1991; Jaagumagi et al. 1991a; Mudroch 1991; Nichols et al. 1991; Pope and Kauss 1995) are abundant. Because of the plethora of data available, only the contaminated areas will be discussed here.

In Lake Ontario, as a result of discharges of effluents from the iron and steel works into Hamilton Bay, total chromium concentrations measured in surface sediments have exceeded 200 $mg \cdot kg^{-1}$ (OME 1989b; Krantzberg 1994), with concentrations in the 0 to 4 cm section of one core collected from Hamilton Harbour averaging 537 ± 37 $mg \cdot kg^{-1}$ (Nriagu et al. 1983). In Toronto Harbour, surface (0 to 5 cm) values of 150 $mg \cdot kg^{-1}$ and 210 $mg \cdot kg^{-1}$ (in August and September 1987, respectively) were recorded in Ashbridges Bay, which receives discharge from the main sewage treatment plant for the city of Toronto (Jaagumagi et al. 1991b). Similar concentrations of 150 ± 14 $mg \cdot kg^{-1}$ and 134.5 ± 5 $mg \cdot kg^{-1}$ were measured by Nriagu et al. (1983) at two sites in the Harbour. In Port Hope Harbour, a surface concentration of 123 $mg \cdot kg^{-1}$ was reported at a station in Turning Basin, located near the cooling water discharge for CAMECO (formerly Eldorado Resources Ltd), a company which undertakes the conversion of uranium trioxide to uranium hexafluoride and ceramic-grade uranium dioxide (Hart et al. 1986; OME 1990). A chromium concentration of 138 $mg \cdot kg^{-1}$ was measured at a depth of 20 cm in a core collected from another station in Turning Basin, indicating either naturally high background levels and/or previous chromium contamination in the basin. The latter would be a result of operations and waste management practices associated with the refinery, most likely between 1933 and 1953 (OME

1990). In the Bay of Quinte, surface chromium concentrations of 67.8 ± 15.5 $mg \cdot kg^{-1}$ were measured near the Belleville sewage treatment plant by Poulton (1992) who notes that these levels are higher than in the open waters of Lake Ontario, but lower than in highly industrialized areas such as Hamilton Harbour. In addition, they are higher than the ambient concentrations (12 to 31 $mg \cdot kg^{-1}$) measured by Mudroch and Capobianco (1979) for the Moira River, which discharges into the Bay of Quinte.

In the connecting channels of the Upper Great Lakes, elevated concentrations of chromium have been found in parts of the St. Mary's River (Hesselberg and Hamdy 1987; Environment Canada 1988; Edsall et al. 1991; Jaagumagi et al. 1991a; Nichols et al. 1991; Mudroch 1991; Pope and Kauss 1995). The St. Mary's River, which connects Lakes Superior and Huron, is one of the 42 areas designated by the IJC (1988) as an area of concern. Contamination of the river has resulted from power generation, shipping and contaminant discharge from both industrial and municipal point sources, and also from urban non-point sources. Chromium concentrations of 1 004 $mg \cdot kg^{-1}$, 193 $mg \cdot kg^{-1}$ and 194 $mg \cdot kg^{-1}$ were measured in the vicinity of Bellview Marine Park, Sault Ste. Marie by Mudroch (1991), who suggests that these high levels may be the result of past discharges from the Northwestern Leather Company tannery, which was located upstream at Ashum Bay on the U.S. side of the river. Levels as high as 140 $mg \cdot kg^{-1}$ were measured in Algoma Slip (Hesselberg and Hamdy 1987), which is adjacent to Algoma Steel cn the north side of the river. The slip serves as a docking area for large commercial vessels and also receives contaminant inputs from three Algoma steel discharges and two tributary creeks (Pope and Kauss 1995). Similarly high concentrations (i.e., up to 170 $mg \cdot kg^{-1}$) were reported by Jaagumagi et al. (1991a) in the area downstream of Algoma Steel and the St. Mary's Paper Mills. However, Jaagumagi et al. (1991a) noted that more than 50% of the chromium that they measured was in the residual phase, suggesting that a good portion of the chromium may originate from a natural source.

Welland River, chromium concentrations in surface sediments averaged 3 712 ± 130 $mg \cdot kg^{-1}$ and reached a maximum concentration of 5 120 $mg \cdot kg^{-1}$ downstream of Atlas Specialty Steels Co., a steel manufacturing plant. These levels are in contrast to concentrations of approximately 30 $mg \cdot kg^{-1}$ upstream in the upper Welland River and Lyons Creek (Dickman et al. 1990). In the Detroit River, elevated chromium concentrations have been recorded at certain locations along the river; for example, 120 $mg \cdot kg^{-1}$ at the mouth of the Little River, near the sewage treatment plant (Thornley and Hamdy 1984), 92.3 $mg \cdot kg^{-1}$ around Fighting Island (Fallon and Horvath 1985) and 360 $mg \cdot kg^{-1}$ in the lower part of the river (Nichols et al. 1991).

An abundance of data is available for surface sediments of the St. Lawrence River (Kuntz 1988; Wilkins 1988; Kauss et al. 1988; Anderson 1990) including its tributaries (Goulet and LaLiberté 1982) and Lac St. Louis (Jarry et al. 1985; Mudroch and Joshi 1991). There appears to be no evidence of a point source contamination of chromium (i.e., < 33 $mg \cdot kg^{-1}$) along the river near Maitland, which lies between Brockville and Prescott (Wilkins 1988). However, concentrations of over 100 $mg \cdot kg^{-1}$ have been observed upstream at both Kingston and Gananoque, most likely as a result of municipal and industrial sources (Kuntz 1988). Further downstream at Cornwall, elevated chromium levels were recorded at the outfall of Courtaulds/British Cellophanes Ltd (BCL). Courtaulds manufactures rayon fibre from wood pulp and provides viscose to its associated company, BCL where cellophane film is produced. Concentrations of 82 $mg \cdot kg^{-1}$ have been measured at that location by Anderson (1990), as well as 53.5 ± 1.91 $mg \cdot kg^{-1}$ by Richman (1991) and 55 $mg \cdot kg^{-1}$ by Kauss et al. (1988). Concentrations of chromium also are elevated above background in the sediments downstream of two other major sources of chromium in Cornwall. These sources are Domtar Fine Papers Ltd., which owns a large pulp and paper plant and specialty paper mill in Cornwall, and the Cornwall Water Pollution Control Plant (Kauss et al. 1988; Kuntz 1988). Further downstream, in Lac St. Louis, concentrations as high as 170 to 190 $mg \cdot kg^{-1}$ have been reported (Jarry et al. 1985; Mudroch and Joshi 1991), most likely because of the lake's proximity to the city of Montréal. Data pertaining to tributary inputs of chromium to the St. Lawrence River can be found in Goulet and LaLiberté (1982).

Although ambient and contaminated levels of chromium in freshwater sediments in other regions of the country are scarce (e.g., North West Territories; Moore 1980), much information is available for the province of Alberta. In the Battle River drainage basin, ambient chromium concentrations generally are less than 25 $mg \cdot kg^{-1}$ (Anderson et al. 1994). In Lake Athabasca, chromium levels of 24.4 to 35.5 $mg \cdot kg^{-1}$ were measured recently in the surface sediments at eight locations in the lake (Bourbonniere et al. 1996). Three cores taken from the lake exhibited a nearly constant concentration with depth suggesting that the surface concentrations are uncontaminated by human activities. Slightly higher levels (35 to 43 $mg \cdot kg^{-1}$) were found in the Athabasca River by Day and Reynoldson (1995), whereas slightly lower concentrations (14 to 23 $mg \cdot kg^{-1}$) were reported recently by Dobson et al. (in press) who studied the Peace, Smoky, and Athabasca Rivers. Identical methods for analysis were employed in both studies. In contrast to these results, Allan and Jackson (1978) reported elevated ambient chromium concentrations of 114.3 ± 12.5 $mg \cdot kg^{-1}$ (Lake Athabasca), 63.3 ± 26.9 $mg \cdot kg^{-1}$ (Athabasca River), and 87.3 ± 31.4 $mg \cdot kg^{-1}$ (Athabasca River delta) as part of the Alberta Oil Sands Environmental Reserch Program (AOSERP). An average chromium concentration of 77.8 ± 28.7 $mg \cdot kg^{-1}$ (range of 34 to 116 $mg \cdot kg$) was

observed for the entire AOSERP study area (Lutz and Hendzel 1977). Allan and Jackson (1978) state that the concentrations they measured were normal or even low when compared to contaminated sediments elsewhere. Thus, the discrepancy among the various studies may be due to differences in digestion methods (Lutz and Hendzel 1977). The AOSERP results include the residual (lithogenic) chromium in the sediments and therefore, probably reflect background levels of chromium. Likewise, in British Columbia, high ambient concentrations of chromium (averaging 206 to 1 045 $mg \cdot kg^{-1}$ for the 13 subregions sampled) were measured in the surface sediments of 369 uncontaminated lakes (Rieberger 1992). Because these lakes are remote from any known point source inputs of chromium, it is likely that, they represent background chromium concentrations for the region.

Marine Sediments

Background Concentrations

Whereas the NGR database provides a consistent database of background levels of trace elements, including chromium in Canadian freshwater sediments, a similar database does not exist for marine sediments. In fact, only limited information is available concerning background concentrations of chromium in marine (including estuarine) sediments in Canada. In this section, the few published data on background concentrations of Chromium in Canadian marine sediments are summarized. It should be noted that some manipulation of the data (e.g., calculation of means and standard deviations) was necessary in order to facilitate their presentation. Although an attempt has been made to summarize the marine Chromium concentration data consistently according to the definitions provided above, quantitative comparisons among the few datasets presented here should be interpreted with caution.

In the Beaufort Sea, background (below 8 cm) Chromium levels of 22.1 ± 36.1 $mg \cdot kg^{-1}$ (n = 546, range of 4.1 to 260) were reported by Thomas et al. (1982), whereas from the data of Mudroch (1987), we calculated an average Chromium concentration of 76.7 ± 14.2 $mg \cdot kg^{-1}$ for 13 samples collected from a depth of 15 to 20 cm. This latter value is very similar to the background (below 5 cm) concentrations of about 74 to 85 $mg \cdot kg^{-1}$ that we calculated for cores collected from the Hudson/James Bay region, using data provided by Buckley et al. (1993).

On the Pacific Coast in Alice Arm, British Columbia, background (below 10 cm) Chromium concentrations in 11 cores ranged between 20.1 and 79.4 mg·kg^{-1} (Goyette and Christie 1982). Similar average Chromium levels (23 to 51 mg·kg^{-1}) were found below 10 cm in cores collected from Quatsino Sound (Harding and Thomas 1987), Loughborough Inlet, and Belize/Seymour Inlets, British Columbia (Goyette 1990).

Ambient and Contaminated Concentrations

Compared to background chromium concentrations, relatively more studies are available that provide data on ambient and contaminated chromium concentrations in marine sediments. Regions of Atlantic Canada have been studied (Cranston et al. 1974; Willey 1976; Willey and Fitzgerald 1980; Loring 1976; 1979a,b; 1982; Pelletier and Canuel 1988; Samant et al. 1990; Buckley and Winters 1992; Coakley and Poulton 1993), as well as Arctic Canada (Campbell and Loring 1980; Thomas et al. 1982; Mudroch 1987; Arctic Laboratories and LGL 1987; Wainwright and Humphrey 1988; Loring 1982, 1984) including Hudson/James Bay (Buckley et al. 1993), and Pacific Canada (Goyette and Christie 1982; Harding and Thomas 1987; Francois 1988; Harding and Goyette 1989; Goyette and Boyd 1989; Goyette 1990; Dunn et al. 1992). For Atlantic Canada, ambient concentrations of chromium in whole sediment samples collected from the Saguenay Fjord were similar in three separate investigations and ranged between 33 and 70 mg·kg^{-1} (Loring 1976), 49 and 97 mg·kg^{-1} (Loring 1982), and 12 and 72 mg·kg^{-1} (Pelletier and Canuel 1988). Chromium concentrations were higher in the < 53 µm fraction than in the whole sediment sample (Loring 1982; Pelletier and Canuel 1988). Despite the fact that the Saguenay Fjord receives contamination from urban outfalls and industrial waste discharges, Loring (1976) suggests that the observed ambient chromium levels are near background concentrations because 91 to 98% of the chromium is in the detrital, or acid insoluble, fraction.

Slightly lower chromium concentrations (i.e., 7 to 63 mg·kg^{-1}; mean of 25.5 ± 10.8 mg·kg^{-1}, n= 59) were measured in the upper St. Lawrence Estuary between Québec City and the Saguenay Fjord (Coakley and Poulton 1993), although Loring (1982) reported that chromium concentrations were similar to those found in the Saguenay Fjord (i.e., 39 to 95 mg·kg^{-1}). In the lower part of the St. Lawrence Estuary, downstream of the Saguenay Fjord, Loring (1982) reported sediment levels that were slightly elevated (i.e., 37 to 120 mg·kg^{-1}), possibly as a result of the influence of various point and non-point source discharges along the river and the Fjord. Similarly, ambient chromium levels were elevated (i.e., 8 to 158 mg·kg^{-1}) in parts of the Gulf of St. Lawrence, particularly along the

west coast of Newfoundland and eastern Nova Scotia. However, with respect to source rock concentrations, these levels approximate the naturally occurring chromium concentrations for the area. Other data concerning surficial chromium concentrations in Atlantic Canada sediments have been reported for Placentia Bay, Newfoundland, (mean of 45.5 ± 10.4 $mg \cdot kg^{-1}$) (Willey 1976), the Miramichi Estuary (mean of 44.3 ± 23.7 $mg \cdot kg^{-1}$ for sediments < 63 µm) (Willey and Fitzgerald 1980), for Dalhousie and Belledune Harbours, New Brunswick (mean of 39 ± 4 $mg \cdot kg^{-1}$ and 57 ± 1 $mg \cdot kg^{-1}$, respectively (Samant et al. 1990), and Loring (1979b) for the Bay of Fundy (range of 15 to 352 $mg \cdot kg^{-1}$).

Ambient chromium concentrations in sediments along the coast of British Columbia often are similar to background concentrations. For example, mean chromium concentrations in surficial sediments (0 to 2 cm) from northern British Columbia ranged from 20.1 to 61.3 $mg \cdot kg^{-1}$ in ten cores (note that the value of 421 $mg \cdot kg^{-1}$ reported for one of the cores appears to be a typographical error in the Appendix) and 9.5 to 43 $mg \cdot kg^{-1}$ in five grabs, collected from Alice Arm (Goyette and Christie 1982). Similarly, chromium concentrations in surficial sediments averaged 28.8 ± 14.5 $mg \cdot kg^{-1}$ for five grab samples collected from Hecate Strait (Harding and Goyette 1989). Further down the coast, ambient concentrations averaged 35.1 ± 0.42 $mg \cdot kg^{-1}$ in two grabs collected from Laredo Sound and 26.6 ± 4.1 $mg \cdot kg^{-1}$ in three grabs collected from Surf Inlet (Harding and Thomas 1987, Harding and Goyette 1989).

Harding and Thomas (1987), Harding and Goyette (1989), Goyette and Boyd (1989), Goyette (1990) and Dunn et al. (1992) reported chromium concentrations of 9 to 59.6 $mg \cdot kg^{-1}$ in surficial sediment samples located along the southern coast of British Columbia. Average concentrations calculated from data provided by Goyette and Boyd (1989) and Goyette (1990) were similar to those calculated by the authors (Harding and Thomas 1987, Harding and Goyette 1989) and ranged between 28.4 ± 4.88 $mg \cdot kg^{-1}$ and 50.6 ± 8.21 $mg \cdot kg^{1}$ in four of the studies. However, concentrations of over 110 $mg \cdot kg^{-1}$ were measured in Saanich Inlet by Francois (1988), with levels of up to 267 $mg \cdot kg^{-1}$ (<149 µm fraction only) being measured in Port Moody Arm, in Vancouver Harbour. This location is near the Ioco oil refinery, which appears to be the major source of chromium to the harbour (Goyette and Boyd 1989). Otherwise, chromium levels measured in Vancouver Harbour sediments were generally < 50 $mg \cdot kg^{-1}$ (Goyette and Boyd 1989).

In the Canadian Arctic, where there is little or no industrial development, mineral deposits and geologic formations are expected to be the primary sources of trace metals, including chromium, to the sediments. Not surprisingly, concentrations of chromium in the surficial sediments

approximate background concentrations. In the eastern Arctic, chromium levels in the surficial sediment averages 57 to 84 $mg \cdot kg^{-1}$ along the Arctic Coast, the Sound, and in Baffin Bay (Campbell and Loring 1980; Loring 1982, 1984), whereas Hudson/James Bay, ambient chromium levels averaged 71 to 86 $mg \cdot kg^{-1}$ (Buckley et al. 1993). In most parts of the Beaufort Sea, similar mean concentrations of 64 to 84 $mg \cdot kg^{-1}$ have been reported (Thomas et al. 1982; Pelletier and Buckley 1984; Mudroch 1987; Wainwright and Humphrey 1988). However, anthropogenic sources of trace metals to the Mackenzie River and the Canadian Beaufort Sea, largely as a result of oil-exploration activities, can result in wastes associated with the disposal and accidental spillage of drilling fluids (Thomas et al. 1982; Wainwright and Humphrey 1988). Thus, chromium concentrations in sediments averaging 126.0 ± 10.5 $mg \cdot kg^{-1}$ have been observed (Arctic Laboratories and LGL Ltd. 1987). Notwithstanding, even these levels may be indicative of ambient conditions as opposed to contaminated conditions, given that background concentrations as high as 260 $mg \cdot kg^{-1}$ have been measured in deeper sediments collected from the Beaufort Sea (Thomas et al. 1982).

Conclusions

Both anthropogenic and natural sources of chromium may contribute to high sediment concentrations in marine and freshwater systems. Anthropogenic inputs of chromium that may result in elevated levels in sediments have the potential to cause adverse effects on the resident biota. On the other hand, higher sediment concentrations that are a result of natural sources of chromium are probably less of a concern as the indigenous biota may be adapted to these levels. It is evident from the data presented in this chapter that estimates of background, ambient, and contaminated chromium concentrations in Canadian freshwater and marine sediments vary substantially within specific regions as well as across the country. Reported mean background concentrations range between 4 and 160 $mg \cdot kg^{-1}$for freshwater sediments (Tables 3 and 4) and between 20 and 85 $mg \cdot kg^{-1}$ in marine sediments, although levels as high as 6 121 $mg \cdot kg^{-1}$ have been reported in Newfoundland lakes (Table 3). Given this wide variation in background sediment concentrations, it is not surprising that mean ambient and contaminated levels may overlap background concentrations in some areas of the country. In freshwater lakes and streams, average ambient and contaminated concentrations range between 12 and 3 712 $mg \cdot kg^{-1}$ (Table 3). The upper value of that range was measured in the Welland River, Ontario, downstream of the Atlas Specialty Steels Co.. In contrast, chromium concentrations in excess of 200 $mg \cdot kg^{-1}$ have been measured in the surface sediments of Hamilton Bay and Toronto Harbour, Lake Ontario

(OME 1989b; Nriagu et al. 1983; Krantzberg 1994). In marine sediments, mean concentrations reported in surficial sediments show less variation than for freshwater sediments, ranging between 10 and 352 $mg \cdot kg^{-1}$. The latter value was measured in the Bay of Fundy by Loring (1979b), although elevated levels have been found also in Halifax Inlet (Buckley and Winters 1992), in Vancouver Harbour (Goyette and Boyd 1989), and in the Beaufort Sea (Thomas et al. 1982; Arctic Laboratories and LGL Ltd. 1987).

Because of the variation in background, ambient, and contaminated concentrations across the country, generalizations regarding the relative contribution of natural versus anthropogenic chromium to observed concentrations cannot be made. Therefore, the mean concentration ranges of chromium in sediments (as stated above) that are presented in this document should be viewed as qualitative estimates for a given region and not as "blanket" zones where background concentrations are expected to be uniform. This review provides an indication of the degree to which sediment chromium concentrations in certain areas may be influenced by natural processes and human activities, and can assist in focussing further sediment assessment and remediation efforts by environmental managers employing sediment quality guidelines. Ultimately, such information, in conjunction with an understanding of the fate, behaviour and toxicity of sediment-associated chromium, can be used to evaluate the potential impact of sediment-associated chromium on aquatic ecosystems.

CHAPTER 3. CHROMIUM IN AQUATIC SYSTEMS

Speciation in the Water Column

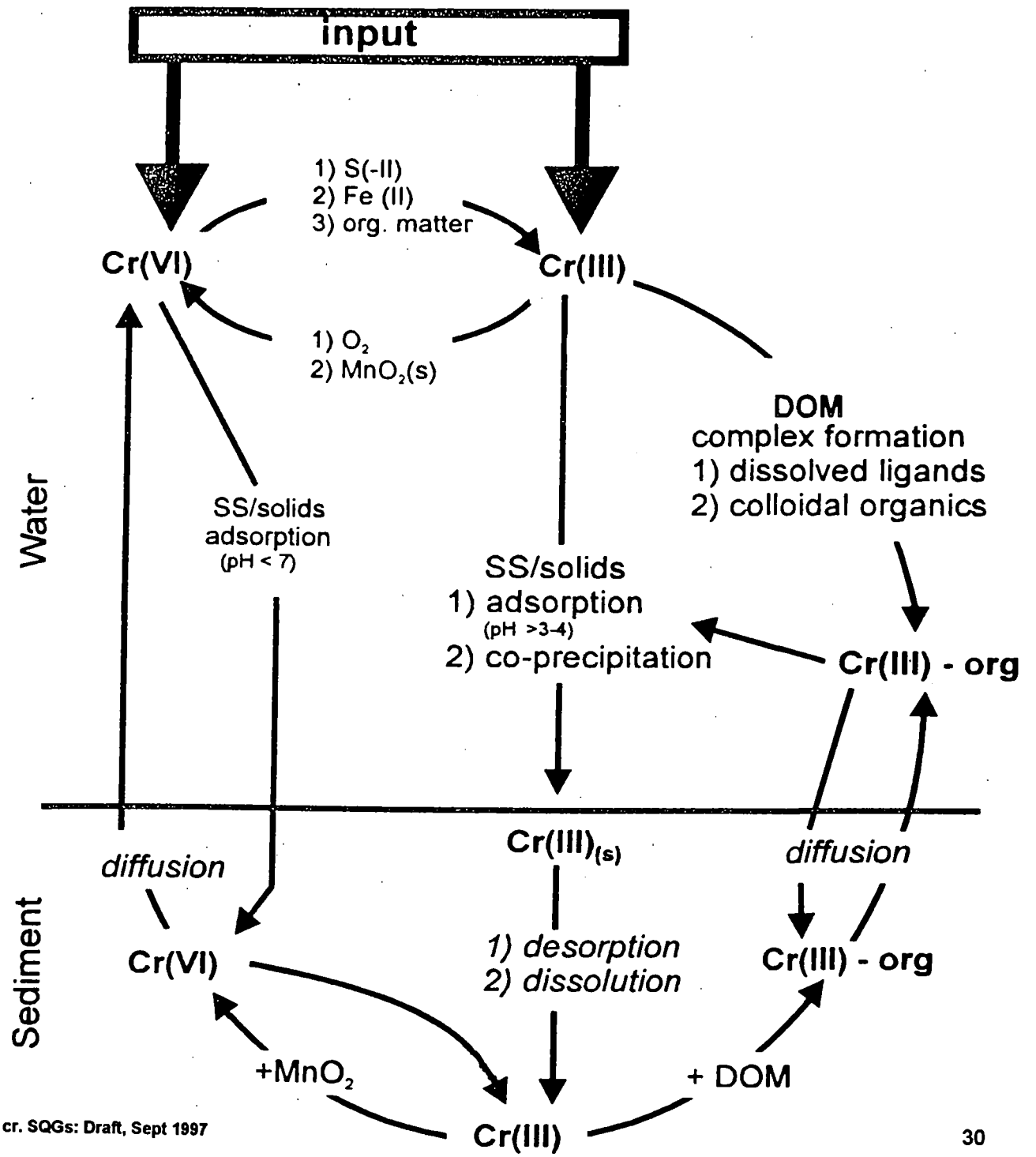
Although chromium can occur in any of the oxidation states ranging from Cr(-II) to Cr(VI), only Cr(0), Cr(III), and Cr(VI) are common, with Cr(III) being the most stable (Mertz 1969). In the water column, chromium occurs as Cr(III) or Cr(VI) (Murray et al. 1983; Richard and Bourg 1991). Cr(VI) is a strong oxidizing agent and it has been shown to penetrate biological membranes readily (Mertz 1969; Fujii et al. 1990). These two characteristics have led Cr(VI) to be considered more toxic than Cr(III) (Oshida et al. 1981; Van Weerelt et al. 1984; Peternac and Legovic 1986; Aboul Dahab et al. 1990). Environmental conditions influencing the speciation and behaviour of chromium in the water column is discussed below, with a focus on Cr(III) and Cr(VI).

Chromium(VI)

Depending on redox conditions, Cr(VI) can form a number of stable oxyacids and anions including chromate (CrO_4^{2-}), hydrochromate ($HCrO_4^-$), and dichromate ($Cr_2O_7^{2-}$), all three of which are highly soluble in water (Saleh et al. 1989). In the pH range of natural waters (i.e., 5 to 9), the concentration of dichromate is generally negligible with chromate prevailing at a pH > 6.5 (Moore 1991). Although Cr(VI) is highly mobile in the aquatic environment, it is relatively unreactive. Its removal from the water column to the sediments is controlled primarily by two mechanisms: 1) adsorption and desorption reactions (Eary and Rai 1987), and 2) reduction to Cr(III) (Schroeder and Lee 1975), which is rapidly scavenged from the water column. An overview of the chromium cycle in the aquatic environment is depicted in Figure 1.

Chromate [i.e., Cr(VI)] ions can be adsorbed by manganese, aluminum and iron oxides (Jackson et al. 1980; Stollenwerk and Grove 1985), clay minerals and natural soils (Mayer and Schick 1981), and colloids. Adsorption of Cr(VI) appears to be a surface complexation reaction between aqueous chromates and hydroxyl-specific surface sites (Richard and Bourg 1991). While the adsorption reaction may be important in acidic media, it decreases rapidly above pH 7 (Richard and Bourg 1991; Johnson and Sigg 1992) and in the presence of competing dissolved anions (Stollenwerk and Grove 1985; Richard and Bourg 1991) such as phosphate (Mayer and Schick 1981). Adsorption of Cr(VI) is considered to be of minor importance in the water column of natural systems (Moore 1991).

Figure 1. Chromium cycling in the aquatic environment (adapted from Richard and Bourg 1991 and Johnson and Sigg 1992). (s) = solid, SS = suspended solids, DOM = dissolved organic matter



Numerous reductants such as reduced sulphur species (Saleh et al. 1989; Johnson et al. 1992; Balistrieri et al. 1992), ferrous iron (Schroeder and Lee 1975; Saleh et al. 1989; Leshchinskaya and Linnik 1990), and organic compounds (Schroeder and Lee 1975; Nakayama et al. 1981b; Harzdorf 1987; Saleh et al. 1989) such as fulvic acid (Leshchinskaya and Linnik 1990; Wittbrodt and Palmer 1995), can reduce Cr(VI) to Cr(III) in both freshwater and marine environments (Figure 1). Although the reduction reactions are favoured under conditions of low pH and Eh (such as in anaerobic fresh- and interstitial waters), in waters with abundant phytoplankton or dissolved organic carbon, reduction of Cr(VI) to Cr(III) also may be rapid (Santschi 1988; Beaubien et al. 1994) because the degradation products act as potential reducing agents (Leshchinskaya and Linnik 1990). In freshwater field studies, the reduction of Cr(VI) to Cr(III) was found to be significant in lacustrine waters with strong reducing conditions (Balistrieri et al. 1992; Johnson et al. 1992; Beaubien et al. 1994). Similarly, in laboratory experiments, reduction of Cr(VI) in the presence of reducing agents was reported to be very rapid. For example, using batch experiments, Saleh et al. (1989) found that the reduction of Cr(VI) to Cr(III) in the water, by reduced sulfur and iron ions, was instantaneous under anaerobic conditions. In interstitial waters, Harzdorf (1987) found that 87% or more of the initial added Cr(VI) was reduced within one hour in sediments from the River Elbe, Büttel Channel, and Lake "Kremperheide".

On the other hand, Harzdorf (1987) also found that in the aerobic water column of the River Rhine, Büttel Channel, and Lake "Kremperheide" (pH 8.0, 8.1, 7.6, respectively), Cr(VI) reduction rates were some orders of magnitude lower than in the sediments. Thus, in alkaline (oxic) fresh waters, the reduction of Cr(VI) may be quite slow. Similarly, the reduction of Cr(VI) may be limited in marine waters. Chipman (1966) observed that over 99% of Cr(VI) in standing seawater remained in solution after 8 days, as did Schulz-Baldes et al. (1983) who reported that 96.6% of the Cr(VI) added to their intertidal enclosures remained in the dissolved phase in the water column after 3 weeks.

Chromium(III)

The principal forms of aqueous Cr(III) species include Cr^{3+}, $CrOH^{2+}$, $Cr(OH)_3^0$, and $Cr(OH)_4^-$, with $CrOH^{2+}$ and $Cr(OH)_3^0$ being the main forms in all natural waters (Richard and Bourg 1991). Unlike Cr(VI), Cr(III) is not expected to migrate in natural systems because it is readily hydrolyzed and binds strongly to particles and organic matter in the water column. In the sediments, Cr(III) is rapidly, strongly and specifically adsorbed by iron and manganese oxides, clay minerals, and sand

(Schroeder and Lee 1975; Richard and Bourg 1991). The adsorption of Cr(III) increases with increasing pH (Young et al. 1992) and increasing organic content of the soil/sediment (Payà Pérez et al. 1988), but decreases when competing cations are present (Richard and Bourg 1991). Studies in which the partition coefficient (K_d) of Cr(III) to soils/sediments has been measured in both marine and freshwater sediments (*e.g.*, Young et al. 1987, 1992; Payà Pérez et al. 1988; Golimowski et al. 1990), have yielded values that differ by several orders of magnitude (i.e., from 10 to 30 000 $L \cdot kg^{-1}$). The orders of magnitude variation in K_d values most likely is a function of many factors including pH, sediment or soil type, initial chromium and other ionic concentrations, and experimental design (see Förstner 1987). Young et al. (1992) suggest that *in vitro* batch experiments may inadequately replicate conditions that are relevant to the adsorption and desorption of metals (including chromium) during resuspension events.

As shown in Figure 1, in freshwater and marine sediments, suspended sediments and interstitial waters, diagenetic processes such as desorption of Cr(III) from the bed sediment and suspended sediment, and oxidation to Cr(VI) can lead to remobilization of chromium, either as chromate or as Cr(III)-dissolved organic matter complexes. Francis and Dodge (1990) found that an anaerobic nitrogen-fixing *Clostridium* sp. solubilized chromium that had co-precipitated with geothite (iron hydroxide) by enzymatic reduction of the ferric iron and subsequent release of the chromium associated with the iron. Generally, however, it is assumed that once Cr(III) is scavenged from the water column, it becomes part of the sediment matrix and is thus unavailable (or at least less available) for uptake by biota (Pfeiffer et al. 1980). This reasoning has lead to the development of techniques for eliminating and/or reducing Cr(VI) toxicity in wastewaters by utilizing Cr-reducing bacteria such as *Pseudomonas* spp. (Vincze et al. 1994; DeLeo and Ehrlich 1994; Gopalan and Veeramani 1994), *Enterobacter* spp. (Fujii et al. 1990; Ohtake et al. 1990), or a mixture of bacteria (Kvasnikov 1988; Fude et al. 1994) to reduce the toxic, highly mobile Cr(VI) to the less toxic, immobile Cr(III).

The remobilization of chromium by the oxidation of Cr(III) to Cr(VI) is a critical process that is dependent on redox and other conditions (e.g., nutrient and dissolved manganese concentrations) in the system (Johnson and Sigg 1992). In fresh waters, some evidence suggests that diagenetic remobilization of chromium from the sediments to the overlying water may not be an important process. For example, in Lake Greifen, Switzerland, Johnson and Sigg (1992) and Johnson et al. (1992) found no evidence for Cr(III) oxidation, once the reduced or adsorbed chromium was removed from the water column. Similarly, Beaubien et al. (1994) detected no increases in either Cr(VI) or Cr(III) concentrations near the sediment water interface in Lake Ontario, nor did

Golimowski et al. (1985) observe evidence for Cr(III) remobilization in Lake Zürich. On the other hand, Balistrieri et al. (1992) found that concentrations of total, dissolved, and particulate chromium all increased in the bottom waters of Lake Sammamish, Washington, during oxic phases in the lake, although there was an overall loss of chromium from the water column during suboxic and anoxic phases. Likewise, in a microcosm experiment conducted using simulated Lake Powell (Utah/Arizona) sediments and water, Medine and Porcella (1980) suggest that a shift in sediment Cr(III) to Cr(VI) during initial mixing of the sediment, may have caused the observed release of chromium from the sediments to the water in the first 20 days of the experiment. In support of their theory, sediment profiles demonstrated an upward migration of chromium and its loss from the deeper sediment.

The conflicting results from these studies are a consequence of the various approaches (field versus laboratory) and the various conditions used in each of the lakes/experiments. Unfortunately, information regarding the ultimate fate of Cr(III) in lacustrine systems will require long-term mass balance studies on a variety of lakes and must include the measurement of both Cr(III) and Cr(VI) species. Even then, the results may not be applicable to other lakes where redox and trophic conditions and metal (manganese and iron) concentrations may be different.

In the marine environment, the release of chromium from bottom sediments through the catalytic oxidation of Cr(III) by manganese oxides has been documented in laboratory experiments using dissolved Cr(III) and particulate manganese oxides (e.g., Schroeder and Lee 1975; Nakayama et al. 1981a; Van der Weijden and Reith 1982). In the field, increases in chromium concentration at the sediment-seawater interface (Jeandel and Minster 1984), as well as increases in porewater concentrations of chromium with depth in the core (Douglas et al. 1986), have been cited as evidence for Cr(III) remobilization from some mineral phases such as chromic hydroxide or chromite. Cr(III) oxidation has been found to decrease in the presence of competing metal sorbates, such as aluminum, at pH > 4 (Fendorf et al. 1993). In addition, Simon et al. (1994) found that concentrations of chromium in the interstitial water of sediment collected from the Calcasieu River and estuary, Louisiana, exceeded water column concentrations when the total concentration of ammonia in the sediment exceeded 1 $\mu mol \cdot g^{-1}$ ww. Although this presented a potential for the diffusive flux of chromium to the overlying water, Ciceri et al. (1992) calculated a negative benthic flux (i.e., flux from the water to the sediments) of dissolved chromium using Fick's law, as well as from benthic chambers located in the Tyrrhenian Sea. Similarly, Zwolsman et al. (1993) found little evidence of post-depositional mobility of chromium in certain cores collected from salt marsh sediments in the Scheldt estuary, the Netherlands. In the marine environment it is likely that

chromium cycling occurs in response to biogeochemistry (Richard and Bourg 1991). When chromium is scavenged by suspended sediment and is deposited in marine sediments, diagenetic processes can lead to remobilization of chromium either as CrO_4^{2-} or as organic Cr(III) complexes (Figure 1).

Chromium(VI) / Chromium(III) Distribution in the Water Column

Thermodynamic calculations predict that in oxidized marine, estuarine, and fresh waters, Cr(VI) should predominate, whereas in reduced (e.g., anoxic sediment) or highly acidic environments Cr(III) should prevail. However, in oxic seawaters, the results from several studies have demonstrated that dissolved Cr(III) can be present at levels greater than would be predicted by thermodynamic considerations (Elderfield 1970; Nakayama et al. 1981a; Van der Weijden and Reith 1982; Jeandel and Minster 1984; Kaczynski and Kieber 1993). Several explanations for the apparent discrepancy have been suggested. One hypothesis relates partially to the kinetic control of the redox processes. Whereas, as discussed above, Cr(VI) can be reduced to Cr(III) in a matter of minutes to hours by several reducing agents, the oxidation reaction of Cr(III) to Cr(VI) is relatively slow (Schroeder and Lee 1975; Balistrieri et al. 1992) or non-existent (Van der Weijden and Reith 1982; Eary and Rai 1987; Johnson et al. 1992), with the importance of oxygen as an oxidant being questionable. For example, Eary and Rai (1987) demonstrated that the oxidation of Cr(III) did not occur by surface-catalyzed reactions with dissolved oxygen, but rather by direct reaction with manganese oxides.

A second explanation for the apparent redox disequilibrium between Cr(III) and Cr(VI) may be a result of the formation of colloidal or dissolved organic matter-chromium complexes (Nakayama et al. 1981b; Johnson 1990; Beaubien et al. 1993) which are unreactive to oxygenation, and precipitation from the water column. A third explanation has been provided by Kaczynski and Kieber (1993) who observed from field studies that Cr(III) was photochemically produced (under oxic conditions) in two freshwater lakes in North Carolina. Furthermore, Cr(III) was present in measurable concentrations in all seven water bodies (representing freshwater, marine and estuarine systems) surveyed in the United States. Although the mechanism of Cr(III) photogeneration remains unclear, as does the photochemical precursor to Cr(III) in natural waters, Kaczynski and Kieber (1993) state that the photogeneration process presents another pathway for the existence of supposedly thermodynamically unstable Cr(III) in oxygenated surface waters.

In conclusion, further studies concerning Cr(III) and Cr(VI) speciation and cycling in both the aquatic and sediment environments are essential for a more complete understanding of the fate and behaviour of chromium in aquatic systems. These studies should focus on both the rates and extent of oxidation/reduction reactions and on the complexation/dissociation of chromium and natural organic material. Volatilization, photolysis, and biotransformation reactions do not appear to be important processes in the environmental fate of chromium (Moore 1991).

Speciation in the Sediment

Physical Aspects

Chromium appears to be associated with fine-grained sediment materials such as silts and clays (Oliver 1973; Loring 1979a; Rossmann 1988; Duzzin et al. 1988; Sager et al. 1990; Anderson 1990; Coakley and Poulton 1993; Weis et al. 1993; Mantei and Sappington 1994; Literathy et al. 1994). For this reason, the analysis of sediments for many metals, including chromium, often is restricted to a standard grain size (e.g., < 63 µm; Luoma 1990; Luoma et al. 1990) or is normalized, post analysis, to a standard grain size (Long et al. 1990) or to a conservative element such as aluminum (Johnson and Nicholls 1988; Coakley and Poulton 1993; Gagnon et al. 1993) or iron (Rule 1985; Gibbs 1994). On the other hand, chromium that is primarily of geologic origin may be more enriched in coarser sediments (Rowan and Kalff 1993). Chromium concentration also is positively correlated to sediment surface area (Oliver 1973; Mayer and Fink 1980). However, its relationship with organic content or percent loss on ignition is variable, with significantly positive trends (Loring 1979a; Mayer and Fink 1980; Duzzin et al. 1988; Fuller et al. 1990) and no trends (Fitchko 1974; Kauss and Post 1987; Mouvet et al. 1987; Aboul Dahab et al. 1990; Mudroch 1991; Gagnon et al. 1993) being observed. A significant negative correlation between chromium concentration in the sediments of Thompson Lake, North West Territories and organic matter was observed by Moore (1980), although he states that this trend may have occurred because metal-contaminated waste discharges from a gold mine into Thompson Lake consisted of materials having low organic content (e.g., crushed rock). Consequently, the deposition of the mine tailings increased the metal loading to the lake while simultaneously lowering the organic content.

Chemical Aspects

As a general rule, chromium derived from anthropogenic sources is found more in the extractable (and thus, more bioavailable) sediment phases (Biney and Beeko 1991; Bryan and Langston 1992), whereas chromium derived from natural sources is associated with the residual fraction (Angelidis and Grimanis 1989; Ruiz et al. 1991). Thus, in uncontaminated marine and freshwater sediments, the lattice bound (residual) chromium usually dominates. For example, in the relatively uncontaminated rivers of the Madeira River watershed, Brazil, greater than 80% of the chromium was found in the residual phase(average chromium concentrations ranged from 33 to 52 $mg \cdot kg^{-1}$) (Lacerda et al. 1990). However, as sediments become more heavily contaminated, organic, sulphide, and iron and manganese oxide phases become increasingly important both in surficial (Lacerda et al. 1990; Elsokkary and Müller 1990; Deely et al. 1992; Giani et al. 1994) as well as in subsurficial (Arafat and Nriagu 1986; Steinberg and Högel 1990; Giani et al. 1994) sediments.

In Canadian sediments, Ajayi and VanLoon (1989) reported that the residual fraction (F5 in the Tessier et al. scheme) comprised 74.6%, 43.3% and 63.7% of the total chromium concentration of 75 $mg \cdot kg^{-1}$, 86 $mg \cdot kg^{1}$, and 115 $mg \cdot kg$, respectively, in sediments collected from three locations in the Bay of Quinte, near Belleville, Ontario. In Ramsey Lake, Ontario where total chromium concentrations in the top two cm ranged between 82 and 132 $mg \cdot kg^{-1}$ at three stations, about 73 to 83% of the chromium was in the residual fraction, with about 10 to 20% being associated with iron and manganese oxides (Arafat and Nriagu 1986). In New Brunswick Harbour sediments, where the total chromium concentration in the sediment was 46 $mg \cdot kg^{-1}$), Samant et al. (1990) found a much higher association of chromium with iron and manganese oxides. They reported that 59% and 43% of the total chromium in Dalhousie Harbour and Belledune Harbour sediments, respectively, was associated with iron and manganese oxides versus 37.3% and 54.8%, respectively, in the residual fraction.

Metals such as chromium, can be retained by iron and manganese oxides by sorption and co-precipitation. Sorption is a process by which metals are bound to the surface of an existing solid by adsorption and surface precipitation, whereas co-precipitation is the simultaneous precipitation of a chemical element with other elements and includes mixed-solid formation, adsorption and inclusion (Francis and Dodge 1990). The reported percentages of chromium associated with iron and manganese oxides (i.e., the reducible fraction) are quite variable and appear to be dependent on the extent of anthropogenic chromium inputs (Angelidis and Grimanis 1989; Bryan and Langston 1992) and/or the nature of the fractionation procedure (Coetzee et al. 1995). In the northern

Adriatic Sea, the percentage of chromium associated with iron and manganese oxides (F3 in the Tessier et al. scheme) increased as the concentration of chromium in surficial sediment (0 to 5 cm) increased. About 4% of the chromium was in the reducible phase when the surficial sediment concentration was 92 $mg \cdot kg^{-1}$, whereas 12.9% was associated with iron and manganese oxides when the chromium concentration was 161 $mg \cdot kg^{-1}$ (Giani et al. 1994).

The fraction of chromium associated with the organic or oxidizable phases of the sediments also appears to be variable in both freshwater and marine sediments. Elsokkary and Müller (1989) found that the highest proportion of non-lithogenous chromium in Nile River sediments was in the organic fraction (i.e., 39.7% of total chromium), which is similar to the percentage of chromium determined to be in the oxidizable phase of the Clyde River, UK sediments (Davidson et al. 1994). Coetzee (1993) reported that 20% of the chromium in the sediments of Hartbeespoort Dam, South Africa was bound to organic matter, whereas Kapkov and Trishina (1990) found that all of the chromium in the White Sea sediments was in the oxidizable phase. Although the reported concentrations/fractions of chromium in the organic phase of sediments may be variable, there is a general consensus concerning chromium in the carbonate phase. Chromium is not usually associated with the carbonate (F2 in the Tessier et al. scheme) sediment fraction (Pestana et al. 1989; Ruiz et al. 1991; Modak et al. 1992; Cheung and Wong 1992; Young et al. 1992) because it is not known to form carbonate minerals even in sea water (Mayer 1988). Similarly, chromium is rarely measured to any extent in the exchangeable (F1 in the Tessier et al. scheme) sediment fraction in either freshwater (Jackson et al. 1980; Ruiz et al. 1991; Young et al. 1992) or marine (Pestana et al. 1989; Nair and Balchand 1993; Giani et al. 1994; McCloskey et al. 1995) sediments. This observation would tend to support the contention that Cr(III) sorbed and scavenged from the water column as insoluble chromium oxides or mixed hydrous oxides of Cr(III) and iron(III) (Jackson et al. 1980; Tuin and Tels 1988) is fundamentally unavailable to biota. The bioavailability of chromium to aquatic biota is discussed further in the proceeding chapter.

Bioavailability

Although there have been many studies that have attempted to define the 'bioavailable' fraction of metals in sediments, including chromium, there is still an incomplete understanding of the factors and processes that influence biological availability. According to Campbell and Tessier (1989), methods for assessing the relative bioavailability of sediment-bound metals and metals dissolved in the interstitial waters to sediment dwelling organisms can be divided into three categories: 1)

laboratory experiments, 2) field experiments, and 3) field surveys. Laboratory experiments include feeding experiments carried out with natural or artificial sediments, as well as bioassay experiments in which benthic or pelagic organisms are exposed to whole sediment, sediment elutriate, or interstitial water extracted from the sediment. Field experiments may involve the transfer of benthic organisms between contaminated and uncontaminated sites (Mersch and Johansson 1993; Camusso et al. 1994; Nelson et al. 1995) or the use of field plots or enclosures (Schulz-Baldes et al. 1983; Jenner and Bowmer 1990) that are subjected to different experimental manipulations such as radiotracer additions (Jackson et al. 1980). Field surveys, on the other hand, involve examining relationships between sediment characteristics (such as chromium concentration, loss on ignition, grain size) and chromium concentrations in the indigenous benthic organisms (Nugent et al. 1980; Dave 1992; Schlekat et al. 1994). In field experiments and field surveys, the use of indicator organisms, such as the blue (bay) mussel, *Mytilus edulis,* in marine sediments (Part 1987; Chassard-Bouchard 1991; Nelson et al. 1995) and bivalve *Elliptio complanata* in freshwater sediments (Campbell and Evans 1991; Metcalfe-Smith et al. 1992) has been exploited frequently.

A relatively new method for assessing the potential bioavailability of certain metals in sediments relates to the ratio of simultaneously extracted metals (SEM; as defined by a cold acid extraction) to acid-volatile sulphide (AVS) concentrations in the sediments (e.g., Hansen et al. 1996). Considering that AVS was found to influence metal porewater concentrations in anoxic test sediments (DiToro et al. 1990) and because the toxicity to benthos of certain divalent metals such as cadmium, lead, nickel, zinc, and copper has been correlated with porewater concentrations, DiToro et al. (1992) suggested that when the total molar concentration of these SEMs is less than the molar concentration of AVS (i.e., [SEM]/ [AVS] < 1), bioavailability and thus toxicity are low. More recently, it has been recommended that the difference between the molar concentrations of SEM and AVS (i.e., [SEM] - [AVS]) should be used instead of [SEM]/[AVS] ratios (MacDonald and Salazar 1995; Hansen et al. 1996) to provide insight into the extent of additional available binding capacity ([SEM] - [AVS] < 0) or the amount of bioavailable metals ([SEM] - [AVS] > 0).

In only one study has the [SEM]/ [AVS] ratio been used to assess chromium bioavailability (Ankley et al. 1994) and the validity of the results of that study, with respect to chromium, is questionable for two reasons. Firstly, the sediments were stored for three years prior to the sediment analyses and bioassay experiments. Young et al. (1992) found that there was a significant change (i.e., 11 to 87%) in the percent distribution of chromium in the various sediment fractions (extracted using the Tessier et al. scheme) collected from the Trenton Channel, Detroit River after 100 days storage in the dark at 4°C. Thus, Environment Canada (1994) suggests a sediment storage time of two

weeks or less with a maximum permissible storage time of 6 weeks. Swartz et al. (1985) recommend that sediment be stored at 4°C for no more than 5 days prior to the initiation of bioassays. The second reason the results of Ankley et al. (1994) should be viewed with caution is because chromium was not included in the SEM measurement. Even if chromium had been included in the SEM measurement, the existence of chromium sulphide species is most likely unimportant given that virtually all Cr(II) in water will convert into other oxidation states such as Cr(0) and Cr(III) (Pankow 1991). In addition, the AVS normalization procedure is invalid if the sediment AVS content is very low. This could occur, for example, in fully oxidized sediments (Adams et al. 1992). Thus, further evaluation of the [SEM]/[AVS] approach is required, although at this time there is no reason to expect that [SEM]/[AVS] or [SEM] - [AVS] should relate to chromium bioavailability or toxicity.

The application of phase-specific or sequential extractions of metals generally has yielded the best practical means for deriving predictive relationships between metal burdens in benthos and sediment concentrations. Thus, some people have attempted to predict bioavailability through the use of these various extraction procedures (Kurita and Pfeiffer 1991; Metcalfe-Smith et al. 1992). Intuitively, one would expect metal bioavailability, and thus uptake (body burden), to be higher for those fractions more readily removed from the sediment (e.g., exchangeable and easily reducible phases). Laboratory and empirical field evidence would suggest this to be true for some metals (see reviews by Luoma 1983 and Campbell and Tessier 1989). Unfortunately, in the case of chromium, studies comparing the uptake by freshwater or marine biota with total concentrations of chromium or concentrations of chromium in specific chemical phases in sediments are scarce.

Metcalfe-Smith et al. (1992) reported significant positive relationships between 0.5 N HCl-extractable chromium and the chromium concentration in two species of freshwater mussels (i.e., *Elliptio complanata* and *Lampsilis radiata*) collected from the Ottawa River, Canada where HCl-extractable chromium ranged between about 5 and 14 $mg \cdot kg^{-1}$. However, Biney and Beeko (1991) were unable to relate 0.5 N HCl-extractable chromium (range of < detection to 4.3 $mg \cdot kg^{-1}$) to concentrations of chromium in two freshwater fish species from the Wiwi River, Ghana. In addition, Sanchez et al. (1994) found that chromium levels in moss, trout (*Salmo trutta*), and eel (*Anguilla anguilla*) at the two sites having the highest HCl-extractable chromium concentrations (i.e., 23.0 $mg \cdot kg^{-1}$ and 20.0 $mg \cdot kg^{-1}$) were no higher than at the other sites in the Oiartzun River Valley, Spain.

In the marine environment, Rule (1985) found no correlation between the metal extraction method and chromium uptake by the grass shrimp, *Palaemonetes pugio,* and the clam, *Mercenaria mercenaria,* collected from four sites located in Hampton Roads Harbour and the Elizabeth River system along the eastern coast of USA. Chromium concentrations in the three sediment extracts: 1) DTPA (diethylenetriamine pentaacetic acid), 2) HNO_3, and 3) HNO_3-H_20_2, ranged between 0.4 to 0.9 $mg \cdot kg^{-1}$, 0.8 to 5.7 $mg \cdot kg^{-1}$, and 3.0 to 49.4 $mg \cdot kg^{-1}$, respectively, depending on the site. Similarly, Bryan and Langston (1992) attempted to relate HCl-extractable chromium and total chromium concentrations in United Kingdcm estuary sediments to body levels of chromium in two deposit-feeding species. Although treatment of the recently collected sediment with HCl released 90% of the total chromium from the most contaminated sediment (compared to only 5% from the pristine sediment), total chromium (i.e., chromium in the HNO_3 extract) was the best predictor of bioavailability with respect to tissue concentrations for both the clam, *Scrobicularia plana* ($r = 0.91$), and the polychaete, *Nereis diversicolor* ($r = 0.91$). Alternatively, Ayling (1974) observed that accumulation of chromium by oysters (*Crassostrea gigas*) was independent of total sediment chromium levels. Similarly, in a marine bioassay experiment, Simmers et al. (1984) found that accumulation of chromium in the sandworm, *Nereis virens,* was similar after exposure to sediment chromium concentrations of 0.9 $mg \cdot kg^{-1}$, 92.3 $mg \cdot kg^{-1}$, and 230.5 $mg \cdot kg^{-1}$.

In conclusion, with the exception of the study by Metcalfe-Smith et al. (1992) conducted in the Ottawa River, it would appear that bioavailability of chromium is essentially independent of the readily extractable chromium phases in both freshwater and marine sediments.

The relative importance of the sediments versus the water column or interstitial water as a source of chromium to benthic organisms is a subject of ongoing debate (Decho and Luoma 1991) that is dependent on many factors including the speciation of the metal and the ecology of the organism. For chromium, the situation is complicated by the presence of two chemical species (Guilizzoni 1991), Cr(III) and Cr(VI), that co-occur in the aquatic environment. Superimposed on the problem of chromium speciation is the fact that the ecology of the organism is critical in assessing the uptake of chromium and the relative importance of exposure route (i.e., sediment versus water). Some biota, such as rooted aquatic plants, sediment bacteria and epipelic algae obtain their nutrients and metals directly from uptake either from the water column or from interstitial water, whereas other biota are deposit- or filter-feeding organisms. In deposit-feeding organisms, chromium uptake can take place following the ingestion of sediment particles (Bryan and Uysal 1978; Bryan and Langston 1992), including sediment bacteria (Aislabie and Loutit 1986; Decho and Luoma 1991), or in some cases, by pinocytosis of particles at the body surface (Martoja

et al. 1988 as cited in Environment Canada 1993). In deposit-and filter-feeding organisms, desorption of chromium from the sediment or suspended sediment followed by uptake can take place during direct contact between particles and surface tissues. Uptake also can occur directly from the water column or interstitial water, either by adsorption to the body wall or exoskeleton, adsorption through the integument or respiratory surfaces, and ingestion of water (Reynoldson 1987).

All other factors being equal, uptake of chromium (or indeed any other metal) from solution is expected to be faster than from sediments (i.e., dissolved metals are inherently more available; Campbell and Tessier 1989). However, relatively higher concentrations of chromium are generally measured in sediments. Although total concentrations of chromium in dissolved and particulate phases in surface waters (e.g., Laurentian Great Lakes, Beaubien et al. 1994; Lake Sammamish, Washington, Balistrieri et al. 1992; Griefensee, Switzerland, Johnson et al. 1992) and marine waters (e.g., Pacific Ocean, Nakayama et al. 1981b; Murray et al. 1983; Japan Sea, Nakayama et al. 1981b) are typically below 1 $\mu g \cdot L^{-1}$, ambient and contaminated concentrations in Canadian sediments exceed 3 000 $mg \cdot kg^{-1}$ and 300 $mg \cdot kg^{-1}$ in freshwater and marine sediments, respectively.

Interstitial waters concentrations of chromium are generally higher than open water concentrations. Salomons (1989) found that the chromium concentration in pore water extracted from the 0 to 2 cm sediment layer in Ijsselmeer was 6.3 $\mu g \cdot L^{-1}$ compared to chromium levels in the overlying water of 0.57 $\mu g \cdot L^{-1}$. Similarly, Aislabie and Loutit (1986) reported that chromium levels in the interstitial waters of Sawyers Bay, New Zealand were 10 $\mu g \cdot L^{-1}$ compared to only 1 $\mu g \cdot L^{-1}$ in the overlying water. However, concentrations of chromium in fresh interstitial waters are generally lower than bulk sediment concentrations. Ankley et al. (1994) found that porewater chromium concentrations from the lower Fox River, Wisconsin, were < 4 $\mu g \cdot L^{-1}$ compared to whole sediment concentrations of 37.5 to 49.2 $mg \cdot kg^{-1}$. Likewise, Santiago et al. (1989) reported elutriate concentrations from Lake Geneva watershed sediments of < 2 $\mu g \cdot L^{-1}$ compared to whole sediment concentrations of 38 to 84 $mg \cdot kg^{-1}$. At one location, Santiago et al. (1989) measured sediment chromium concentrations of 228 $mg \cdot kg^{-1}$, with only 6.1 $\mu g \cdot L^{-1}$ extracted in the elutriate. Similarly, Lee and Mariani (1977) reported that chromium concentrations in bioassay elutriate waters from Duwamish River sediments, Seattle, Washington were < 2% of the sediment concentrations. The low concentrations of chromium in elutriate and pore waters, relative to concentrations in the bulk sediments would support the results obtained from sediment fractionation studies where it has been found that chromium is rarely measured to any extent in the exchangeable (i.e., F1 in the Tessier

et al. scheme) sediment fraction (Jackson et al. 1980; Pestana et al. 1989; Ruiz et al. 1991; Nair and Balchand 1993; Giani et al. 1994; McCloskey et al. 1995).

Alternately, in one contaminated marine system, Neff et al. (1980) found that the chromium levels in drilling mud filtrate (i.e., liquid phase of the whole mud obtained with a filter press) obtained from the northwestern Gulf of Mexico, measured 27 $mg \cdot L^{-1}$ compared to 500 $mg \cdot kg^{-1}$. Lignosulphonates, which are by-products of the pulp and paper industry, are used in offshore drilling muds as dispersants and viscosity agents. One of the most frequently used drilling muds is a water-based chrome lignosulphonate, containing variable amounts of bentonite clay, barium sulphate, and other minor additives (Neff et al. 1980).

Thus, the net uptake of chromium by benthos is expected to be a result of uptake from the easily available but low concentrations of chromium in open waters and the less readily available but potentially higher concentrations of chromium in the sediments. Intermediate between these two extremes is the interstitial water where chromium availability is expected to be high, but concentrations may be high or low depending on the location.

In conclusion, the bioavailability of chromium will be affected not only by the chemical equilibrium between dissolved chromium and chromium adsorbed by particles, but also by the speciation of chromium with respect to Cr(III) and Cr(VI) in the water, as well as the ecology of the organism itself. The chemical (Cr(III) versus Cr(VI)) and physical (dissolved versus particulate) speciation of chromium is expected to be a function of the factors which affect chromium behaviour in the water (e.g., pH, nutrient concentration, redox potential, salinity, water hardness, light, microbial activity; Bergbäck et al. 1989, Guilizzoni 1991). It is likely that Cr(VI) may be taken up directly from the interstitial and overlying water. Cr(III), however, is rapidly scavenged by particles such as oxides of iron or by surfaces (see Figure 1), and thus is more likely to be taken up by the ingestion of particles. Although the bioavailability of Cr(III) that is adsorbed to, or co-precipitated with, is dependent on the ecology of the organism, it is also a function of the extent of remobilization and desorption of previously scavenged Cr(III) as well as the oxidation of Cr(III) to Cr(VI) (Bergbäck et al. 1989). Thus, for organisms that live in intimate contact with the sediments but are unable to ingest particulate matter, and for organisms that have various routes of uptake, chromium bioavailability is still dependent on its physical and chemical speciation.

Bioaccumulation

Bioaccumulation is the net uptake into an organism of a metal or other contaminant from all exposure routes. It is important to understand the bioaccumulation of any metal in an organism in order to link concentrations in the sediments to toxic effects on the organism and to determine food chain transfer of the metal. Unfortunately, bioaccumulation of chromium from sediments to biota is difficult to assess because of the various methods used for determining total chromium in the two media (sediments and biota) and also because of the lack of standardization in the reporting of results (e.g., wet weight versus dry weight values, sieved versus unsieved sediment analyses, depurated versus non-depurated animals etc.). Thus, comparisons among bioaccumulation factors (BAFs) reported in different studies is difficult. Notwithstanding, a summary of some of the BAFs (i.e., concentrations of chromium in biota·g^{-1} wet weight versus concentrations in sediment·g^{-1} dry weight) reported in the literature is given in Table 5. These data have not been normalized for organic carbon content of the sediment or for lipid content of the organisms.

It is evident from Table 5, as well as from other studies (Phelps et al. 1974; Namminga and Wilhm 1977; Kostic and Draskovic 1982; Kulikova et al. 1985; Leland and Scudder 1990; Manny et al. 1991; Metcalfe-Smith et al. 1992), that BAFs (weight wet of biota/dry weight of sediment) in both freshwater and marine systems are generally less than one. Therefore, concentrations of chromium in biota living within, on, or feeding on the sediments generally appear to be less than or similar to chromium concentrations in the sediments, even in contaminated areas. For example, Schreier et al. (1987) measured chromium levels in several fish species collected from the Sumas River Basin (a tributary of the Fraser River, British Columbia). This basin was contaminated in 1975 with chromium and several other metals as a result of a landslide containing chrysotile asbestos. Although total chromium concentrations in the sediments averaged 196 mg·kg^{-1} in 1983-84 (range of 170 to 217 mg·kg^{-1}), chromium levels in the fish averaged only 0.17 to 0.30 mg·kg^{-1} ww, depending on the species (see Table 5). Similarly, in a marine bioassay experiment, Simmers at al. (1984) reported that accumulation of chromium by the sandworm, *Nereis virens,* was similar after 7 and 14 days exposure to sediment chromium concentrations of 0.9 mg·kg^{-1}, 92.3 mg·kg^{-1}, and 230.5 mg·kg^{-1}. Likewise, Comber et al. (1989) reported that the chromium content of *Nereis diversicolor* after depuration was not related to the metal's concentration in the surrounding sediment. In addition, Leland and Scudder (1990) reported that, although chromium concentrations in the < 62 µm fraction of sediments from the San Joaquin River, California, and selected tributaries ranged between 72 and 180 mg·kg^{-1}, chromium levels

Table 5 Reported bioaccumulation factors (BAF) (concentration of chromium in organism in $mg \cdot kg^{-1}$ ww / concentration of chromium in sediments in $mg \cdot kg^{-1}$ dw) measured in freshwater and marine environments.

[Organism] $mg \cdot kg^{-1}$ ww	[Sediment] $mg \cdot kg^{-1}$ dw	BAF ww/dw	Organism	Location	Comments	Reference
FRESHWATER INVERTEBRATES						
~0.68	118.4	~0.006	*Lumbriculus variegatus*	lower Fox R., Wisconsin	30-day bioassay	Ankley et al. (1994)
~0.85	70.8	~0.012	*Lumbriculus variegatus*	lower Fox R., Wisconsin	30-day bioassay	Ankley et al. (1994)
~1.04	71.4	~0.015	*Lumbriculus variegatus*	lower Fox R., Wisconsin	30-day bioassay	Ankley et al. (1994)
1.03±0.08[1]	50.4±8.0	0.02	40 taxa of aquatic insects	from N. Carolina rivers		Smock (1983a)
0.70±0.01[1]	134.3±0.01	0.005	*Potamonautes warreni*	S. Africa wetland		van Eeden & Schoonbee (1991)
0.003±0.001[1]	185.7±0.04	nd	*Potamonautes warreni*	S. Africa wetland		van Eeden & Schoonbee (1991)
0.42± 0.001[1]	151.9±0.06	nd	*Potamonautes warreni*	S. Africa wetland		van Eeden & Schoonbee (1991)
0.98± 0.001[1]	106.5±0.04	nd	*Potamonautes warreni*	S. Africa wetland		van Eeden & Schoonbee (1991)
0.42± 0.001[1]	124.7±0.04	nd	*Potamonautes warreni*	S. Africa wetland		van Eeden & Schoonbee (1991)
3.22±0.003[1]	110.8±0.05	nd	*Potamonautes warreni*	S. Africa wetland		van Eeden & Schoonbee (1991)
7.7*	17*	0.453	*Fusconaia flava*	Illinois R.		Mathis & Cummings (1973)
4.4*	17*	0.259	*Amblema plicata*	Illinois R.		Mathis & Cummings (1973)
4.7*	17*	0.276	*Quadrula quadrula*	Illinois R.		Mathis & Cummings (1973)
10.0*	17*	0.588	Tubificid worms	Illinois R.		Mathis & Cummings (1973)
0.18±0.13[1]	1.6±1.2	0.12	*Crassostrea virginica*	Lake Pontchartrain, Louisiana		Byrne & DeLeon (1986)
0.29±0.13[1]	8.8±3.8	0.03	*Rangia cuneata*	Lake Pontchartrain, Louisiana		Byrne & DeLeon (1986)
0.53±0.13[1]	8.6±1.8	0.06	*Rangia cuneata*	Lake Pontchartrain, Louisiana		Byrne & DeLeon (1986)
5.91±3.99[1]	1098±495.2	0.005	*Chironomous decorous*	New Jersey impoundment		Kraus (1989)
FRESHWATER FISH						
0.28±0.32[2]	5.8±7.5	0.05	*Heterobranchus sp.*	Wiwi R., Ghana		Biney and Beeko (1991)
0.205±0.559	21.8±8.43[4]	0.009	*Lepomis microlophus*	Florida stormwater		Campbell (1995)
0.437±0.215	42.7±58.2[4]	0.010	*Lepomis microlophus*	reference site		Campbell (1995)
0.13*	17*	0.008	*Esox lucius*	Illinois R.		Mathis & Cummings (1973)
0.11*	17*	0.006	*Micropterus salmoides*	Illinois R.		Mathis & Cummings (1973)
0.06*	17*	0.004	*Morone chrysops*	Illinois R.		Mathis & Cummings (1973)
0.14*	17*	0.008	*Lepisosteus platostomus*	Illinois R.		Mathis & Cummings (1973)
0.16*	17*	0.009	*Micropterus dolomieui*	Illinois R.		Mathis & Cummings (1973)

Table 5 Continued. Reported bioaccumulation factors (BAF) (concentration of chromium in organism in $mg \cdot kg^{-1}$ ww / concentration of chromium in sediments in $mg \cdot kg^{-1}$ dw) measured in freshwater and marine environments.

[Organism] $mg \cdot kg^{-1}$ ww	[Sediment] $mg \cdot kg^{-1}$ dw	BAF ww/dw	Organism	Location	Comments	Reference
0.16*	17*	0.009	*Micropterus dolomieui*	Illinois R.		Mathis & Cummings (1973)
0.13*	17*	0.008	*Ictiobus cyprinellus*	Illinois R.		Mathis & Cummings (1973)
0.45*	17*	0.026	*Dorosoma cepedianum*	Illinois R.		Mathis & Cummings (1973)
0.09*	17*	0.005	*Moxostoma macrolepidotum*	Illinois R.		Mathis & Cummings (1973)
0.21*	17*	0.012	*Carpiodes cyprinus*	Illinois R.		Mathis & Cummings (1973)
0.16*	17*	0.009	*Cyprinus carpio*	Illinois R.		Mathis & Cummings (1973)
0.17	196	<0.001	Redside shiner	Sumas River, BC		Schreier et al. (1987)
0.16	196	<0.001	Northern squawfish	Sumas River, BC		Schreier et al. (1987)
0.22	196	0.001	Coho salmon	Sumas River, BC		Schreier et al. (1987)
0.23	196	0.001	Prickly sculpin	Sumas River, BC		Schreier et al. (1987)
0.27	196	0.001	3-spine stickleback	Sumas River, BC		Schreier et al. (1987)
0.30	196	0.002	Large-scale sucker	Sumas River, BC		Schreier et al. (1987)
FRESHWATER MACROPHYTES $mg \cdot kg$ dw						
-	11.3	1.1	filamentous algae	Georgian Bay		Jackson (1988)
-	24.5	0.4	filamentous algae	Georgian Bay		Jackson (1988)
-	25.0	0.7	filamentous algae	Georgian Bay		Jackson (1988)
-	13.7	0.6	filamentous algae	Niagara R.		Jackson (1988)
-	27.7	0.4	filamentous algae	Niagara R.		Jackson (1988)
1.2	13	0.09	*Myriophyllum verticulatum*	Wolf Lake		Mudroch & Capobianco (1979)
2.7	13	0.21	*Nymphaea odoratae*	Wolf Lake		Mudroch & Capobianco (1979)
7.2	19	0.379	*Myriophyllum verticulatum*	Hawkins Bay (Moira Lake)		Mudroch & Capobianco (1979)
2.5	19	0.132	*Nymphaea odoratae*	Hawkins Bay (Moira Lake)		Mudroch & Capobianco (1979)
6.9	31	0.223	*Myriophyllum verticulatum*	Bend Bay (Moira Lake)		Mudroch & Capobianco (1979)
3.2	31	0.103	*Elodea canadensis*	Bend Bay (Moira Lake)		Mudroch & Capobianco (1979)
3.5	31	0.113	*Nymphaea odoratae*	Bend Bay (Moira Lake)		Mudroch & Capobianco (1979)
13.0	69.5	0.187	*Eriocaulon septangulare*	Clearwater L., Ontario		Reimer & Duthie (1993)
3.9	35.8	0.109	*Eriocaulon septangulare*	Crosson L., Ontario		Reimer & Duthie (1993)

Table 5 Continued. Reported bioaccumulation factors (BAF) (concentration of chromium in organism in $mg \cdot kg^{-1}$ ww / concentration of chromium in sediments in $mg \cdot kg^{-1}$ dw) measured in freshwater and marine environments.

[Organism] $mg \cdot kg^{-1}$ ww	[Sediment] $mg \cdot kg^{-1}$ dw	BAF ww/dw	Organism	Location	Comments	Reference
8.5	36.6	0.232	*Eriocaulon septangulare*	Dickie Lake, Ontario		Reimer & Duthie (1993)
10.6	45.1	0.235	*Eriocaulon septangulare*	Fawn Lake, Ontario		Reimer & Duthie (1993)
3.6	35.1	0.103	*Eriocaulon septangulare*	Gullfeather L., Ontario		Reimer & Duthie (1993)
23.9	90.6	0.264	*Eriocaulon septangulare*	Hannah L., Ontario		Reimer & Duthie (1993)
6.8	57.4	0.118	*Eriocaulon septangulare*	Harp Lake, Ontario		Reimer & Duthie (1993)
6.6	69.2	0.095	*Eriocaulon septangulare*	Heney L., Ontario		Reimer & Duthie (1993)
5.5	46.7	0.118	*Eriocaulon septangulare*	Leech L., Ontario		Reimer & Duthie (1993)
8.5	45.5	0.187	*Eriocaulon septangulare*	Leonard L., Ontario		Reimer & Duthie (1993)
12.5	89.9	0.139	*Eriocaulon septangulare*	Lohi Lake, Ontario		Reimer & Duthie (1993)
9.3	58.5	0.159	*Eriocaulon septangulare*	McKay L., Ontario		Reimer & Duthie (1993)
9.0	38.6	0.233	*Eriocaulon septangulare*	Moot Lake, Ontario		Reimer & Duthie (1993)
4.2	44.5	0.094	*Eriocaulon septangulare*	Plastic L., Ontario		Reimer & Duthie (1993)
5.7	42.2	0.135	*Eriocaulon septangulare*	Ril Lake, Ontario		Reimer & Duthie (1993)
10.3	37.0	0.278	mostly *Vallisneria americana*	Detroit R.		Manny et al. (1991)
FRESHWATER MOSSES						
5	15	0.067	*Fontinalis antipyretica*	Ave River, Portugal	Reference Site	GonHalves et al. (1992)
16	71	0.225	*Fontinalis antipyretica*	Ave River, Portugal	Station A1	GonHalves et al. (1992)
23	132	0.174	*Fontinalis antipyretica*	Ave River, Portugal	Station A2	GonHalves et al. (1992)
42	135	0.311	*Fontinalis antipyretica*	Ave River, Portugal	Station A3	GonHalves et al. (1992)
30	179	0.168	*Fontinalis antipyretica*	Ave River, Portugal	Station A5	GonHalves et al. (1992)
46	201	0.229	*Fontinalis antipyretica*	Ave River, Portugal	Station A6	GonHalves et al. (1992)
2.1	27	0.078	*Fontinalis antipyretica*	Ave River, Portugal	Station A9	GonHalves et al. (1992)
1.4	16	0.088	*Fontinalis antipyretica*	Ave River, Portugal	Station A10	GonHalves et al. (1992)
1.4	14	0.100	*Fontinalis antipyretica*	Ave River, tributary	Station A11	GonHalves et al. (1992)
38	66	0.576	*Fontinalis antipyretica*	Ave River, tributary	Station E1	GonHalves et al. (1992)
65	246	0.264	*Fontinalis antipyretica*	Ave River, tributary	Station E2	GonHalves et al. (1992)
21	33	0.636	*Fontinalis antipyretica*	Ave River, tributary	Station E4	GonHalves et al. (1992)
4.9	16	0.306	*Fontinalis antipyretica*	Ave River, tributary	Station E5	GonHalves et al. (1992)

Table 5 Continued. Reported bioaccumulation factors (BAF) (concentration of chromium in organism in $mg \cdot kg^{-1}$ ww./ concentration of chromium in sediments in $mg \cdot kg^{-1}$ dw) measured in freshwater and marine environments.

[Organism] $mg \cdot kg^{-1}$ ww	[Sediment] $mg \cdot kg^{-1}$ dw	BAF ww/dw	Organism	Location	Comments	Reference
7.5	39	0.192	*Fontinalis antipyretica*	Ave River, tributary	Station Ph1	GonHalves et al. (1992)
14	81	0.173	*Fontinalis antipyretica*	Ave River, tributary	Station P11	GonHalves et al. (1992)
6.5	33	0.197	*Fontinalis antipyretica*	Ave River, tributary	Station V1	GonHalves et al. (1992)
5.0	14	0.357	*Fontinalis antipyretica*	Ave River, tributary	Station V2	GonHalves et al. (1992)
534	1187	0.500	*Fontinalis antipyretica*	Ave River, tributary	Station S1	GonHalves et al. (1992)
7.5	16	0.469	*Fontinalis antipyretica*	Ave River, tributary	Station S2	GonHalves et al. (1992)
MARINE INVERTEBRATES						
0.104[1]	37	0.003	*Mytilus Galloprovincialis*	Black Sea	meat tissue	Andreev & Simeonov (1990)
0.14[1]	8.5	0.016	*Mytilus Galloprovincialis*	Black Sea	meat tissue	Andreev & Simeonov (1990)
0.057[1]	7.8	0.007	*Mytilus Galloprovincialis*	Black Sea	meat tissue	Andreev & Simeonov (1990)
0.134[1]	7	0.019	*Mytilus Galloprovincialis*	Black Sea	meat tissue	Andreev & Simeonov (1990)
11.83±2.77	9.87±0.85	1.2	*Mytilus edulis*	Cork Harbour, Ireland		Berrow (1991)
5.50±1.30	11.60±0.70	0.5	*Mytilus edulis*	Cork Harbour, Ireland		Berrow (1991)
11.50±1.44	9.60±1.78	1.2	*Mytilus edulis*	Cork Harbour, Ireland		Berrow (1991)
8.50±1.15	15.57±1.49	0.6	*Mytilus edulis*	Cork Harbour, Ireland		Berrow (1991)
7.00±0.58	15.10±0.58	0.5	*Mytilus edulis*	Cork Harbour, Ireland		Berrow (1991)
6.85±2.85	17.0 ±0.58	0.4	*Mytilus edulis*	Cork Harbour, Ireland		Berrow (1991)
13.33±1.36	11.30±2.69	1.2	*Mytilus edulis*	Cork Harbour, Ireland		Berrow (1991)
20.50±0.58	12.50±0.99	1.6	*Mytilus edulis*	Cork Harbour, Ireland		Berrow (1991)
19.83±6.83	17.17±1.23	1.15	*Mytilus edulis*	Cork Harbour, Ireland		Berrow (1991)
12.83±1.36	11.30±0.66	1.1	*Mytilus edulis*	Cork Harbour, Ireland		Berrow (1991)
14.67±1.20	11.0 ±0.06	1.3	*Mytilus edulis*	Cork Harbour, Ireland		Berrow (1991)
0.32[1]	1.88[3]	0.17	*Mytilus edulis*	Lymfjord, Denmark	soft tissue	Brix and Lyngby (1985)
2.24[1]	307	0.007	*Mytilus edulis*	New Zealand	soft tissue	Brooks & Rumsby (1965)
1.4[1]	307	0.005	*Ostrea sinuata*	New Zealand	soft tissue	Brooks & Rumsby (1965)
0.42[1]	307	0.001	*Pecten novae-zelandiae*	New Zealand	soft tissue	Brooks & Rumsby (1965)
0.296[1]	64	0.005	*Scrobicularia plana*	Tamar Estuary, England	whole soft body	Bryan & Uysal (1978)
6.79±6.04[1]	3.0±0.6	2.26	*Palaemonetes pugio*	Eastern coast of USA	whole animal, bioassay	Rule (1985)

Table 5 Continued. Reported bioaccumulation factors (BAF) (concentration of chromium in organism in $mg \cdot kg^{-1}$ ww / concentration of chromium in sediments in $mg \cdot kg^{-1}$ dw) measured in freshwater and marine environments.

[Organism] $mg \cdot kg^{-1}$ ww	[Sediment] $mg \cdot kg^{-1}$ dw	BAF ww/dw	Organism	Location	Comments	Reference
6.79±6.04[1]	3.0±0.6	2.26	*Palaemonetes pugio*	Eastern coast of USA	whole animal, bioassay	Rule (1985)
5.10±6.43[1]	43.4±3.1	0.12	*Palaemonetes pugio*	Eastern coast of USA	whole animal, bioassay	Rule (1985)
3.89±4.54[1]	29.6±0.9	0.13	*Palaemonetes pugio*	Eastern coast of USA	whole animal, bioassay	Rule (1985)
0.78±1.30[1]	49.4±4.4	0.02	*Palaemonetes pugio*	Eastern coast of USA	whole animal, bioassay	Rule (1985)
2.39±3.04[1]	9.6±0.9	0.25	*Palaemonetes pugio*	Eastern coast of USA	whole animal, bioassay	Rule (1985)
0.64± 0.42[1]	3.0±0.6	0.215	*Mercenaria mercenaria*	Eastern coast of USA	soft tissues, bioassay	Rule (1985)
0.36 ±0.42[1]	43.4±3.1	0.008	*Mercenaria mercenaria*	Eastern coast of USA	soft tissues, bioassay	Rule (1985)
0.22±0.34[1]	29.6±0.9	0.007	*Mercenaria mercenaria*	Eastern coast of USA	soft tissues, bioassay	Rule (1985)
0.17±0.27[1]	49.4±4.4	0.003	*Mercenaria mercenaria*	Eastern coast of USA	soft tissues, bioassay	Rule (1985)
0.95±1.62[1]	9.6±0.9	0.099	*Mercenaria mercenaria*	Eastern coast of USA	soft tissues, bioassay	Rule (1985)
MARINE FISH						
0.106[2]	24	0.004	*Gobius niger*	Black Sea	fillet	Andreev & Simeonov (1990)
0.116[2]	24	0.005	*Gobius niger*	Black Sea	liver	Andreev & Simeonov (1990)
0.066[2]	7	0.009	*Gobius niger*	Black Sea	fillet	Andreev & Simeonov (1990)
0.064[2]	7	0.009	*Gobius niger*	Black Sea	liver	Andreev & Simeonov (1990)
0.06[2]	8.5	0.007	*Gobius niger*	Black Sea	fillet	Andreev & Simeonov (1990)
0.042[2]	8.5	0.005	*Gobius niger*	Black Sea	liver	Andreev & Simeonov (1990)
0.086[2]	7.8	0.011	*Gobius niger*	Black Sea	fillet	Andreev & Simeonov (1990)

[1] dw to ww conversion assuming 86% moisture (i.e., mg/kg dw / 7.14 = mg/kg ww) (Byrne and DeLeon 1986 assumed moisture content of 90% for *Crassostrea virginica* and 84% for *Rangia cuneata*, Lytle and Lytle 1990 assumed moisture content of 84% for *Crassostrea virginica*, Metcalfe-Smith et al. 1992 assumed moisture content of 90% for *Elliptio complanata* and *Lampsilis radiata*, Lindstrom et al. 1988 measure average moisture content of 86%in *Mytilus edulis*, Bradford and Luoma 1980 used moisture content of 83% for *Mya arenaria*, Hendriks 1995 used dry to wet weight fractions of 0.1 (i.e., moisture content of 90%) for plants and invertebrates)
[2] dw to ww conversion assuming 80% moisture (i.e., mg/kg dw / 5 = mg/kg ww) (Schreier et al. 1987, Winger et al. 1990)
[3] LOI to dw conversion assuming 1% organic content (i.e., mg/kg LOI/ 100 = mg/kg dw)
[4] ww to dw conversion assuming 90% moisture (by volume) in the sediments (i.e., mg/kg ww x 10 = mg/kg dw)
* assuming that sediment concentrations were reported as dw and biota as ww

in the soft tissues of the freshwater bivalve *Corbicula fluminea* were only 0.5 to 0.8 $mg \cdot kg^{-1}$ (i.e., near or below the detection limit of their analysis).

Jorgensen (1990) modelled the distribution of chromium in a contaminated Danish firth and found that:

$$[chromium]_{Mytilus}\ \mu g/g\ dw = 0.015\ [chromium]_{sed}\ \mu g/g\ dw$$

whereas Metcalfe-Smith et al. (1992) reported that:

$$[chromium]_{Elliptio}\ \mu g/g\ dw = 3.63 + 0.17\ [Cr_{ext}]_{sed}\ \mu g/g\ dw$$
$$[chromium]_{Lampsilis}\ \mu g/g\ dw = 1.36 + 0.17\ [Cr_{ext}]_{sed}\ \mu g/g\ dw$$

where $[chromium_{ext}]_{sed}$ represents the HCl extractable chromium in Ottawa River sediments. The order of magnitude difference in the slopes of the lines (i.e., bioaccumulation) between the two studies may be due to the fact that different bivalve species were examined and/or because Jorgensen (1990) analyzed for total chromium, whereas Metcalfe-Smith et al. (1992) examined bioavailable metal. Kurita and Pfeiffer (1991) found that HCl extractable chromium in the sediment of Sepetiba Bay, Brazil, was only about 1% of the total chromium concentration (i.e., 0.7 $mg \cdot kg^{-1}$, whereas total chromium averaged 61.3 $mg \cdot kg^{-1}$). Thus, all other factors being equal, the slope of equations relating chromium levels in biota to 'extractable' chromium concentrations would be expected to be considerably higher than those utilizing total chromium concentrations.

As discussed in the previous section, studies that have attempted to distinguish among the various modes of uptake of chromium by benthic organisms, either from sediment or from interstitial/open water, are scarce. In one laboratory experiment, Capuzzo and Sasner (1977) found that the whole-body levels of chromium in bivalves exposed to Cr(III) in solution [1 mg $Cr(III) \cdot L^{-1}$ sea water] were 10 to 30 (*Mytilus edulis*) and 35 to 100 (*Mya arenaria*) times higher than in those exposed to artificial sediment suspensions. Lytle and Lytle (1990) also report that the enrichment of chromium in oysters, *Crassostrea virginica,* collected from two of their study sites in Mississippi Sound was a result of enhanced levels in the water column and not ingestion or inclusion of sediment particles. On the other hand, Schulz-Baldes et al. (1983) reported that *Mytilus edulis* placed directly on top of the sediment surface of their intertidal mesocosm had significantly higher sediment accumulation factors for Cr(VI) than did those suspended in the water column. They observed that uptake was related more to taxonomic position of the benthic invertebrate (i.e., calcareous shell versus chitinous cuticle) than feeding behaviour, so that bivalves and gastropods with hard shells exhibited

lower chromium concentrations than did polychaetes and other worms with an unprotected body surface. In fresh waters, however, Smock (1983a) observed that feeding behaviour was very important in determining chromium concentrations in aquatic insects collected from river systems in central North Carolina. In his classification scheme, sediment-dependent insects (i.e., deposit feeders living within the sediment and indiscriminately ingesting it along with detritus) had significantly higher chromium levels than did filter feeders (animals that live on the sediment and remove algae, detritus and other suspended material from the water column), sediment-associated organisms (omnivores living on and within the sediment), carnivores or surface feeders (predators living on an organism, trapped in the surface film).

Ayling (1974) observed that accumulation of chromium by oysters (i.e., *Crassostrea gigas*) was independent of sediment chromium levels but dependent on oyster size. This suggested to him that there was some sort of homeostatic regulation of this metal. The possibility of homeostatic regulation of chromium has also been observed in other marine organisms. For example, in mesocosm experiments, chromium was not accumulated from pulverised fuel ash above control levels by the polychaete, *Nereis virens* (Jenner and Bowmer 1992), or by three intertidal invertebrates, the cockle, *Cerastoderma edule*, the clam, *Macoma balthica,* and the lugworm, *Arenicola marina* (Jenner and Bowmer 1990). More recently, Williams et al. (1994) suggested that chromium may be regulated within plant tissues because chromium concentrations were consistent among, and within, halophilic plant species at several salt marsh sites in Essex.

There also appears to be a trend towards decreasing chromium concentration with increasing weight, shell length, size, and age for benthic organisms. Chromium has been found to be inversely correlated with age, weight and/or size in marine organisms, including *Mytilus spp.* (Brix and Lyngby 1985; Lyngby 1991; Catsiki et al. 1991), *Scrobicularia plana* (Bryan and Uysal 1978), and *Saccrostrea sp.* (Paez-Osuna and Marmolejo-Rivas 1990) and in the freshwater species, *Lampsilis radiata* (Metcalfe-Smith et al. 1995) and *Lymnaea stagnalis*, (a pond snail) (V.-Balogh et al. 1988). No relation was found between chromium concentration and shell length in the marine mussel, *Perna viridis,* (Cheung and Wong 1992) or between chromium concentration and dry weight in the marine clam, *Scrobicularia plana* (Bryan and Hummerstone 1978). Chromium concentration also has been negatively correlated with fish length (Hornung and Ramelow 1977), fish weight (Elwood et al. 1980) and fish age (Tong 1974). However, Campbell (1995) found no significant correlation between chromium concentration and the length and weight of three species of freshwater centrarchid fish in Florida. It has been suggested that a negative relationship may arise if metal uptake by smaller or younger individuals is more rapid than by larger or older

organisms (Brix and Lyngby 1985; Catsiki et al. 1991). On the other hand, the lower concentrations of chromium in larger, older animals may be a reflection of metal dilution (Brix and Lyngby 1985). Smock (1983b) propose that in freshwater invertebrates, the inverse trend is a result of surface adsorption of chromium.

The wide variation in the results discussed above emphasizes the confusion in assessing results from laboratory and field investigations where different chromium species and also different organisms have been examined. In laboratory experiments using marine organisms, Chassard-Bouchard et al. (1991), reported that most of chromium uptake in *Mytilus edulis* occurred through the gills. Peternac and Legovic (1986) also found that by far the highest concentration of Cr(III) and Cr(VI) was measured in the gills of the crab, *Xantho hydrophilus*. However, evidence would suggest that for certain organisms, uptake of Cr(III) from food and sediment particles may be more important than direct uptake of the soluble Cr(VI) metal from water (Phelps et al. 1974; Preston 1971; Bryan and Langston 1992). The high concentration of chromium in the digestive gland relative to other tissues also would support ingestion as a primary route of uptake, for example in marine gastropods (Nott and Nicolaidou 1989) and bivalve molluscs (Bryan and Uysal 1978; Young et al. 1979), and in sediment-dependent freshwater insects (Smock 1983a,b). Ferreira et al. (1990) measured relatively similar chromium levels of 1.0 to 3.3 $mg \cdot kg^{-1}$ dw in the mantle, 0.3 to 2.3 $mg \cdot kg^{-1}$ dw in the gills, 0.2 to 3.3 $mg \cdot kg^{-1}$ dw in the muscle, and 0.2 to 3.7 $mg \cdot kg^{-1}$ dw in the remaining soft tissues of 68 composite samples of the oyster, *Crassostrea angulata* collected from the Sado Estuary, Portugal. In addition, partitioning of chromium within the oysters seemed to be related to environmental conditions such as salinity.

Laboratory studies in which the relative uptake of Cr(VI) versus that of Cr(III) from sediment or water, or uptake/depuration kinetics of chromium are examined, are scarce. This is unfortunate because the data would be of great use for understanding the mechanism of chromium uptake and an organism's ability to eliminate this metal. Decho and Luoma (1991) compared the ingestion, retention, and release via the faeces of ^{51}Cr(III)-labelled food in two intertidal bivalves (*Potamocorbula amurensis* and *Macoma balthica*) having different feeding strategies. Loss of chromium was tri-phasic in both clams, with each phase being much shorter in *P. amurensis*. Half-life of the ^{51}Cr for the three phases/pathways (i.e., intestinal, glandular, and physiological) was 8.6 h, 115 h, and 138 h, respectively in *M. balthica* and 5.9 h, 99 h, and 693 h, respectively, for *P. amurensis*. Thus the overall gut passage time varied by an order of magnitude between the two species (i.e., 9.6 ± 1.8 h in *M. balthica* versus 0.86 ± 0.8 h in *P. amurensis*).

In other depuration studies involving marine invertebrates, Van Weerelt et al. (1984) found that the half-life of Cr(VI), incorporated by barnacles in a laboratory experiment in which ^{51}Cr was added to unfiltered seawater, was 120 days. Trivalent chromium that was added to the sea water was precipitated and removed from the water by the filtering activity of the barnacles. Unlike Cr(VI), Cr(III) did not concentrate in the soft tissue of the barnacles. In the crab, *Xantho hydrophilus*, Peternac and Legovic (1986) observed that the loss rate of both Cr(VI) and Cr(III) was inversely proportional to exposure time. The half-life for Cr(VI) was 7 d and 101 d following uptake of 2 d and 35 d, respectively. For Cr(III), the half-life was 27 d and 64 d following uptake of 2 d and 28 d, respectively. Marmolejo-Rivas and Paez-Osuna (1990) determined that the half-life of chromium in the mussel, *Mytella striata,* was approximately one month. In summary, the half-life of chromium that has been measured in various experiments ranges between approximately 6 and 120 days, and is dependent on the organism, the exposure time, and the chromium species.

Biomagnification

Although there is general consensus regarding the absence of biomagnification of chromium throughout the food chain (Bryan and Uysal 1978; Friant 1979; Biddinger and Gloss 1984; Mance 1987; Holdway 1988; Janus and Krajnc 1990; Hatcher et al. 1992), some evidence for food chain transfer of chromium does exist. For example, it has been reported that Cr-rich periphytic bacteria living on the carapace and gills of marine crabs (Johnson et al. 1981), or epiphytic bacteria living on freshwater aquatic plants (e.g., *Alisma plantago-aquatica*; Patrick and Loutit 1977), or marine sediment bacteria (Bremer and Loutit 1986; Aislabie and Loutit 1986) might be ingested by fish and other biota and thus act as a vehicle for entry into the food chain. In addition, the transfer of chromium from sediments to the terrestrial food chain has been suggested through birds feeding directly on benthic invertebrates (Kraus 1989) or insects that have emerged from benthic insect larvae (Krantzberg 1985).

It is also thought that contaminated sediments may be contributing to enhanced levels of chromium in the tissues of certain bottom-feeding fish (Mearns et al. 1976; Winger et al. 1990), aquatic macrophytes (Mudroch and Capobianco 1979), mosses (Mouvet et al. 1987), molluscs (Nicolaidou and Nott 1990; Luoma et al. 1990; Mimicos et al. 1991) and other benthic invertebrates (Aislabie and Loutit 1986; Duzzin et al. 1988; Krantzberg and Boyd 1991). However, as can be seen in Table 5, concentrations of chromium in both marine and freshwater sediments are generally as high or higher than the concentrations of chromium in the organisms dwelling on or in them.

Furthermore, these concentrations do not increase with increasing trophic level. Mathis and Cummings (1973) found that chromium levels in the Illinois River were highest in the sediment, followed by tubificid worms > clams > omnivorous fish > carnivorous fish > river water. A comparable chromium concentration order (i.e., sediments > seston > plankton > benthic particle feeder > bottom feeding fish) was found by Phelps et al. (1974) for Naragansett Bay/Rhode Island Sound and also by Ramelow et al. (1989) (i.e., periphyton > zooplankton > oyster > mussel > shrimp > fish) for the Calcasieu River and Calcasieu Lake, Louisiana. In addition, Kurita and Pfeiffer (1991) reported that chromium concentrations in molluscs collected from Sepetiba Bay, Brazil averaged 2.2 $mg \cdot kg^{-1}$ dw, with fish concentrations reported as 10 times lower. Vukadin and Odzak (1991) reported that chromium levels in mussels collected from Kastela Bay (middle Adriatic Sea) were about 100 times lower than that in the sediment. On the other hand, Campbell (1995) recently discovered that largemouth bass (*Micropterus salmoides*, a predator) had higher chromium concentrations than either red ear sunfish (*Lepomis microlophus*, a bottom feeder) or bluegill sunfish (*Lepomis macrochirus* , an omnivore) that were living in stormwater treatment ponds in Florida. This is in direct contrast to the results of Mathis and Cummings (1973) who found that chromium concentrations in omnivorous fish species collected from the Illinois River were greater than those in carnivorous fishes.

Toxicity

Similar to the approaches used for assessing bioavailability, toxicity can be determined from: laboratory experiments, field experiments, and field surveys. Laboratory experiments include bulk sediment-, interstitial water-, elutriate- and spiked-sediment toxicity tests carried out with natural or artificial sediments. Field experiments involve, for example, the transfer of benthic organisms between contaminated and uncontaminated sites (Mersch and Johansson 1993; Nelson et al. 1995) or the use of field plots, called mesocosms (Schulz-Baldes et al. 1983; Jenner and Bowmer 1990), which are subjected to different experimental manipulations. Field surveys entail examining relationships between sediment characteristics (such as chromium concentration, loss on ignition, grain size) and either the presence/absence of benthic species or the incidence of adverse effects of the contaminant on indigenous biota (such as reduced number of offspring, delayed time to hatching of eggs, etc.).

Spiked Sediment Toxicity Tests (SSTT) Approach

Probably the most direct approach for determining the toxicity of chromium in sediments is to employ spiked- sediment toxicity tests. Unfortunately, laboratory experiments in which artificial or natural sediments are spiked with chromium and the subsequent toxicity to a benthic or planktonic organism is assessed are scarce, in part because of the difficulty in spiking artificial sediment with chromium (Trefor Reynoldson, NWRI, Canada, 1995, pers. com.) Dave (1992) spiked sediment from Lake Hyälmaren, Sweden by mixing 5.0 g of fresh sediment with a concentrated stock solution of $KCr(SO_4)_2 \cdot 12H_2O$ [i.e., Cr(III)] or $K_2Cr_2O_7$ [i.e., Cr(VI)]. After addition of dilution water to a final volume of 50 mL (10% wet sediment in water) and mixing, the suspension was allowed to settle and equilibrate in a Petri dish for 3 d at 20°C. Twenty juvenile *Daphnia magna* (4 to 5 d old) were then added and mobility was recorded after 24 h and 48 h. The reported 24 h and 48 h EC_{50} values were 436 $mg \cdot kg^{-1}$ and 195 $mg \cdot kg^{-1}$, respectively for Cr(III) and 170 $mg \cdot kg^{-1}$ and 167 $mg \cdot kg^{-1}$ for Cr(VI). However, these figures reflect the amount of chromium added to the control sediment (from Lake Hyälmaren) and do not include the background concentration of chromium of 92 $mg \cdot kg^{-1}$. From this experiment, it is evident that Cr(VI) is more toxic (or more available to the *Daphnia*) than Cr(III).

In marine studies, Gardner et al. (1992) added $K_2(VI)CrO_4$ to a reference sediment from Long Island Sound and measured chromium concentrations that were similar to (1 460 $mg \cdot kg^{-1}$) and about 10 times greater than (14 600 $mg \cdot kg^{-1}$) those measured in contaminated Black Rock Harbour sediments (1 480 $mg \cdot kg^{-1}$). Although tumours developed in oysters (*Crassostrea virginica*) exposed for a 30-day period, to all three sediments in a flow-through test system with an overlying water concentration of 20 $mg \cdot L^{-1}$, the tumours could not be attributed specifically to the chromium because 10 other inorganic and organic contaminants had been added to the sediment concurrently with chromium. Capuzzo and Sasner (1977) measured the accumulation and toxicity of chromium in natural sediment (polluted by tannery wastes) and in artificial sediment (either bentonite or kaolinite treated with $CrCl_3$) using *Mytilus edulis* and *Mya arenaria*. Using a flow-through system, after 24 weeks exposure to natural sediment, the filtration rate of *M. edulis* was significantly reduced in the sediment containing chromium concentrations of 150 $mg \cdot kg^{-1}$ clay and 950 $mg \cdot kg^{-1}$ clay, although there was no reduction in the control sediment (10 mg/kg clay). Exposure to artificial sediment (bentonite containing 1 000 mg/kg^{-1} or kaolinite containing 1 200 $mg \cdot kg^{-1}$) for 4 to 6 weeks also resulted in significantly reduced filtering rates, except for *M. arenaria* exposed to bentonite. In contrast, no effect of Cr(III) was observed on juvenile *Neanthes arenaceodentata*. Mearns et al. (1976) added $CrCl_3$ to seawater, but it immediately formed a

precipitate that settled to the bottom of the jar. *N. arenaceodentata* living in this precipitate showed no apparent adverse effects from exposure to Cr(III); they were able to build mucous tubes and spawn, and there was no significant difference in brood size.

In conclusion, the few results from spiked-sediment toxicity tests would indicate that Cr(III) is less toxic than Cr(VI). These toxic concentrations are lower than concentrations of chromium that have been measured in contaminated freshwater and marine sediments in Canada (see Chapter 2).

Bioassays

A common approach for determining sediment toxicity is by means of bioassay experiments during which test organisms are exposed to whole sediment, interstitial water, or elutriate obtained from natural sediments. A thorough discussion of the sediment toxicity procedures that are available for both animals (Swartz 1987; Giesy et al. 1988; Burton 1991) and plants (Munawar and Munawar 1987; Lewis 1995), including recommendations and methods for extracting interstitial water and elutriates (Sasson-Brickson and Burton 1991; Schults et al. 1992; Bufflap and Allen 1995; Carr and Chapman 1995) and the handling and storing of sediment (Swartz et al. 1985; Othoudt et al. 1991; Reynoldson et al. 1991; Pagano et al. 1993; Environment Canada 1994), is beyond the scope of this report. Sufficed to say, the major disadvantage with bioassay procedures utilizing natural sediments, interstitial waters or elutriates, in addition to the sampling difficulties, is that any observed deleterious effects cannot necessarily be attributed to a specific contaminant. In other words, a direct cause-effect relationship for an individual contaminant cannot be established. For this reason, scientists have tended towards using correlation (Laskowski-Hoke and Prater 1984) and multivariate analyses, such as discriminant function and principal component analyses, in an effort to obtain evidence regarding individual metal toxicity (Santiago et al. 1993; Reynoldson 1994, Schlekat et al. 1994).

As a result of these efforts, as well as other circumstantial evidence, information concerning chromium toxicity can be derived from bioassay experiments. In marine sediments, Long et al. (1990) found that during their bioassay experiments, three samples collected from Tomales Bay, California, were among the most toxic despite the fact that most of the chemicals analyzed were the least concentrated in those samples. However, chromium and nickel levels were elevated in 2 of the 3 toxic samples, with chromium reaching concentrations of about 235 $mg \cdot kg^{-1}$ (compared to 144 to 190 $mg \cdot kg^{-1}$ in 12 of the other 13 samples). In support of this, Schlekat et al. (1994)

reported no significant correlation between survival and growth of *Hyalella azteca*, macroinvertebrate richness and abundance, and chromium concentrations for sediment levels of 69 to 156 $mg \cdot kg^{-1}$ in tidal river sediments in the Washington, D.C. area. Similarly, Winger and Lasier (1995) did not find a significant correlation between survival of *Hyalella azteca* exposed to either whole sediment or interstitial water collected from the Savannah Harbour, Georgia, and sediment chromium concentrations between 10 and 60 $mg \cdot kg^{-1}$. However, for chromium levels of 110 $mg \cdot kg^{-1}$, 357 $mg \cdot kg^{-1}$, and 500 $mg \cdot kg^{-1}$ in southern California, Thompson et al. (1989) reported a significant correlation between mortality, reduced growth, and reduced female gonad development, respectively, in the sea urchin, *Lytechinus pictus*, for contaminated sites compared to control levels of 32 $mg \cdot kg^{-1}$.

In fresh waters, Reynoldson (1994) conducted whole sediment bioassay experiments with the oligochaete, *Tubifex tubifex*, on sediment collected from 67 sites located in 11 areas of concern, as well as 5 sites located in a reference area, in the Laurentian Great Lakes. Using cluster analysis, he was able to identify 6 site groups with chromium concentrations being elevated in the sediments of two of the four groups of toxic sites. Chromium concentrations in these two toxic groups were 167.5 $mg \cdot kg^{-1}$ and 483.5 $mg \cdot kg^{-1}$ (within the contaminated chromium concentrations reported for Ontario lakes; Table 3) compared to concentrations of 41.5 to 97.6 $mg \cdot kg^{-1}$ in the other site groups. Santiago et al. (1989) also demonstrated that the uptake of ^{14}C by freshwater phytoplankton assemblages exposed to various dilutions of sediment elutriates obtained from Lake Geneva and its tributaries was significantly reduced (30 to 40%) at one site, where sediment chromium concentrations were 228 $mg \cdot kg^{-1}$ (elutriate concentration of 6.1 $\mu g \cdot L^{-1}$). However, there was little or no effect at chromium levels of 38 to 84 $mg \cdot kg^{-1}$ (elutriate concentration < 2 $\mu g \cdot L^{-1}$). Wentsel et al. (1978) found that emergence of chironomid larva was reduced over three times and delayed for two days when the insects were exposed for 14 days to sediment collected from Palestine Lake, Indiana, which contained 1 030 $mg \cdot kg^{-1}$ cadmium, 17 300 $mg \cdot kg^{-1}$ zinc, and 1 640 $mg \cdot kg^{-1}$ chromium. On the other hand, Westerman (1989) reported that survival of fathead minnows, *Pimephales promelas*, in whole sediment bioassay experiments was reduced by about 80% (at Site 4) for chromium concentrations ranging from 21 to 52 $mg \cdot kg^{-1}$, and by about 54% (at Site 5) for chromium concentrations of 22 to 37 $mg \cdot kg^{-1}$ in stream sediment collected from drainage of the Yellow Creek in Kentucky. However, it is likely that these toxicological responses were a result of elevated concentrations of metals other than chromium, in particular cadmium.

One issue pertaining to the question of filtering interstitial water and elutriates after collection and/or prior to bioassay experiments, needs to be mentioned. For chromium, the issue regarding filtering

is of particular importance because of the affinity for Cr(III) to sorb onto dissolved and suspended matter. Neff et al. (1980) found that >75% of the chromium in the unfiltered mud aqueous fraction (i.e., elutriate obtained from used chrome lignosulphonate drilling mud) was in the trivalent form. Likewise, Qasim et al. (1980) found that filtering removed most of the chromium in water collected from the Trinity River, Texas, and Schubauer-Berigan and Ankley (1991) reported that chromium concentrations in whole interstitial water collected from an Illinois River tributary were 38 $\mu g \cdot L^{-1}$ compared to values of only 5 $\mu g \cdot L^{-1}$ after filtration.

Schubauer-Berigan and Ankley (1991) point out that methods for recovering interstitial water that include filtration may eliminate a major component of the sediment contaminants that are responsible for toxicity because of the removal of particle associated chemicals. Evidence for decreasing toxicity following filtration has been reported by Sasson-Brickson and Burton (1991) who found that the survival of *Ceriodaphnia dubia* in Little Scioto River, Ohio, sediment phases increased from 0% survival in unfiltered water to 56.7% in filtered interstitial water and from 10% to 86.7% in unfiltered and filtered elutriate, respectively. Neff et al. (1980) also observed that the filtered mud aqueous fraction, was slightly less toxic than the unfiltered mud aqueous fraction to marine worms.

In conclusion, observed toxicity may or may not decrease following filtration, depending on the ecology of the organism. However, even if filtration removes much of the Cr(III), there is some evidence that Cr(III), bound to dissolved organic matter is also unavailable to certain biota. Stackhouse and Benson (1989) reported that humic acid concentrations of 50 $mg \cdot L^{-1}$ significantly decreased the toxicity of both chromic chloride and chrome lignosulphonate [both Cr(III)] to *Daphnia pulex* and that concentrations of only 5 $mg \cdot L^{-1}$ decreased the percentage free Cr(III) (as $CrCl_3$). On the other hand, the acute toxicity and bioavailability of Cr(VI) to the *Daphnia* was unaffected by humic acid levels of 0.5 to 50 $mg \cdot L^{-1}$ (Stackhouse and Benson 1988).

Field Surveys

A third approach to assessing the toxicity of chromium in sediments is to use the information from field surveys to examine relationships between sediment characteristics (such as chromium concentration, loss on ignition, grain size) and the presence/absence of benthic species or the incidence of adverse effects of the contaminant (such as reduced number of offspring, delayed time to hatching of eggs, etc.) on indigenous or transplanted species. Unfortunately, under field

conditions, examples of deleterious effects on benthic organisms that can be attributed to specific metals are rare (Bryan and Langston 1992). Furthermore, measured chemical and physical parameters do not always account for all the observed differences in community structure or abundance, suggesting some unmeasured toxicity (Lenat and Crawford 1994).

Although several authors have presented data on the community structure or abundance of benthos being affected by sediment contamination (Winner et al. 1980; Cotta Ramusino et al. 1981, 1983; Duzzin et al. 1988; Edsall et al. 1991; Ferraro et al. 1991; Griffiths 1991; Hatcher et al. 1992; Lenat and Crawford 1994) few, if any, can be related specifically to chromium contamination. Dave (1992) reported that benthic abundance and biomass in the River Kolbäcksån, Sweden, sediments were correlated with 24 h EC_{50} values derived from *Daphnia magna* whole sediment toxicity tests. Nugent et al. (1980) observed a 10 fold decrease in microbial numbers in Palestine Lake, Indiana, which was affected by inputs of chromium, cadmium and zinc from an electroplating plant about 3 km away. Although chromium levels in the sediment reached 4 380 ± 1 910 $mg \cdot kg^{-1}$ they could not say conclusively that the observed effects were solely a result of high chromium because of the presence of the other two metals. Similarly, Dickman et al. (1990) conducted an extensive survey of benthic invertebrates and aquatic plants downstream of Atlas Specialty Steels Co. in the Niagara River near Welland, Ontario. They identified four zones in the distribution of plants and animals downstream of the sewer discharge; as distance downstream of the sewer discharge increased, abundance and diversity of the flora and fauna also increased. However, although an exceedingly high concentration of chromium (i.e., 5 120 $mg \cdot kg^{-1}$) was measured at station 67S (located in Zone 3), it is not known whether the observed community structure was a result of chromium, or more likely, a combination of the effects of high chromium, lead, iron, nickel, zinc, copper, and/or other metal concentrations.

CHAPTER 4. DEVELOPMENT OF CANADIAN SEDIMENT QUALITY GUIDELINES

Chromium, a naturally-occurring metal, is imported into Canada in substantial quantities and used in many production processes. The extraction and use of chromium has resulted in anthropogenic inputs of chromium into Canadian aquatic environments, and subsequently elevated levels of chromium in sediments have been reported (Chapter 2). Elevated chromium levels are in the same range as those reported to be associated with toxicity in aquatic biota (Chapter 3). Exposure of benthic organisms to chromium in sediment (whether of natural or anthropogenic origin) may result in uptake of, and adverse biological effects associated with this metal. Effective assessment of the potential ecological impact of chromium in sediments requires an understanding of relationships between chromium concentrations in sediments and the occurrence of adverse biological effects. National sediment quality guidelines are scientific tools that synthesize information regarding such relationships in a form that is easily communicated and understood. In Canada, they are developed using toxicological information and are intended to be national benchmarks (i.e., reference points) to be used in making decisions regarding the protection and management of sediment quality.

A number of jurisdictions in North America and elsewhere have developed sediment quality guidelines or other sediment quality assessment values (e.g., criteria, objectives, or standards) to assist in the assessment and management of sediment quality (Appendix Ia and Ib). Individual jurisdictions have adopted philosophically different approaches to the development of such values, depending on the receptors that are to be protected (e.g., sediment-dwelling organisms, wildlife, humans), the degree of protection afforded (e.g., protection of all species, protection of commercially important species), the geographic extent to which the values apply, and their intended uses (e.g., as scientific benchmarks, screening tools, remediation goals). As a result, sediment quality guidelines or other assessment values established for chromium in differ among jurisdictions.

In Canada, the CCME has developed a formal protocol to derive numerical sediment quality guidelines for both freshwater and marine (including estuarine) sediments to protect aquatic life associated with sediments (CCME 1995). This protocol relies on the National Status and Trends Program (NSTP) approach (with modifications) and the Spiked-Sediment Toxicity Test (SSTT) approach. Subsequent to an evaluation of the toxicological information, Canadian sediment quality guidelines (also referred to as 'full' sediment quality guidelines) are recommended if information exists to support both approaches. Generally, the lower of the two values derived using either the modified NSTP approach or SSTT approach is recommended. (However, it should be noted that,

while sufficient data currently exist to calculate SSTT values for several substances, concerns regarding spiked-sediment toxicity testing methodology limit the degree to which these values may be used as the scientific basis for recommending full sediment quality guidelines at this time). Interim Canadian sediment quality guidelines are recommended if information is available to support only one approach. The guidelines may also be derived to reflect predictive relationships that have been established between the concentration of the chemical in sediments, and any environmental factor or condition (e.g., characteristics of the sediment, such as the concentration of organic carbon; characteristics of the overlying water column, such as hardness) that clearly modifies the expression of the toxicity of the chemical. If insufficient information exists to derive interim guidelines using either the modified NSTP approach or the SSTT approach, guidelines from other jurisdictions are evaluated and may be adopted in the short-term as provisional sediment quality guidelines.

Details of the derivation and evaluation of Canadian interim sediment quality guidelines for chromium for both freshwater and marine sediments are outlined in the following sections. At present, insufficient information exists to derive a sediment quality guideline for chromium using the SSTT approach. Interim sediment quality guidelines for chromium, therefore, have been derived using the modified NSTP approach only. CCME (1995) should be consulted for a detailed description of the protocol and its supporting rationale.

The Modified National Status and Trends Program (NSTP) Approach

The NSTP approach to deriving sediment quality assessment values was originally developed by Long and Morgan (1990). This approach was adopted, with some modifications, for use in developing sediment quality guidelines in Canada (CCME 1995), and is therefore referred to in this document as the *modified* NSTP approach. Modifications to this approach include the separate evaluation of information for freshwater and marine systems, an expansion of the information originally compiled, and the use of derivation procedures that consider all of the compiled information (see also Long and MacDonald 1992; MacDonald 1994; CCME 1995; and MacDonald et al. 1992 for more detailed explanations of these modifications).

The NSTP approach primarily relies on the use of field-collected data, in which chemical mixtures occur, to derive sediment quality assessment values (Long and Morgan 1990; Long 1992; Long and MacDonald 1992; MacDonald 1994; CCME 1995; and Long et al. 1995). This approach

involves the evaluation and compilation of sediment quality data from numerous studies conducted throughout North America, in which sediment chemical and biological data have been collected synoptically (also referred to as co-occurrence data). Data for individual studies are evaluated to establish an *association* between the concentration of each of the chemicals measured in the sediments and any adverse biological effect observed. Cause-and-effect relationships between the concentration of individual chemicals and the observed adverse biological effect cannot be inferred from this evaluation. This information is compiled in a database, referred to as the Biological Effects Database for Sediments (BEDS).

Sediment chemical and biological data currently included in BEDS were obtained from various studies, including models of equilibrium partitioning in sediments, sediment quality assessment values from other jurisdictions (e.g., those derived using the apparent effects threshold approach and the screening level concentrations approach), spiked-sediment toxicity tests, and field studies (including acute and chronic toxicity results from sediment bioassays and analyses of benthic community composition). Candidate studies were critically evaluated according to a number of screening criteria to ensure that high quality data sets were incorporated into BEDS and that the information compiled was internally consistent. These screening criteria (e.g., appropriate sediment handling procedures, acceptable toxicity test procedures) have been described in detail elsewhere (MacDonald 1994; CCME 1995; Long et al. 1995; Environment Canada 1996).

Information contained in BEDS was sorted for each chemical and sediment type (i.e., freshwater versus marine) and arranged in ascending order of the chemicals' concentration to produce separate *ascending data tables,* or *guideline derivation tables*, for individual chemicals and sediment types (CCME 1995; MacDonald 1994; Long et al. 1995). For chromium, these tables summarize the compiled information that associates chromium concentrations with either adverse effects or no adverse effects on aquatic organisms (also referred to as the effect data set and the no-effect data set, respectively; Appendix IIa,b). Concentrations are expressed as total concentrations in sediments on a dry-weight basis.

The guideline derivation tables for individual chemicals (e.g., chromium; Appendix IIa,b) consist of a number of entries, each of which belongs to either the effect data set (concentrations associated with an adverse biological effect) or the no-effect data set (concentrations not associated with an adverse biological effect), that are sorted according to ascending chromium concentrations. Entries in the guideline derivation tables were designated as being associated with an effect (an asterisk in the 'Hit' column) if an adverse biological effect was reported. These effects included

acute or chronic toxicity observed during a controlled spiked-sediment test, apparent effect thresholds (concentrations above which specific biological effects would always be expected), and predicted toxicity based on equilibrium partitioning theory (which determines a sediment chemical concentration that ensures the concentration of the chemical in the interstitial water does not exceed the water quality guideline for that chemical at equilibrium). An entry was also assigned an asterisk (effect descriptor) if concordance was apparent between the observed biological response and the measured chemical concentration in a field (i.e., co-occurrence) study. As was described in Chapter 3, concentrations of individual chemicals reported in field studies were considered to be associated with the observed toxic response (i.e., concordance was apparent) if the mean concentration at sites at which significant adverse effects were observed was a factor of two or more greater than the mean concentration at sites at which effects were not observed (i.e., at toxic versus non toxic sites) (Long et al. 1995). A factor of two was chosen to ensure that the difference in the response (i.e., adverse effect) was associated with a significant difference in the chemical concentration. For each chemical, all of the entries designated by an asterisk, as described above, are collectively referred to as the effect data set.

All entries in the guideline derivation tables other than those designated by an asterisk are collectively referred to as the no-effect data set, and are represented by those entries for which chromium concentrations were not associated with adverse biological effects. These entries include those associated with non-toxic, reference, or control conditions (i.e., no effects; NE), as well as those for which toxicity may have been observed but the mean chromium concentration differed by less than a factor of two between the toxic and non-toxic groups (i.e., no gradient, NG; small gradient, SG; or no concordance, NC). In the latter case, it was assumed that other factors (whether measured or not) were more important in the etiology of the observed effect than the chemical concentration (Long et al. 1995).

Individual entries in the guideline derivation tables consist of the chromium concentration in the sediment, an indication of whether this concentration was part of the effect data set (i.e., as indicated with an asterisk *) or the no-effect data set (i.e., NE, SG, NG, or NC), location of the study, analysis type (or approach used), test duration, toxicological end-point measured, species and life stage tested, information on the characteristics of the sediments (e.g., concentrations of TOC or AVS, physical classification of the sediments; when available), and the study reference. In most cases, information for individual entries (i.e., copper concentrations and sediment chemical and physical characteristics) represents the means of several samples. Standard deviations of these means are provided wherever possible. The following discussion summarizes the

information contained in the freshwater and marine guideline derivation tables for chromium (Appendix IIa,b).

Guideline Derivation Tables for Chromium

The freshwater guideline derivation table for chromium (Appendix IIa) contains 452 entries. A major proportion of the freshwater entries were from field (COA) studies (96.5%), while the remaining data are numerical sediment quality guidelines from other jurisdictions. The marine guideline derivation table for chromium (Appendix IIb) contains 354 entries. Similar to the freshwater guideline derivation table, the majority of data entries in the marine table are results from field studies (96.9%). Chromium concentrations in sediment range from 5.5 to 1954 $mg \cdot kg^{-1}$ and 0.6 to 1646 $mg \cdot kg^{-1}$ in the freshwater (Appendix IIa) and marine (Appendix IIb) guideline derivation tables, respectively. The lowest concentration of chromium in field collected sediment that was associated with adverse effects was 29.7 $mg \cdot kg^{-1}$ and 17.4 $mg \cdot kg^{-1}$ for freshwater and marine sediment, respectively. Both adverse effects and no adverse effects are seen throughout the range of chromium concentrations in the guideline derivation tables. This observation is not surprising, however, as toxicological responses in aquatic biota are affected also by the mixture of chemicals in sediment, as well as other factors that affect organism health (e.g., availability of food, suitable habitat).

A number of freshwater and marine taxa are represented in the guideline derivation tables for chromium, including benthic (e.g., amphipods, clams, worms) and water column (e.g., amphipods, rainbow trout) species. In addition, many different toxicological endpoints are used to link chromium concentrations in sediment with effects on aquatic biota. In the freshwater guideline derivation table, common endpoints include abundance, species diversity, mortality and, less commonly, reproduction, deformities, sexual maturity, and avoidance to contaminated sediments. Similarly in the marine guideline derivation table, common endpoints include mortality, abundance, and species diversity, with the former being the most common of all toxicological endpoints represented. The large variation in aquatic biota and toxicological endpoints represented in the guideline derivation tables for chromium emphasizes an important strength of using the modified NSTP approach; this approach is useful because it draws results from many varied studies, species, endpoints, and sediment conditions and is appropriate for a weight of evidence approach to sediment quality guideline development.

As previously mentioned, physical and chemical data reported in the guideline derivation tables are the means reported in the studies included in BEDS, and not individual data points. Therefore, the ranges of the sediment characteristics summarized below may be narrower than the ranges of these variables reported in the original studies from which the tables were derived and are provided for purposes of sediment characterization only. Nonetheless, these ranges provide an indication of the variety of sediment types represented in the ascending data tables. Sediment characteristics were not reported in all studies found in BEDS. For studies reporting sediment characteristics, TOC content of sediments ranges from 0.18 to 20.4% (n=418) and 0.009 to 15% (n=220) in the freshwater and marine tables, respectively. Acid volatile sulphide is reported in BEDS as $\mu mol \cdot g^{-1}$ or $\mu g \cdot g^{-1}$. For freshwater sediments, AVS ranges from 0.041 to 44.4 $\mu mol \cdot g^{-1}$ (n=64) and 7.4 to 547 $\mu g \cdot g^{-1}$ (n=50). In the marine studies, AVS ranges from 0.004 to 45.3 $\mu mol \cdot g^{-1}$ (n=136) and 9 to 14 009 $\mu g \cdot g^{-1}$ (n=17). Total sulphide concentrations and total free sulphide concentrations were available for marine data and range from 10 to 2277 $mg \cdot kg^{-1}$ (n=42) and 0.33 to 13.6 $mg \cdot L^{-1}$ (n=20). Concentrations of unionized ammonia range from 0.003 to 0.846 $mg \cdot L^{-1}$ (n=52) and 0.006 to 0.08 $mg \cdot L^{-1}$ (n=20) for the freshwater and marine guideline derivation tables respectively. The mean proportions of sand, silt, and clay in freshwater sediments represented in the data tables range from 2.8 to 98.2% (n=308), 1 to 60.8% (n=282) and 0.5 to 92% (n=280), respectively. For marine sediments, the mean proportions of sand, silt, and clay are 0.5 to 100% (n=214), 0.8 to 88.5% (n=158), and 0.4 to 53.4% (n=159) respectively. Thus, the data in the guideline derivation tables represent a wide variety of sediment types.

Derivation of TELs and PELs

As is described in CCME (1995), the derivation procedures used to calculate two assessment values in the original NSTP approach (Long and Morgan 1990) were modified to consider both the effect and no-effect data compiled in the guideline derivation tables. In the modified NSTP approach, a lower value, referred to as the threshold effect level (TEL), represents the concentration below which adverse biological effects are expected to occur rarely. An upper value, referred to as the probable effect level (PEL), defines the level above which adverse effects are expected to occur frequently. Concentrations that fall in the range between the TEL and the PEL are occasionally expected to be associated with adverse biological effects. The definition of these ranges (also referred to as the minimal, possible, and probable effect ranges) is based on the assumption that the potential for observing toxicity of sediment-associated chemicals increases with increasing chemical concentrations (Long et al. 1995).

Minimum toxicological data requirements have been set to ensure that the TELs calculated from the guideline derivation tables provide adequate protection of aquatic life and that these values are supported by a weight of evidence for a broad range of sediment types and characteristics. For a given guideline derivation table, both the effect data set and the no-effect data set must contain at least 20 entries (CCME 1995). These minimum data requirements have been met for both the freshwater (68 effect entries and 384 no-effect entries) and marine (53 effect entries and 301 no-effect entries) guideline derivation tables for chromium. Therefore, TELs and PELs for chromium have been calculated as described below. These two assessment values refer to the total concentration (i.e., as determined following digestion with a strong acid) of chromium in surficial sediments (i.e., upper few centimetres) on a dry-weight basis.

The TEL is calculated as the geometric mean of the lower 15th percentile concentration of the effect data set and the 50th percentile concentration of the no-effect data set as follows:

$$TEL = \sqrt{E_{15} \times NE_{50}}$$

where: E_{15} = the 15th percentile concentration of the effect data set
NE_{50} = the 50th percentile concentration of the no-effect data set

Freshwater Sediments: The TEL for chromium in sediment calculated from data in the freshwater guideline derivation table (Appendix IIa) is:

$$TEL_{freshwater} = \sqrt{42.0 \times 33.1} = 37.3\ mg \cdot kg^{-1}$$

Marine Sediments: The TEL for chromium in sediment calculated from data in the marine guideline derivation table (Appendix IIb) is:

$$TEL_{marine} = \sqrt{59.0 \times 46.3} = 52.3\ mg \cdot kg^{-1}$$

The PELs are calculated from the guideline derivation tables as the geometric mean of the 50th percentile concentration of the effect data set and the 85th percentile concentration of the no-effect data set as follows:

$$PEL = \sqrt{E_{50} \times NE_{85}}$$

where: E_{50} = the 50th percentile concentration of the effect data set
NE_{85} = the 85th percentile of the no-effect data set

Freshwater sediments: The PEL for chromium in sediment calculated from data in the freshwater guideline derivation table (Appendix IIa) is as follows:

$$PEL_{freshwater} = \sqrt{100 \times 81} = 90\ mg \cdot kg^{-1}$$

Marine Sediments: The PEL for chromium in sediment calculated from data in the marine guideline derivation table (Appendix IIb) is as follows:

$$PEL_{marine} = \sqrt{201.0 \times 128.0} = 160.4\ mg \cdot kg^{-1}$$

Evaluation of TELs and PELs

The objective of establishing the TELs and PELs according to a standard formula, as described above, is to consistently define the range of chemical concentrations within which adverse effects rarely occur (i.e., the minimal effect range; below the TEL) and within which adverse biological effects frequently occur (i.e., the probable effect range; above the PEL), respectively. The definition of the TEL is therefore consistent with the definition of a Canadian sediment qualtiy guideline. The PEL is an additional tool that can be used in conjunction with the sediment qualtiy guideline, and other relevant information, in assessing sediment quality.

The incidence of adverse biological effects below the TEL (i.e., within the minimal effect range) and above the PEL (i.e., within the probable effect range), can be used to evaluate the degree to which the TEL and PEL satisfy their objectives (MacDonald 1994; CCME 1995; Smith et al. 1996a,b). The incidence of effects within each range is quantified by dividing the number of effect entries by the total number of entries in that range, and expressing this ratio as a percentage. The TEL and PEL calculated from a guideline derivation table are considered to meet their objectives when the incidence of effects below the TEL is less than 25% and the incidence of effects above the PEL is greater than or equal to 50% (MacDonald 1994; Smith et al. 1996a).

The incidence of adverse biological effects associated with chromium concentrations in freshwater sediments below the TEL is 2.1% (Figure 2). The low incidence of adverse biological effects observed below the TEL suggests that the TEL for freshwater sediments adequately represents a level below which effects are expected to occur rarely. Therefore, sediment chromium concentrations which fall below 37.3 $mg \cdot kg^{-1}$ would not be expected to be associated with adverse biological effects. The incidence of adverse effects between the TEL and the PEL and above the PEL increases to 18.7% and 49.4% respectively (Figure 2). Concentrations that fall in the range between the TEL and the PEL are occasionally expected to be associated with adverse biological effects (CCME 1995). The range of chromium concentrations between the TEL and PEL for freshwater sediments appears to coincide with a range in which adverse biological effects will be occasionally observed (i.e., 18.7% incidence of effects). Concentrations of chromium in freshwater sediments greater than or equal to the PEL are expected to be frequently associated with adverse biological effects. The incidence of effects (49.4%) for chromium does not technically meet the objectives of the PEL (i.e., $\geq$ 50%), using the available data for chromium in BEDS. However, the incidence of effects above the PEL is substantially greater than either the incidence of effects below the TEL or between the TEL and PEL, and would appear to be a good assessment of the chromium concentration above which biological effects would be expected to occur relatively frequently.

The incidence of adverse biological effects associated with chromium concentrations in sediment in the marine guideline derivation table below the TEL is 3.5% (Figure 3). As is the case for freshwater sediments, the low incidence of adverse biological effects observed below the marine TEL suggests that the TEL adequately represents a level below which effects are expected to occur rarely. Chromium concentrations in marine and estuarine sediments that fall below 52.3 $mg \cdot kg^{-1}$ would not be expected to be associated with adverse biological effects. The incidence of adverse effects between the TEL and the PEL, and above the PEL increases to 15.4%

and 52.9%, respectively (Figure 3). The high incidence of effects observed above the PEL suggests that it adequately represents a concentration above which adverse effects would be expected to occur frequently (i.e., an incidence of effects ≥ 50% is observed).

The TEL and PEL values for chromium in marine and freshwater sediments can also be evaluated by comparing them to the toxicity results reviewed in chapter 3. The results of spiked-sediment toxicity tests indicate that toxic chromium concentrations are considerably higher than our TEL. Therefore, the available toxicological information for chromium in freshwater and marine sediments suggest that the TEL and PEL sediment quality assessment values accurately describe the chromium concentrations in sediment which will, and will not, be associated with adverse effects of aquatic biota.

Information in the guideline derivation tables was evaluated further to determine whether the calculated TELs and PELs could, or should, be adjusted to account for variation in chromium toxicity associated with variation in sediment characteristics. In order to determine whether chromium concentrations are correlated with sediment characteristics, Spearman rank-order correlation coefficients were calculated between chromium concentrations in the freshwater and marine guideline derivation tables and sediment characteristics. In addition, analysis of variance (t-test and Mann-Whitney) tests were performed to determine whether chromium concentrations and sediment characteristics differed between effect and no-effect data sets.

For data included in the chromium derivation tables, statistical results indicate that chromium concentrations in both freshwater and marine sediments are significantly correlated with sediment characteristics. For freshwater sediments, chromium concentrations are positively correlated with TOC ($r=0.659$, $p<0.001$, $n=418$), %silt ($r=0.187$, $p=0.002$, $n=282$), %clay ($r=0.583$, $p<0.001$, $n=280$), AVS ($r=0.425$, $P<0.005$, $n=62$), ammonia ($r=0.498$, $p<0.005$, $n=52$), and negatively correlated with %sand ($r=-0.576$, $p<0.001$, $n=308$). Similarly for marine sediments, chromium concentrations are positively correlated with TOC ($r=0.571$, $p<0.001$, $n=221$), %silt ($r=0.416$, $p<0.001$, $n=158$), AVS ($r=0.652$, $p<0.001$, $n=119$, $\mu mol \cdot g^{-1}$; $r=0.866$, $p<0.005$, $n=17$, $\mu g \cdot g^{-1}$), total free sulphide (TFS) ($r=0.849$, $p<0.005$, $n=20$), total sulphide (TS) ($r=0.818$, $p<0.005$, $n=42$), and negatively correlated with %sand ($r=-0.459$, $p<0.001$, $n=213$) and ammonia ($r=-0.680$, $p=0.001$, $n=20$). These results suggest that chromium concentrations are positively associated with sediment that have higher binding capacities (i.e., fine-grained fraction, organic fraction, and sulphide-rich sediments). Conversely, lower chromium concentrations would be expected in sediments with a large sand component. The positive associations between chromium

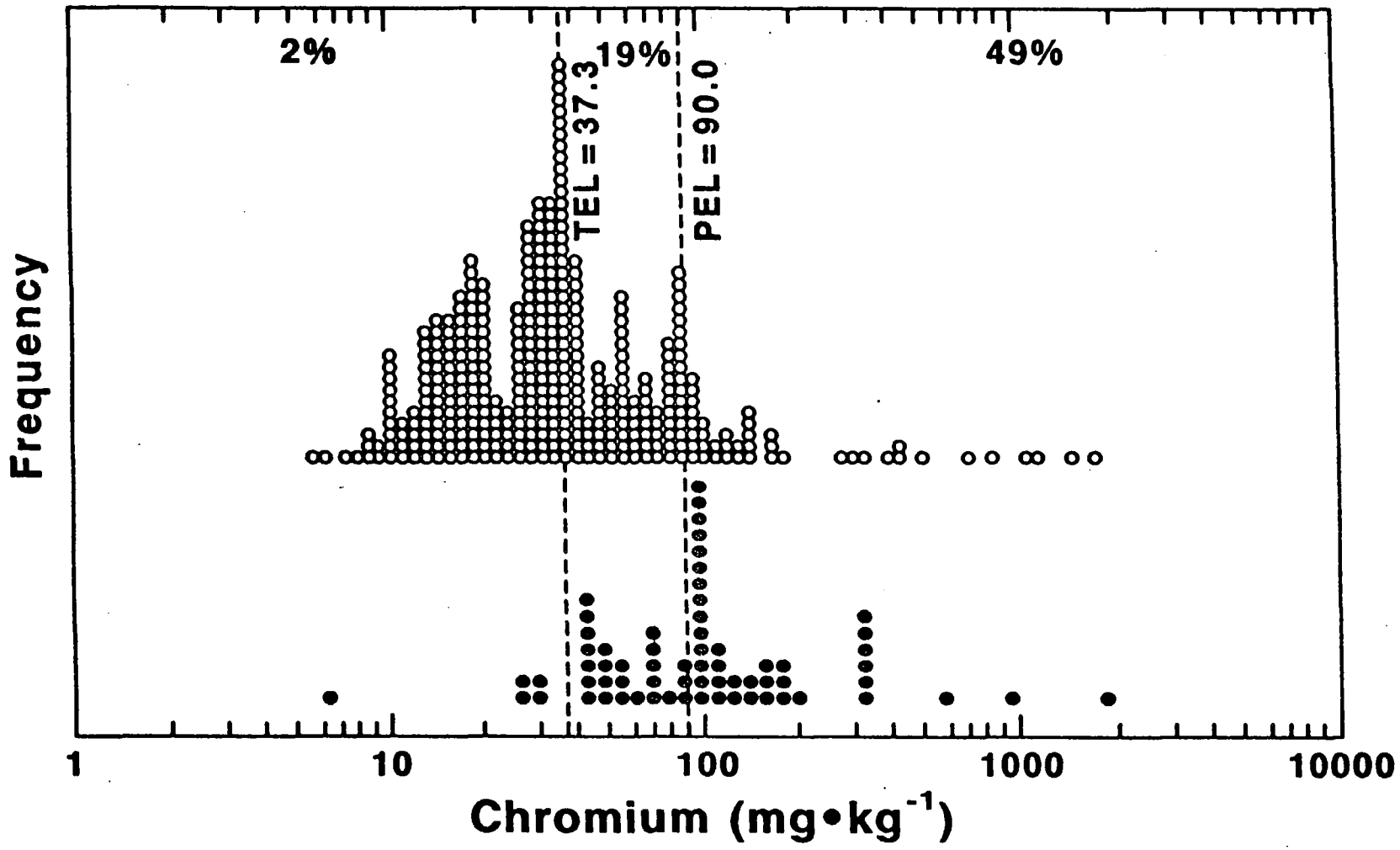


Figure 2. **Distribution of chromium concentrations in freshwater sediments that are associated with adverse biological effects (•) and no adverse biological effects (o). Percentages indicate proportions of concentrations associated with effects in ranges below the Threshold Effect Level (TEL), between the TEL and the Probable Effect Level (PEL), and above the PEL.**

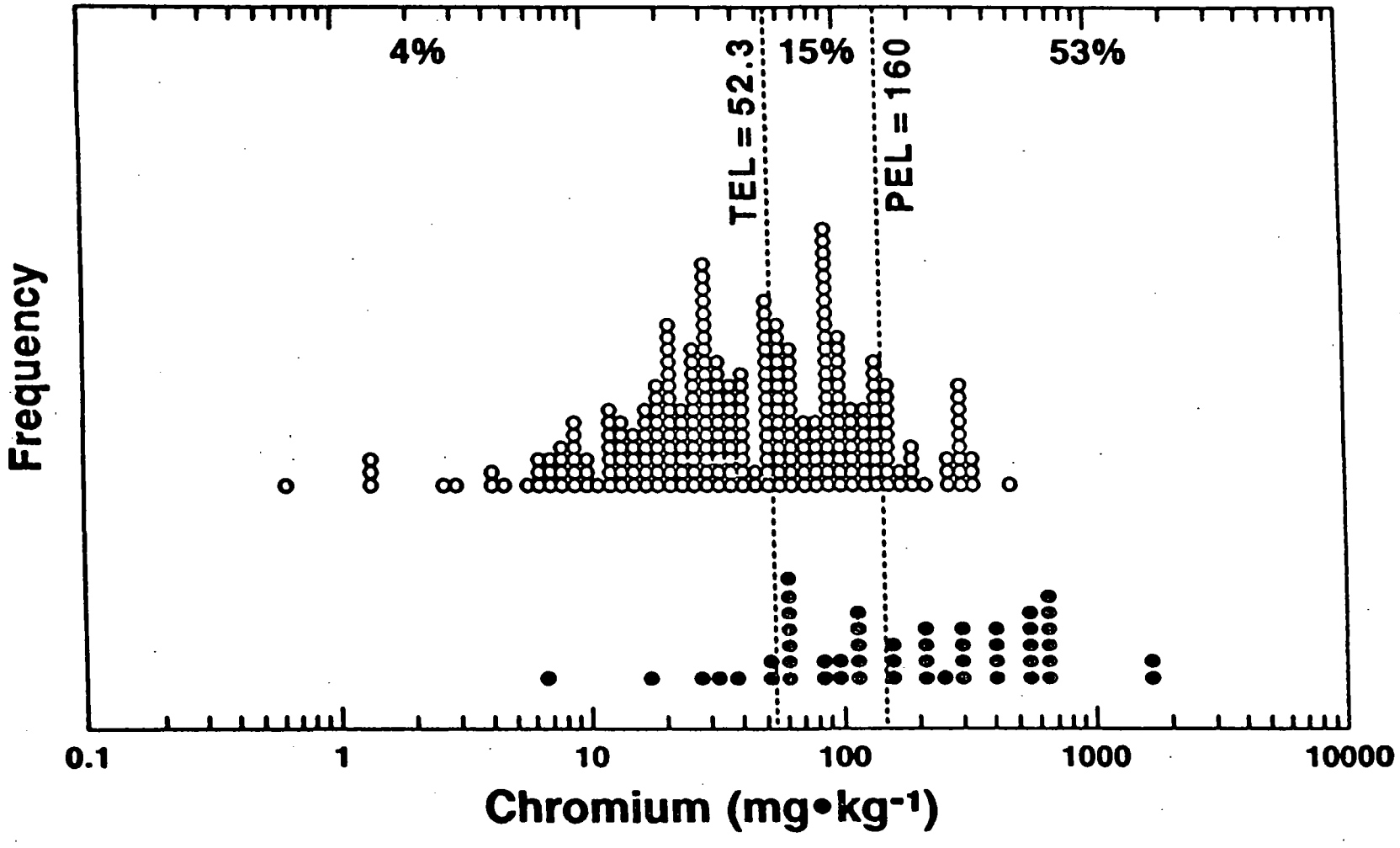


Figure 3. **Distribution of chromium concentrations in marine and estuarine sediments that are associated with adverse biological effects (•) and no adverse biological effects (o). Percentages indicate proportions of concentrations associated with effects in ranges below the Threshold Effect Level (TEL), between the TEL and the Probable Effect Level (PEL), and above the PEL.**

concentrations in sediment and TOC (Loring 1979a; Mayer and Fink 1980; Duzzin et al. 1988; Fuller et al. 1990) and fine-grained sediments (Oliver 1973; Loring 1979a; Rossman 1988; Duzzin et al. 1988; Sager et al. 1990; Anderson 1990; Coakley and Poulton 1993, Weis et al. 1993; Mantei and Sappington 1994; Literathy et al. 1994) have been previously reported.

The implication of the above associations, in the context of adverse effects on biota, is not entirely clear. Higher chromium concentrations occur in sediments with higher binding capacities, however, the binding capacities of such sediments can also limit the availability of chromium, reducing the probability of a toxic response. That is, higher chromium concentrations in sediment will only be important to the health of biota if the chromium is available for uptake.

Differences in chromium concentrations and sediment characteristics between the effect and no-effect data sets in the guideline derivation tables were examined using analysis of variance (t-test) statistics. This analysis was performed to ascertain whether chromium concentrations and sediment characteristics can influence the observations of effects. If chromium concentration, organic content, sulphide, and small particle size in sediment were significantly greater in the effect group, this would suggest that these characteristics increase the potential for toxic response. If the characteristics are significantly lower in the effect group, this could mean that chromium is bound by particles making chromium unavailable for uptake. For the freshwater derivation table, chromium concentrations ($p<0.001$, $n=450$),TOC ($p<0.001$, $n=418$), %silt ($p=0.003$, $n=282$), and AVS ($p=0.004$, $n=62$) were all significantly greater in the effect data set than in the no-effect data set. Similarly, for the marine derivation table, chromium concentrations ($p<0.001$, $n=354$), TOC ($p=0.018$, $n=221$), TFS ($p<0.001$, $n=20$), TS ($p<0.001$, $n=42$), and ammonia ($p=0.036$, $n=20$) were significantly greater in the effect data set.

The finding that chromium concentration is significantly greater in the effect data set is not surprising, and reaffirms an important assumption; the frequency of adverse biological effects will increase with increasing chromium concentration. For sediment characteristics, it appears that adverse biological effects from chromium exposure are more frequently observed in fine-grain sediments with high TOC and sulphide content. These sediment types appear to be a source of chromium to biota, which contradicts the hypothesis that chromium bound to organic carbon, sulphide, and fine-grain sediments will be unavailable for biota uptake.

Information in the ascending data tables does not facilitate an adjustment of the calculated TELs and PELs for any potential role of organic carbon, sulphide, or particle size. Further, evaluations

on individual data sets are necessary to determine definitive relationships among chromium concentrations and sediment characteristics and the relationship to adverse effects on benthic organisms. These relationships should be evaluated further using results from individual field studies (i.e., as opposed to a compilation of information such as that in BEDS) and additional laboratory studies, in order to determine whether quantifiable and predictable relationships between sediment characteristics, chromium concentration, and toxicity can be established for a range of sites and/or conditions. Likewise, relationships between particle size and observed toxicity in field-collected sediments should be examined further.

The data from which the TELs and PELs for freshwater and marine sediments were derived are primarily from field studies representing a variety of geographic locations, mixtures of chemicals, a wide range of sediment types and characteristics, organisms at different life stages, and many biological endpoints. The heterogeneous physical and chemical factors to which organisms are exposed may influence the occurrence of biological responses associated with chromium. This influence is implicitly incorporated in the calculation of the TELs and PELs because the data used in their derivation represent such a range of conditions. Likewise, utilization of information on a wide range of sediment-resident species, on a variety of life stages, and on several experimental endpoints ensures that the TELs and PELs adequately reflect the range of biological responses that could be observed in association with chromium concentrations in sediments. Incorporation of data from a range of geographic locations increases the likelihood that TEL and PEL values will be broadly applicable in assessing sediment quality in Canada.

The Spiked-Sediment Toxicity Test (SSTT) Approach

The SSTT approach, which involves an independent evaluation of information from spiked-sediment toxicity tests (some of which are included in the guideline derivation tables described above), is a complementary procedure to the modified NSTP approach for estimating the concentration of a chemical in sediments below which adverse effects are not expected to occur (CCME 1995). In contrast to the modified NSTP approach, which is used to derive TELs and PELs using information on *associations* between chemical concentrations in sediments and effects on sediment-associated organisms, the SSTT approach is used to derive SSTT values using data from controlled laboratory tests in which organisms are exposed to sediments that have been spiked with known concentrations of a chemical. Such studies provide quantifiable cause-and-effect relationships between the concentration of a chemical in sediments and the observed biological

response (e.g., survival, reproductive success, growth). Spiked-sediment toxicity tests are generally used to evaluate the toxic effect of a single chemical, or specific mixture of chemicals, to exposed organisms. They may also be used to determine the extent to which environmental conditions modify the bioavailability of a chemical, and ultimately the response of organisms exposed to the spiked-sediments.

Minimum toxicological data requirements have been set for the SSTT approach to ensure that SSTT values derived using spiked-sediment toxicity information provide adequate protection to aquatic organisms (CCME 1995). Insufficient data currently exist to derive SSTT values for chromium in either freshwater or marine sediments. The limited number of spiked sediment toxicity tests that meet CCME (1995) data quality guidelines are discussed in Chapter 3.

CHAPTER 5. RECOMMENDED CANADIAN SEDIMENT QUALITY GUIDELINES

Freshwater Sediments

Sufficient toxicological data exist for freshwater sediments to use the modified NSTP approach to develop a freshwater TEL and PEL. However, insufficient toxicological data exist to calculate a freshwater SSTT value using the SSTT approach. According to the formal protocol (CCME 1995), a TEL calculated using the modified NSTP approach can be recommended as an interim sediment quality guideline. An evaluation of the incidence of effects observed below the freshwater TEL of 37.3 $mg \cdot kg^{-1}$ indicates that it adequately defines the concentration of sediment-associated chromium below which adverse effects rarely occur. In addition, spiked sediment toxicity tests conducted to date have not demonstrated adverse biological effects of chromium below this concentration (Appendix IIIa). Because the information upon which the TEL is based is representative of sediments with a broad range of characteristics, concentrations below the TEL are unlikely to be associated with adverse biological effects under a wide variety of conditions. Therefore, a high degree of certainty exists in the TEL as a sediment-associated chromium concentration that would protect freshwater aquatic life.

The recommended interim freshwater sediment quality guideline for chromium is, therefore, 37.3 $mg \cdot kg^{-1}$ dry weight.

This interim sediment quality guideline is recommended for total (or more accurately, "near total"; see Chapter 1) chromium concentrations in freshwater surficial sediments (i.e., top 0 to 5 cm centimetres). Because Canadian sediment quality guidelines are intended to be used for evaluating the potential for biological effects, this guideline refers to concentrations of chromium as quantified by digestion with a strong acid (e.g., with aqua regia, HNO_3, HCl, etc.) followed by detection with a standard analytical device for trace metals. Such methods are comparable to those used in studies upon which this interim guideline is based (Appendix IIa; CCME 1995; Environment Canada 1996).

In order to derive a full freshwater sediment quality guideline for chromium, at least four independent spiked-sediment toxicity studies on two or more North American freshwater sediment-resident invertebrates are required (CCME 1995). One of these studies must be on a benthic crustacean species and one must be on a benthic arthropod species other than a crustacean (CCME 1995). In addition, at least two of these studies must be partial or full life-cycle tests that consider ecologically relevant endpoints (e.g., growth, reproduction, developmental effects).

Because concerns exist regarding sediment spiking methods (Environment Canada 1995a), advances in methods standardization and/or the development of methods performance criteria is also required before such data could be used as the basis for the recommendation of a full freshwater sediment quality guideline for chromium. Existing evidence is sufficient to determine that the freshwater TEL is applicable to a wide range of sediment types. Although factors such as sediment TOC, metal oxide and AVS concentration are believed to influence the bioavailability (and therefore toxicity) of chromium, no convincing evidence (i.e., from spiked-sediment toxicity tests or field studies) exists to suggest that these influences are quantifiable and predictable. Therefore for the purposes of national sediment quality guideline development in Canada, no justification therefore exists to modify, or adjust, the TEL to account for variation in these sediment characteristics. Nonetheless, additional studies should examine relationships between the bioavailability and toxicity of sediment-associated chromium and characteristics of the sediment and overlying water column at specific sites. As is discussed further in Chapter 6, results of such studies would assist in refining the means by which such factors are considered in the site-specific implementation of freshwater sediment quality guidelines for chromium.

The freshwater PEL of 90.0 $mg \cdot kg^{-1}$ dry weight, is recommended as an additional sediment quality assessment tool. Although this value does not technically meet its narrative objective (i.e., > 50% effects above the PEL), it is anticipated that the PEL for chromium in freshwater sediments will be useful in identifying sediments in which adverse biological effects are more likely to occur (i.e., sediments with chromium concentrations above the PEL). As is discussed further in Chapter 6, such sediments would likely be considered a higher priority for sediment quality management action.

The recommended interim sediment quality guideline and PEL for freshwater sediments are within the range of philosophically-similar sediment quality guidelines recommended by other jurisdictions (Appendix Ia). For freshwater sediments in Ontario, Persaud et al. (1992), have used the screening level concentration approach to establish a lowest observable effect level (i.e., LOEL which is comparable, philosophically, to the TEL) of 26 $mg \cdot kg^{-1}$ (versus 37.3 $mg \cdot kg^{-1}$ for Canada) while the severe effect level (comparable, philosophically, to the PEL) has been set at 110 $mg \cdot kg^{-1}$ (versus 90 $mg \cdot kg^{-1}$ nationally). Similarly, in the St. Lawrence River, the screening effect level approach was used to establish a minimal effect level of 55 $mg \cdot kg^{-1}$ and a toxic effect level of 110 $mg \cdot kg^{-1}$, whereas background concentrations were used to establish a no-effect level of 55 $mg \cdot kg^{-1}$ (Environnement Canada et Ministère de l'Environnement du Québec (EC/MENVIQ) 1992). Likewise, background levels at the *Ambrosia* horizon in Lake Michigan sediments, were

used by the International Joint Commission (IJC 1988) to recommend a threshold chromium concentration of 37.1 $mg \cdot kg^{-1}$. In other Great Lakes Harbours, the US EPA (1977) used background concentrations to establish that chromium concentrations less than 25 $mg \cdot kg^{-1}$ constituted non-polluted sediments.

Marine and Estuarine Sediments

Sufficient toxicological data exist for marine sediments to use the modified NSTP approach to develop a marine (including estuarine) TEL and PEL. However, insufficient toxicological data exist to calculate a marine SSTT value using the SSTT approach. According to the formal protocol (CCME 1995), a TEL calculated using the modified NSTP approach can be recommended as an interim sediment quality guideline. An evaluation of the incidence of effects below the marine TEL of 52.3 $mg \cdot kg^{-1}$ indicates that it adequately defines the concentration of sediment-associated chromium below which adverse effects rarely occur. Data from spiked-sediment toxicity tests further confirm that chromium concentrations in this concentration range would not be expected to be associated with adverse biological effects (Appendix IIIb). Because the information upon which the TEL is based is representative of sediments with a broad range of characteristics, concentrations below the TEL are unlikely to be associated with adverse biological effects under a wide variety of conditions. Therefore, a high degree of certainty exists in the TEL as a sediment-associated chromium concentration that would protect marine and estuarine aquatic life.

The recommended interim marine sediment quality guideline for chromium is, therefore, 52.3 $mg \cdot kg^{-1}$ dry weight.

This interim sediment quality guideline is recommended for total (or more accurately "near total"); see Chapter 1) chromium concentrations in marine and estuarine surficial sediments (i.e., top 0 to 5 cm). Similar to the freshwater guideline, it refers to "near total" concentrations of chromium as quantified by digestion with a strong acid (e.g., with aqua regia, HNO_3, HCl, etc.) followed by detection with a standard analytical device for trace metals.

In order to derive a full marine sediment quality guideline for chromium, at least four independent spiked-sediment toxicity studies on two or more North American marine sediment-resident invertebrates are required (CCME 1995). One of these studies must be on a benthic amphipod species (CCME 1995). In addition, at least two of these studies must be partial or full life-cycle

tests that consider ecologically relevant endpoints (e.g., growth, reproduction, developmental effects). Although factors such as sediment TOC content and AVS concentration are believed to influence the bioavailability (and therefore toxicity) of chromium, no convincing evidence (i.e., from spiked-sediment toxicity tests or field studies) exists to suggest that these influences are quantifiable and predictable. Therefore, for the purposes of national sediment quality guideline development in Canada, no justification exists to modify, or adjust, the TEL to account for variation in these characteristics. As is discussed further in Chapter 6, however, these characteristics should be considered along with other site-specific information in the implementation of Canadian sediment quality guidelines for chromium. Because the information upon which the TEL is based (i.e., data in the marine guideline derivation table) is representative of sediments with a broad range of characteristics, concentrations below the TEL are unlikely to be associated with adverse biological effects under a wide variety of conditions. Therefore, a high degree of certainty exists in the TEL as a sediment-associated chromium concentration that would protect marine and estuarine aquatic life.

The marine PEL of 160 $mg \cdot kg^{-1}$ is recommended as an additional sediment quality assessment tool. The high incidence of adverse biological effects (53%) observed in association with chromium concentrations (in the marine guideline derivation table) above the PEL suggests that this value will be useful in identifying sediments in which adverse biological effects are likely to occur frequently.

The recommended interim sediment quality guideline and PEL for marine sediments are within the range of philosophically-similar sediment quality guidelines recommended by other jurisdictions (Appendix Ib). For marine sediments, the marine sediment quality objective for Burrard Inlet, British Columbia, determined from background concentrations and apparent effects threshold data for Puget Sound, has been set at 60 $mg \cdot kg^{-1}$ (Swain and Nijman 1991). This is comparable to the marine TEL value of 52.3 $mg \cdot kg^{-1}$. In the United States, the apparent effects threshold approach (AETA) has been used in several jurisdictions. Marine AETA concentrations of 27 to 270 $mg \cdot kg^{-1}$ were developed in Puget Sound (Bellar et al. 1986; Barrick et al. 1988), >240 to >820 $mg \cdot kg^{-1}$ along the coast of California (Becker et al. 1989, 1990) and 280 or 370 $mg \cdot kg^{-1}$ in San Francisco Bay (Long and Morgan 1990). Alden and Rule (1992) calculated an AETA value of 68 $mg \cdot kg^{-1}$ for a data set which included sediment chromium and toxicity measurements in the port of Hampton Roads, Virginia. The range in these AETA values results from the different organisms (e.g., *Rhepoxynius, Crassostrea*) and bioassay procedures employed, as well as from the different localities/databases upon which the AETA values were developed.

In Europe, the Netherlands have used the equilibrium partitioning approach (EqPA), to initiate target values in sediments of 100 $mg \cdot kg^{-1}$ (Appendix Ia,b) with an upper limit being proposed at 380 $mg \cdot kg^{-1}$ (Netherlands Ministry of Housing, Spatial Planning and the Environment, 1994). In Norway, sediment background levels have been used to develop four classes of contamination. The proposed boundary between Class II, representing moderate pollution/contamination and Class III, representing marked pollution/contamination is 300 $mg \cdot kg^{-1}$ or 4.3 times the background level of 70 $mg \cdot kg^{-1}$ measured in aerobic Norwegian fjord sediments. The proposed boundary between Class III and Class IV, representing strong pollution/contamination is 1 400 $mg \cdot kg^{-1}$ or 20 times the background level (NIVA 1992).

The general applicability of the modified NSTP (National Status and Trends Program) approach using the weight of evidence of available toxicological information is evidenced by the fact that other jurisdictions have already adopted or are considering adopting a similar strategy. For example, the Agence de l'Eau Rhin-Meuse is contemplating using a combination of WEA and EqPA (Marc Babut, Agence de l'Eau Rhin-Meuse, France, 1995, pers. com.) as is the Joint Monitoring Group of the Oslo and Paris Commissions (Rachel Fleming, WRc, UK., 1995, pers. com.). MacDonald (1996) utilized the identical NSTP database to develop sediment qualtiy guidelines for Florida coastal waters, which are the same as the marine values of 52.3 $mg \cdot kg^{-1}$ (TEL) and 160 $mg \cdot kg^{-1}$ (PEL) established in the previous section. Long and Morgan (1990) compiled an 'effects' database containing both marine and estuarine sediment data collected as part of the NSTP and calculated an effects range-low value (comparable to TEL) of 80 $mg \cdot kg^{-1}$ and an effects range-median value (comparable to PEL) of 145 $mg \cdot kg^{-1}$. More recently Long et al. (1995) applied only 'effects' data to ascertain effects range-low and effects range-median values of 81 $mg \cdot kg^{-1}$ and 370 $mg \cdot kg^{1}$, respectively. Since agreement within a factor of three or less among guidelines developed with different methods has been recommended as an indication of good precision (Lorenzato et al. 1991), it would appear that the interim marine and freshwater sediment qualtiy guidelines developed in the previous section are comparable to those derived for other locations in the Northern hemisphere.

The information upon which the sediment qualtiy guidelines are based (i.e., data in the freshwater and marine guideline derivation tables) is representative of sediments with a broad range of characteristics and mixtures of chemicals, which means that concentrations of chromium below the sediment qualtiy guidelines are unlikely to be associated with adverse biological effects under a wide variety of conditions. Therefore, a high degree of certainty exists in the sediment qualtiy guidelines as chromium concentrations in sediment that will protect freshwater and marine aquatic

life. Because the sediment qualtiy guidelines are applicable to a wide range of sediment types, it is not necessary to adjust these values for variation in sediment characteristics. Nonetheless, additional studies should examine relationships between the bioavailability and toxicity of chromium in sediments and characteristics of the sediment and overlying water column at specific sites. As is discussed further in Chapter 6, results of such studies would assist in refining the means by which such factors are considered in the site-specific implementation of freshwater sediment qualtiy guidelines for chromium.

The recommended interim Canadian freshwater and marine sediment quality guidelines and PELs for chromium adequately reflect our current understanding of the behaviour and toxicity of sediment-associated chromium. The means by which they can be used together, and in conjunction with other relevant information, to prioritize and focus sediment quality assessments are discussed further in the following chapter.

CHAPTER 6. IMPLEMENTATION GUIDANCE

The Canadian interim sediment quality guidelines (ISQGs) and PELs for chromium developed in this report are scientific benchmarks (i.e., reference points) that can be used, along with other relevant information, as the basis for evaluating, protecting and enhancing sediment quality. Although these sediment quality guidelines are considered interim at this time, they should not be used differently than they would be if they were full sediment quality guidelines. However, during their application it should be recognized that these values reflect associative information only (i.e., no reliable spiked-sediment toxicity data exist to evaluate cause-and-effect relationships between sediment-associated chromium concentrations and the occurrence of adverse biological effects). It should also be noted that the interim freshwater sediment quality guideline is derived largely from data collected in the Great Lakes. Generic guidance regarding the implementation of Canadian sediment quality guidelines is provided in CCME (1995) and discussed briefly below.

Sediment quality guidelines, like other environmental quality guidelines, have a broad range of potential applications. As is described by the CCME (1995) and Gaudet et al. (1995), they can serve as goals or interim targets for national and regional toxic substance management programs, as benchmarks or targets in the assessment and remediation of contaminated sites, or as the basis for the development of site-specific objectives. These values can also be used as environmental benchmarks for international discussions on emission reductions, as environmental guidelines on trade agreements, in reports on the state of regional or national sediment quality, in the assessment of the efficacy of environmental regulations, in evaluations of potential impacts of developmental activities, and in the design, implementation and evaluation of sediment quality monitoring programs. Despite the variety of potential uses of sediment quality guidelines, they are likely to be routinely applied as screening tools in the site-specific assessment of the potential risk of exposure to sediment-associated substances and in formulating initial management decisions (e.g., acceptability for open-water disposal, required remediation or further investigation, prioritization of sites).

The following discussion reviews the existing framework for assessing sediment quality (CCME 1995) as exemplified by the use of Canadian ISQGs for chromium. In the application of this framework, it is important to recognize that these guidelines for chromium are intended to be used in conjunction with other supporting information. Such information includes Canadian sediment quality guidelines and PELs (or other relevant sediment quality assessment values) for other chemicals, including trace metals and organics (e.g., CCME 1996a,b). In addition, supporting information such as site-specific background concentrations of chromium and other naturally-

occurring substances, biological assessments, and environmental quality guidelines for other media (i.e., water, aquatic tissues) is also important. It should also be noted that the ISQGs and PELs for chromium are developed using scientific information only (i.e., field studies, as summarized in Appendix IIa,b). Socioeconomic (e.g., cost) or technological (e.g., remedial technology) factors that may influence their application are not considered in the development process. However, such factors may play a varying role in the application of sediment quality guidelines (and/or in the development of site-specific sediment quality objectives) within the decision-making framework of different jurisdictions and programs (Gaudet et al. 1995).

The ISQGs for chromium define the concentrations below which adverse biological effects are not expected to occur in Canadian sediments. In conjunction with the PELs, these values can be used to define three ranges in chromium concentrations; those that are rarely ($\leq$ ISQGs), occasionally (between the ISQGs and the PELs) and frequently ($\geq$ PELs) associated with adverse biological effects (CCME 1995; Gaudet et al. 1995). Evaluation of sediment chromium concentrations with reference to these ranges (and concurrent evaluations of concentrations of other chemicals in sediments relative to their guidelines and PELs) can help to focus sediment quality assessment efforts and prioritize sites for further investigation or management action.

Sediments in which concentrations of all measured chemicals are less than or equal to their respective national sediment quality guidelines would generally be considered to be of acceptable quality for the protection of aquatic life. In general, further investigations of these sediments would be considered of relatively low priority unless scientific information suggesting otherwise (e.g., the presence of high levels of previously unmeasured chemicals) were advanced. Sediments with measured concentrations of one or more chemicals between the sediment quality guideline and the PEL would be considered to represent potential hazards to sediment-associated organisms. The actual hazard to exposed organisms would be dependent on a number of site-specific factors such as local background concentrations (in the case of naturally-occurring substances like chromium) and chemical and geochemical parameters that affect chromium bioavailability, particularly chromium speciation and the factors that affect it (e.g., redox, pH, organic matter, metal oxides, see Chapter 3). Sediments with concentrations of one or more substances equal to or greater than their respective PELs would be considered to have a high probability of being associated with adverse biological effects and would therefore be the highest priority for further investigation or sediment quality management action.

The evaluation of the origin (i.e., natural versus anthropogenic) of sediment-associated chromium

and other naturally-occurring substances is an important component of the above framework. Although background concentrations of such substances may have an adverse effect on certain organisms, realistically, most management actions would focus on concentrations that are primarily influenced by human activities. As was described in Chapter 2, data summarized by the NGR program (Table 3) indicate that naturally-occurring chromium concentrations in individual freshwater sediment samples vary from well below the ISQG for chromium to well above the freshwater PEL. When compared with chromium concentrations in the combined lake and stream sediment database of the NGR program, the freshwater ISQG and PEL for chromium fall at the 38.6th and 83rd percentile, respectively (n = 51 3111), of background chromium concentrations (R. Garrett 1997, Geological Survey of Canada, Ottawa, pers. com.). Concentrations measured in freshwater surficial sediments, which may reflect ambient or contaminated conditions, also vary substantially and in some cases overlap with estimated background concentrations for a given area. Naturally-occurring chromium concentrations in marine (and estuarine) sediments (Chapter 2) range from well below the marine ISQG for chromium to just above the marine PEL. Concentrations in Canadian marine surficial sediments also vary considerably and in some cases overlap with estimated background concentrations for a given area. Because of this variation, site-specific information regarding the contribution of natural processes versus human activities to measured chromium concentrations in Canadian freshwater and marine sediments should be considered in decision-making processes. Site-specific information on background concentrations of chromium in sediments can be used, along with surficial sediment chemistry data, ISQGs, and the PELs to further focus sediment assessment activities. Therefore, the effective implementation of ISQGs and PELs for chromium requires an understanding of the degree to which natural and anthropogenic sources contribute to observed concentrations in sediments at specific sites.

In the above framework, scientific and management considerations would continue to focus the decision-making process on sediments in which the concentrations of one or more substances exceed their respective sediment quality guidelines or PELs, and in which the concentrations are the result of human activities rather than natural processes. Additional scientific studies would focus on determining whether chromium and other substances were bioavailable and whether effects on resident organisms were occurring or were likely to occur from sediment-associated chromium. The effect of physicochemical and geochemical factors upon chromium bioavailability (e.g., redox, pH, organic matter, metal oxides; see Chapter 3) should be considered. Although evidence suggests that geochemical characterization of sediments and selective extraction procedures may be used to estimate the relative bioavailability of chromium in sediments (see Chapter 3), at present substantial uncertainty exists in the prediction of metal bioavailability

(Tessier and Campbell 1987; Luoma 1989; Burton 1995; Chapman 1995). This is particularly true of chromium due to its speciation and the different aqueous behaviours of Cr (III) and Cr(VI) (see Chapter 3). However, biological assessments, including laboratory bioassays, *in situ* investigations of benthic invertebrate community structure, and the examination of concentrations of substances in tissues of benthic organisms and higher trophic levels can be used to estimate bioavailability and determine the extent and severity of effects (CCME 1997). Concurrent measurement of additional factors such as redox, pH, metal oxides and TOC concentrations and selectively-extracted metal fractions may assist in the interpretation of the results of such investigations.

The need to undertake additional biological assessments should be prioritized in a tiered manner. For example, in the routine screening of dredged sediment for ocean disposal, Environment Canada is considering a cost-effective tiered testing scheme in which sediments with concentrations of one or more chemicals that exceed national sediment quality guidelines (for a specified list of substances, including chromium) would be subject to additional testing with a suite of bioassays before a decision is made regarding their acceptability for disposal (Environment Canada 1995b). In other cases (e.g., when the goal of a study is to focus remedial action at a contaminated site), more detailed studies or ecological risk assessments, may be desirable (e.g., Canfield et al. 1994; Pascoe et al. 1994; Reynoldson et al. 1995; CCME 1996c). This process, in which sediment quality guidelines and PELs are used to prioritize and focus assessment activities, provides a cost-effective alternative to sole reliance on site-specific investigations.

Some management decisions may have to be made on an ecosystem basis, taking into account the processes that may be specific to one ecosystem but not another (Luoma and Carter 1993). Given the wide range of ecosystems and sediments for which the effects of sediment-associated chromium were assessed, the interim sediment qualtiy guidelines and PELs are applicable to all systems, unless unique features with implications upon bioavailability and effects are identified. In identifying unique systems, chromium concentrations and unusual physical, chemical, or geochemical features that may enhance or mitigate chromium bioavailability (e.g., redox status), should be considered with respect to the biology of resident organisms. The relationship of organisms to the sediment (i.e., whether the organism lives in it, consumes it, consumes organisms that live in it, and any combination of these) should be considered with respect to addressing factors that could modify the application of sediment quality guidelines. The extent to which organisms take up metals from particulate or dissolved sources is useful information that can provide guidance for the use of sediment or water quality guidelines. Such information is seldom available for many benthic organisms. Further, organisms are never exposed to chromium as a

single entity. Therefore, joint actions of chemicals should be taken into consideration, as in the modified NSTP approach.

Sediment quality guidelines for the protection of aquatic life should be used in conjunction with other relevant information in a holistic manner (e.g., Pascoe and DalSoglio 1994; CCME 1996c). They should also be used along with tissue quality guidelines and water quality guidelines (for the protection of aquatic life and other resource uses), which account for the potential for adverse biological effects from other exposure routes. Because ISQGs and PELs are derived from data regarding the toxicity of sediment-associated substances to benthic organisms, they do not specifically account for the potential for adverse biological effects on higher trophic levels that may result from dietary exposure. Canadian tissue quality guidelines are being developed for substances that are expected to pose a threat to consumers of aquatic life (S.L. Walker, Environment Canada Ottawa, 1996, pers. com.). Such guidelines, which specify maximum recommended concentrations of substances in the tissues of aquatic organisms, should be used in conjunction with sediment quality guidelines to evaluate the potential for adverse biological effects on other components (e.g., wildlife) of aquatic ecosystems. Because chromium does not biomagnify, (see Chapter 3), the trophic transfer of sediment-associated chromium would not be expected to result in adverse effects on aquatic organisms. Similarly, the ISQGs and PELs for chromium recommended in this report should also protect sensitive aquatic species that are exposed to such substances in the water column. However, potential effects on such organisms should also be evaluated relative to Canadian water quality guidelines for chromium or other relevant water quality guidelines for other substances. Water quality guidelines for chromium have been recommended (CCREM 1987) and should be used along with the sediment quality guidelines and PELs recommended in this report to evaluate the potential for adverse biological effects on aquatic ecosystems.

It is widely recognized that no single sediment quality assessment tool can always predict whether adverse biological effects will occur as a result of sediment exposure. Rather, the appropriate use of different tools provides the most useful information (Luoma and Carter 1993; Chapman 1995). Sediment quality guidelines are only one of the many scientific tools available to assist in the protection and management of sediment quality. The use of sediment quality guidelines in the exclusion of other information (such as background concentrations of naturally occurring substances, other assessment values such as the PEL, biological assessments, or guidelines for other media) can lead to erroneous conclusions or predictions about sediment quality. Decisions are more defensible if they acknowledge scientific uncertainties and allow for management

modifications as scientific knowledge improves (Luoma and Carter 1993). In the framework discussed above, sediment quality guidelines and PELs provide nationally-consistent benchmarks with which to evaluate the ecological significance of sediment chemical concentrations and determine the relative priority of sediment quality concerns. Canadian ISQGs for chromium should be used along with all other relevant information in making practical and informed decisions regarding sediment quality. These considerations are equally important whether the focus is to maintain, protect, or improve sediment quality conditions at a particular site in Canada.

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Appendix 1a: Freshwater sediment quality guidelines for chromium from other jurisdictions

Jurisdiction	Approach	Guideline $mg \cdot kg^{-1}$ dw	Level of Protection	Reference
Ontario	SLCA	26	Lowest Effect Level	Persaud et al. 1992
USEPA Region V	SBA	< 25	Non Polluted	USEPA 1977
Lake Michigan	SBA	37.1	Threshold Level	IJC 1988
St. Lawrence River	SBA	55	No Effect Threshold	EC/MENVIQ 1992
St. Lawrence River	SLCA	55	Minimal Effect Threshold	EC/MENVIQ 1992
Norway	SBA	< 70	Class I	NIVA 1992
USEPA Region V	SBA	25 - 75	Moderately Polluted	USEPA 1977
USEPA Region V	SBA	> 75	Heavily Polluted	USEPA 1977
Netherlands	EqPA	100	Target value; multifunctional sediment quality for the long-term; risk of adverse effects on man and the ecosystem are negligible	Ministry of Housing, Physical Planning and the Environment 1994
Ontario	SLCA	110	Severe Effect Level	Persaud et al. 1992
St. Lawrence River	SLCA	110	Toxic Effect Level	EC/MENVIQ 1992
Norway	SBA	70 - 300	Class II	NIVA 1992
Netherlands	EqPA	380	Limit value; quality level to be realized by the year 2000; determined by an ecotoxicological risk evaluation	Ministry of Housing, Physical Planning and the Environment 1994
Netherlands	EqPA	380	Intervention value; level above which there is serious pollution; urgency of cleanup measures must be determined	Ministry of Housing, Physical Planning and the Environment 1994
Norway	SBA	300 - 1400	Class III	NIVA 1992
Norway	SBA	> 1400	Class IV	NIVA 1992

Appendix 1b: Marine sediment quality guidelines for chromium from other jurisdictions

Jurisdiction	Approach	Guideline mg·kg^{-1} dw	Level of Protection	Reference
Puget Sound, WA	AETA	27	15-min Microtox	Bellar et al. 1986
Puget Sound, WA	AETA	> 37	48-h Crassostrea	Bellar et al. 1986
OSPAR	EqPA & WEA	5 - 50	Provisional Value; concentration below which no harm to the marine environment is expected	OSPAR 1994
Florida	WEA	52.3	Threshold Effect Level	MacDonald 1996
Puget Sound, WA	AETA	59	Benthic Community	Bellar et al. 1986
Burrard Inlet, B.C.	SBA & AETA	60	Protection of aquatic biota	Swain and Nijman 1991
Virginia	AETA	68		Alden and Rule 1992
Norway	SBA	< 70	Class I	NIVA 1992
Netherlands	EqPA	100	Target value; multifunctional sediment quality for the long-term; risk of adverse effects on man and the ecosystem are negligible	Ministry of Housing, Physical Planning and the Environment 1994
Puget Sound, WA	AETA	> 130	10-d Rhepoxynius	Bellar et al. 1986
Florida	WEA	160	Probable Effect Level	MacDonald 1996
Norway	SBA	70 - 300	Class II	NIVA 1992
Northern California	AETA	> 240	Benthic community, 10-d Rhepoxynius	Becker et al. 1989, 1990
California	AETA	> 240	48-h Mytilus	Becker et al. 1989, 1990
Puget Sound, WA	AETA	260	Benthic community	Barrick et al. 1988
Puget Sound, WA	AETA	270	10-d Rhepoxynius	Barrick et al. 1988
San Francisco Bay, CA	AETA	280	48-h bivalve larvae	as cited by Long and Morgan 1980
Southern California	AETA	310	Benthic Community	Becker et al. 1989, 1990
California	AETA	310	Benthic Community	Becker et al. 1989, 1990
San Francisco Bay, CA	AETA	370	10-d Rhepoxynius	as cited by Long and Morgan 1980

Appendix 1b (cont): Marine sediment quality guidelines for chromium from other jurisdictions

Jurisdiction	Approach	Guideline $mg \cdot kg^{-1}$ dw	Level of Protection	Reference
Netherlands	EqPA	380	Limit value; quality level to be realized by the year 2000; determined by an ecotoxicological risk evaluation	Ministry of Housing, Physical Planning and the Environment 1994
Netherlands	EqPA	380	Intervention value; level above which there is serious pollution; urgency of cleanup measures must be determined	Ministry of Housing, Physical Planning and the Environment 1994
Norway	SBA	300 -1400	Class III	NIVA 1992
Southern California	AETA	> 820	10-d Rhepoxynius	Becker et al. 1989, 1990
California	AETA	> 820	10-d Rhepoxynius	Becker et al. 1989, 1990
Norway	SBA	> 1400	Class IV	NIVA 1992

Appendix Ia. Summary of the available biological effects and related physicochemical data for

Chromium ($mg \cdot kg^{-1}$)		Hit	Analysis Type	Test Type	Endpoint Measured	Species	Life Stage
5.5		NC	COA		Low diversity (1.34; Shannon Weaver Index)	Benthic species	
6		NE	COA		High abundance (169 N/sq.m.)	Gastropoda	
6.25		*	EqPA		Chronic Marine EqP Threshold	Aquatic biota	
7.4	±2.97	NE	COA		High diversity (3.8±0.0212; Shannon Weaver Index)	Benthic species	
8.2	±2.4	NC	COA		Low abundance (1036±696 N/sq.m.)	Benthic invertebrates	
8.8	±1.56	NE	COA		High abundance (523±26.2 N/sq.m.)	Gastropoda	
8.93	±2.72	NE	COA		High taxa (8±1.73 S)	Ephemeroptera, Plecoptera, Tricoptera	
8.93	±2.72	NE	COA		High taxa (5±1 S)	Ephemeroptera, Plecoptera, Tricoptera	
9.05	±0.636	NE	COA		High abundance (1118±803 N/sq.m.)	Amphipoda	
9.05	±0.636	NE	COA		High abundance (437±227 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
9.8	±1.7	NE	COA		High abundance (4034±1459 N/sq.m.)	Chironomidae	
9.8	±1.7	NE	COA		High abundance (16306±859 N/sq.m.)	Benthic invertebrates	
9.87	±2.15	NC	COA		Moderate abundance (841±258 N/sq.m.)	Amphipoda	
9.91	±2.66	NC	COA	10-d	Toxic (44±15.2% mortality)	*Chironomus tentans* (midge)	LAR
10		NE	COA		High species richness (2.37; SRUs)	Benthic species	
10		NE	COA		High abundance (20308 N/sq.m.)	Benthic invertebrates	
10.2	±1.84	NE	COA		High abundance (812±183 N/sq.m.)	Chironomidae	
10.2	±1.84	NE	COA		High species richness (2.13±0.219; SRUs)	Benthic species	
10.3	±2.08	NC	COA	48-h	Significantly toxic (24.2±2.89% mortality)	*Daphnia pulex* (water flea)	
10.4	±5.18	NC	COA		Low diversity (1.32±0.318; Brillouin's Diversity Index)	Benthic species	
10.8	±2.96	NC	COA	48-h	Significantly toxic (27% mortality)	*Daphnia magna* (water flea)	NEO
11		NE	COA		High abundance (4374 N/sq.m.)	Amphipoda	
11.3	±3.57	NC	COA	10-d	Significantly toxic (36% mortaltiy)	*Hyalella azteca* (amphipod)	<6 mm
11.4	±3.69	SG	COA		Low abundance (567±453 N/sq.m.)	Benthic invertebrates	
11.6	±3.2	NC	COA		Low species (6.88±3.72 S/sq.m.)	Benthic species	
11.7	±3.01	NE	COA		High biotic integrity (45.8±4.66; AIBI)	Benthic invertebrates	
11.9	±2.28	NE	COA		High diversity (2.88±0.424; Brillouin's Diversity Index)	Benthic species	
12		NE	COA		High abundance (774±53.7 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
12.2	±3.04	NE	COA	10-d	Not toxic (5.8±5.14% mortality)	*Hyalella azteca* (amphipod)	<6 mm
12.6	±2.56	NE	COA	48-h	Not significantly toxic (5.5±5.8% mortality)	*Chironomus tentans* (midge)	LAR
12.6	±5.3	NE	COA	96-h	Not toxic (2.5±3.1% mortality)	*Hyalella azteca* (amphipod)	
12.6	±9.25	SG	COA		Moderate abundance (1446±323 N/sq.m.)	Chironomidae	
13	±1.73	NE	COA		High abundance (746±133 N/sq.m.)	Gastropoda	
13.2	±2.37	NE	COA	48-h	Not significantly toxic (6±5.12% mortality)	*Daphnia magna* (water flea)	NEO
13.3	±4.82	NE	COA		High species (24.3±9.84 S/sq.m.)	Benthic species	
13.4	±2.65	NE	COA		High species (22±5.77 S/sq.m.)	Benthic species	
13.5	±2.05	SG	COA		Moderate diversity (2.05±0.041; Brillouin's Diversity Index)	Benthic species	
13.5	±2.12	NE	COA		High abundance (2843±706 N/sq.m.)	Chironomidae	
13.5	±2.12	NE	COA		High abundance (1550±978 N/sq.m.)	Amphipoda	
13.5	±2.12	NE	COA		High abundance (6127±1621 N/sq.m.)	Benthic invertebrates	
13.5	±5.98	NE	COA	96-h	Not significantly toxic (9.69±3.12% mortality)	*Daphnia pulex* (water flea)	
13.7	±1.87	SG	COA		Moderate abundance (5164±1594 N/sq.m.)	Benthic invertebrates	
13.9	±2.14	NE	COA	10-d	Not toxic (17.2±7.56% mortality)	*Chironomus tentans* (midge)	LAR
14	±1.41	SG	COA		Low abundance (4±5.66 N/sq.m.)	Gastropoda	
14.2	±3.54	NE	COA		High diversity (3.18±0.429; Shannon Weaver Index)	Benthic species	
14.2	±3.54	NE	COA		High species richness (3.27±0.509; SRUs)	Benthic species	
14.4	±5	NC	COA		Low abundance (1297±1052 N/sq.m.)	Benthic invertebrates	
14.4	±6.21	NC	COA		Low biotic integrity (32.5±3.42; AIBI)	Benthic invertebrates	
14.6		SG	COA	48-h	Significantly toxic (33% mortality)	*Chironomus tentans* (midge)	LAR
14.6	±3.78	SG	COA		Low abundance (145±145 N/sq.m.)	Chironomidae	
14.6	±3.78	SG	COA		Low abundance (90±109 N/sq.m.)	Amphipoda	
14.6	±3.78	SG	COA		Low abundance (1017±746 N/sq.m.)	Benthic invertebrates	
14.6	±5.05	NC	COA	10-d	Toxic (30±10% mortality)	*Chironomus tentans* (midge)	LAR
14.8	±2	SG	COA		Low biotic integrity (31±9.9; AIBI)	Benthic invertebrates	
14.9	±8.7	NC	COA		Low diversity (1.05±0.205; Shannon Weaver Index)	Benthic species	
14.9	±8.7	NC	COA		Low species richness (0.805±0.163; SRUs)	Benthic species	
15		SG	COA		Low diversity (1.45; Shannon Weaver Index)	Benthic species	
15		SG	COA		Low species richness (1.36; SRUs)	Benthic species	
15.1	±5.7	NE	COA	10-d	Not toxic (8.9±3.33% mortality)	*Chironomus tentans* (midge)	LAR
15.2	±3.49	SG	COA		Low abundance (101±142 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
15.4		NC	COA		Low biotic integrity (20; AIBI)	Benthic invertebrates	
15.4	±5.33	NE	COA		Typical diversity (2.65±0.396; Brillouin's Diversity Index)	Benthic species	
15.7	±5.68	SG	COA		Low species (10.4±2.2 S/sq.m.)	Benthic species	
15.9	±5.2	NE	COA		High species (27.6±5.27 S/sq.m.)	Benthic species	

sediment-associated chromium (mg·kg^{-1} dry weight) in freshwater systems.

TOC (%)	AVS (µmol·g^{-1})	Unionized Ammonia (mg·L^{-1})	Sand (%)	Silt (%)	Clay (%)	Area	Reference
0.25			96	2	1	Humber Bay, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
0.25			91	7	2	Detroit River, MI	Jaagumagi 1988; et al. 1989
1						United States	Bolton et al. 1985
0.715±0.262						St. Mary's River, ON	Jaagumagi 1988; et al. 1989
0.25			97	1.5±0.71	0.5±0.71	Toronto Outer Harbour & Eastern Headland, ON	Jaagumagi 1988; et al. 1989
0.25			94.5±2.12	3.5±0.707	1	Toronto Harbour, ON	Jaagumagi 1988; et al. 1989
0.98±0.19						River Adige, Italy	Duzzin et al. 1988
0.98±0.19						River Adige, Italy	Duzzin et al. 1988
0.55±0.028						St. Mary's River, ON	Jaagumagi 1988; et al. 1989
0.55±0.0283						St. Mary's River, ON	Jaagumagi 1988; et al. 1989
0.41±0.226						St. Mary's River, ON	Jaagumagi 1988; et al. 1989
0.41±0.226						St. Mary's River, ON	Jaagumagi 1988; et al. 1989
0.433±0.318			91.3±5.69	5±2.65	1.33±0.57	Toronto Harbour, ON	Jaagumagi 1988; et al. 1989
0.362±0.27						Trinity River, TX	Dickson et al. 1989
0.25						Toronto Outer Harbour & Eastern Headland, ON	Jaagumagi 1988; et al. 1989
1.07						Trinity River, TX	Dickson et al. 1989
0.388±0.275			91.3±4.65	4.75±2.22	1.25±0.5	Toronto Harbour, ON	Jaagumagi 1988; et al. 1989
0.388±0.275			91.3±4.65	4.75±2.22	1.25±0.5	Toronto Harbour, ON	Jaagumagi 1988; et al. 1989
0.2±0.1						Lower Columbia River, WA	Johnson and Norton 1988
0.35±0.335						Trinity River, TX	Dickson et al. 1989
0.505±0.403						Trinity River, TX	Dickson et al. 1989
0.25			91	4	1	Toronto Harbour, ON	Jaagumagi 1988; et al. 1989
0.32±0.085						Trinity River, TX	Dickson et al. 1989
0.418±0.282						Trinity River, TX	Dickson et al. 1989
0.456±0.249						Trinity River, TX	Dickson et al. 1989
0.523±0.297						Trinity River, TX	Dickson et al. 1989
0.66±0.363						Trinity River, TX	Dickson et al. 1989
0.855±0.148			81±4.24	13.5±3.54	3	St. Clair River, ON	Jaagumagi 1988; et al. 1989
0.548±0.276						Trinity River, TX	Dickson et al. 1989
0.59±0.262						Trinity River, TX	Dickson et al. 1989
0.627±0.837						Lower Columbia River, WA	Johnson and Norton 1988
1.71±1.73			53±32.5	40±31.1	4.5±3.54	St. Mary's River, ON	Jaagumagi 1988; et al. 1989
0.82±0.209			81.7±3.21	13.7±2.08	3	St. Clair River, ON	Jaagumagi 1988; et al. 1989
0.607±0.235						Trinity River, TX	Dickson et al. 1989
0.743±0.384						Trinity River, TX	Dickson et al. 1989
0.73±0.26						Trinity River, TX	Dickson et al. 1989
0.595±0.115						Trinity River, TX	Dickson et al. 1989
0.77±0.269			81±4.24	14±2.83	3±2.83	St. Clair River, ON	Jaagumagi 1988; et al. 1989
0.77±0.269			81±4.24	14±2.83	3±2.83	St. Clair River, ON	Jaagumagi 1988; et al. 1989
0.77±0.269			81±4.24	14±2.83	3±2.83	St. Clair River, ON	Jaagumagi 1988; et al. 1989
0.788±0.943						Lower Columbia River, WA	Johnson and Norton 1988
0.598±0.116						Trinity River, TX	Dickson et al. 1989
0.68±0.205						Trinity River, TX	Dickson et al. 1989
0.625±0.53			88±2.83	6±7.07	1±1.41	St. Clair River, ON	Jaagumagi 1988; et al. 1989
0.743±0.288			80.5±6.69	13.2±2.32	3.83±2.56	St. Clair River, ON	Jaagumagi 1988; et al. 1989
0.743±0.288			80.5±6.69	13.2±2.32	3.83±2.56	St. Clair River, ON	Jaagumagi 1988; et al. 1989
0.531±0.247						Trinity River, TX	Dickson et al. 1989
0.533±0.314						Trinity River, TX	Dickson et al. 1989
0.57						Trinity River, TX	Dickson et al. 1989
0.634±0.362			82.2±8.38	10.4±5.64	3.4±3.36	St. Clair River, ON	Jaagumagi 1988; et al. 1989
0.634±0.362			82.2±8.38	10.4±5.64	3.4±3.36	St. Clair River, ON	Jaagumagi 1988; et al. 1989
0.634±0.362			82.2±8.38	10.4±5.64	3.4±3.36	St. Clair River, ON	Jaagumagi 1988; et al. 1989
0.867±0.325						Trinity River, TX	Dickson et al. 1989
7.67±0.085						Trinity River, TX	Dickson et al. 1989
0.25			96	2	1	Port Weller Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
0.25			96	2	1	Port Weller Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
0.25			90	1		St. Clair River, ON	Jaagumagi 1988; et al. 1989
0.25			90	1		St. Clair River, ON	Jaagumagi 1988; et al. 1989
0.564±0.288						Trinity River, TX	Dickson et al. 1989
0.6±0.356			82.2±8.38	10.6±5.68	3.4±3.36	St. Clair River, ON	Jaagumagi 1988; et al. 1989
0.56						Trinity River, TX	Dickson et al. 1989
0.681±0.311						Trinity River, TX	Dickson et al. 1989
0.611±0.284						Trinity River, TX	Dickson et al. 1989
0.75±0.273						Trinity River, TX	Dickson et al. 1989

Appendix Ia. Summary of the available biological effects and related physicochemical data for

Chromium (mg·kg^{-1})		Hit	Analysis Type	Test Type	Endpoint Measured	Species	Life Stage
16		NE	COA		High abundance (1318 N/sq.m.)	Amphipoda	
16		NE	COA		High abundance (199 N/sq.m.)	Gastropoda	
16.1	±4.93	NE	COA		High biotic integrity (45.7±3.9; AIBI)	Benthic invertebrates	
16.4	±16.4	NE	COA		High abundance (13.7±2.31 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
16.4	±7.15	NE	COA		High abundance (9774±1103 N/sq.m.)	Benthic invertebrates	
16.5	±6.36	SG	COA		Moderate abundance (263±94 N/sq.m.)	Gastropoda	
17		NC	COA		Low diversity (1.32±0.0212; Shannon Weaver Index)	Benthic species	
17	±2.18	NC	COA	10-d	Toxic (22.3±4.03% mortality)	*Hyalella azteca* (amphipod)	<6 mm
17	±3.46	NE	COA		High abundance (3696±837 N/sq.m.)	Amphipoda	
17	±3.46	NC	COA		Moderate abundance (11319±1993 N/sq.m.)	Benthic invertebrates	
17.1	±6.97	NE	COA		High abundance (9751±975 N/sq.m.)	Benthic invertebrates	
17.2	±6.84	NE	COA	48-h	Not significantly toxic (2.2±4.64% mortaltiy)	*Chironomus tentans* (midge)	LAR
17.2	±7.41	NC	COA		Low abundance (0 N/sq.m.)	Gastropoda	
17.2	±7.41	NC	COA		Low abundance (147±135 N/sq.m.)	Benthic invertebrates	
17.3	±7.22	NE	COA	10-d	Not toxic (15±5.22% mortaltiy)	*Chironomus tentans* (midge)	LAR
17.4	±7.55	NE	COA		High biotic integrity (46±5.29; AIBI)	Benthic invertebrates	
17.5	±2.12	NE	COA		High abundance (2605±109 N/sq.m.)	Gastropoda	
17.5	±3.87	NC	COA		Moderate abundance (876±277 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
17.6	±13	NE	COA		High abundance (154±30.2 N/0.00203 sq.m.)	Benthic invertebrates	
17.6	±4.16	SG	COA		Moderate abundance (6218±804 N/sq.m.)	Benthic invertebrates	
17.6	±5.85	NE	COA	48-h	Not significantly toxic (0.7±2.21% mortality)	*Chironomus tentans* (midge)	LAR
17.6	±7.44	NE	COA	48-h	Not significantly toxic (2.22±4.66% mortaltiy)	*Daphnia magna* (water flea)	NEO
17.8	±0.502	NC	COA		Low biotic integrity (35.3±3.06; AIBI)	Benthic invertebrates	
18		NE	COA		High abundance (414 N/sq.m.)	Amphipoda	
18		NE	COA		High abundance (888 N/sq.m.)	Benthic invertebrates	
18	±4.58	SG	COA		Low abundance (108±141 N/sq.m.)	Gastropoda	
18.1	±2.69	SG	COA		Low species (8.5±2.12 S/sq.m.)	Benthic species	
18.1	±2.85	NC	COA		Low abundance (1642±1449 N/sq.m.)	Chironomidae	
18.4	±7.27	NE	COA	10-d	Not significantly toxic (5.5±6.26% mortaltiy)	*Hyalella azteca* (amphipod)	<6 mm
19		NC	COA		Low biotic integrity (7.7; MBI)	MacroInvertebrates	
19		NE	COA		High abundance (483 N/sq.m.)	Amphipoda	
19		NE	COA		High abundance (1930 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
19		NC	COA		Low abundance (0 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
19	±2.45	NC	COA		Low abundance (4337±773 N/sq.m.)	Benthic invertebrates	
19	±4	NC	COA		Low abundance (40±17 N/sq.m.)	Chironomidae	
19.1	±7.66	NE	COA		High biotic integrity (46.2±3.46; AIBI)	Benthic species	
19.2	±2.17	SG	COA		Low abundance (325±239 N/sq.m.)	Amphipoda	
19.3	±0.577	SG	COA		Moderate abundance (1340±484 N/sq.m.)	Gastropoda	
19.3	±6.96	NE	COA	10-d	Not toxic (3.3±4.92% mortality)	*Chironomus tentans* (midge)	LAR
19.5	±0.707	NE	COA		High abundance (2375±542 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
19.5	±0.707	NE	COA		High abundance (5646±3216 N/sq.m.)	Benthic invertebrates	
19.6	±7.23	NE	COA		High diversity (2.95±0.612; Brillouin's Diversity Index)	Benthic species	
20		NE	COA		High species richness (1.97; SRUs)	Benthic species	
20		NE	COA		High abundance (12813 N/sq.m.)	Chironomidae	
20		NE	COA		High abundance (19248 N/sq.m.)	Benthic invertebrates	
20		NE	COA		High abundance (5699 N/sq.m.)	Chironomidae	
20		NE	COA		High abundance (490 N/sq.m.)	Gastropoda	
20		NE	COA		High abundance (80 N/sq.m.)	Amphipoda	
20		SG	COA		Low diversity (1.47; Brillouin's Diversity Index)	Benthic species	
20	±6.51	SG	COA	48-h	Significantly toxic (27% mortaltiy)	*Daphnia magna* (water flea)	NEO
20.6	±0.849	NE	COA		High abundance (16910±2877 N/sq.m.)	Benthic invertebrates	
20.6	±0.849	SG	COA		Low biotic integrity (36; AIBI)	Benthic invertebrates	
20.9	±26.2	NE	COA		High species richness (3.06±0.329; SRUs)	Benthic species	
21.3	±7.9	NE	COA	96-h	Not toxic (12.6±5.4% mortality)	*Hexagenia limbata* (mayfly)	NYM
21.5	±2.12	NE	COA		High abundance (69±32.5 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
22		NE	COA		High abundance (9177 N/sq.m.)	Benthic invertebrates	
22	±13.5	NE	COA		Low species (11.2±1.94 S/0.00203 sq.m.)	Benthic species	
22.4	±12	NC	COA		Low diversity (1.6±0.226; Brillouin's Diversity Index)	Benthic species	
23		NG	COA		Low diversity (1.22; Shannon Weaver Index)	Benthic species	
23		NC	COA		Low species richness (0.89; SRUs)	Benthic species	
23.3	±8.35	SG	COA		Moderate species (18.6±2.3 S/sq.m.)	Benthic species	
23.9	±14	NE	COA	28-d	Not significantly toxic (6.43±8.89% mortality)	*Hyalella azteca* (amphipod)	
24.5	±5.26	NE	COA		High species richness (2.05±0.385; SRUs)	Benthic species	
<25		NE	SQG		Non Polluted - USEPA Region 5 Harbour Classification		

sediment-associated chromium (mg·kg^{-1} dry weight) in freshwater systems.

TOC (%)	AVS (µmol·g^{-1})	Unionized Ammonia (mg·L^{-1})	Sand (%)	Silt (%)	Clay (%)	Area	Reference
0.25			94	4	1	Toronto Outer Harbour	Jaagumagi 1988; et al. 1989
0.25			94	4	1	Toronto Outer Harbour	Jaagumagi 1988; et al. 1989
0.679±0.313						Trinity River, TX	Dickson et al. 1989
1.64±2.13			70±30.4	24.3±25.8	4.33±4.04	Penetang Harbour, Lake Huron, ON	Jaagumagi 1988; et al. 1989
1.03±0.081						Trinity River, TX	Dickson et al. 1989
0.5±0.354			76±11.3	13.5±3.54	6±4.24	St. Clair River, ON	Jaagumagi 1988; et al. 1989
0.25			68.5±10.6	19±5.66	11.5±4.95	Oakville Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
0.443±0.199						Trinity River, TX	Dickson et al. 1989
1.47±0.551			82.7±8.5	14.7±9.5	2	St. Clair River, ON	Jaagumagi 1988; et al. 1989
1.47±0.551			82.7±8.5	14.7±9.5	2	St. Clair River, ON	Jaagumagi 1988; et al. 1989
0.562±0.298						Trinity River, TX	Dickson et al. 1989
0.656±0.369						Trinity River, TX	Dickson et al. 1989
0.25			94.7±2.31	3.67±2.89	1	Port Weller Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
0.25			94.7±2.31	3.67±2.89	1	Port Weller Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
0.618±0.353						Trinity River, TX	Dickson et al. 1989
0.623±0.37						Trinity River, TX	Dickson et al. 1989
1.8±0.424			76±2.83	20±5.66	3.5±2.12	St. Clair River, ON	Jaagumagi 1988; et al. 1989
1.7±0.424			71.3±13.6	24.3±11.6	4.25±2.87	St. Clair River, ON	Jaagumagi 1988; et al. 1989
0.47±0.27	9.07±2.94		32.8±5.56	35.6±13	31.7±9.67	Galveston Bay, TX	Carr 1992
0.838±0.221						Trinity River, TX	Dickson et al. 1989
0.494±0.188						Trinity River, TX	Dickson et al. 1989
0.56±0.206						Trinity River, TX	Dickson et al. 1989
0.51±0.278						Trinity River, TX	Dickson et al. 1989
2			82	9	2	Bay of Quinte, ON	Jaagumagi 1988; et al. 1989
2			82	9	2	Bay of Quinte, ON	Jaagumagi 1988; et al. 1989
1.77±0.493			69±15.7	27±12.5	4±3.46	St. Clair River, ON	Jaagumagi 1988; et al. 1989
0.655±0.445						Trinity River, TX	Dickson et al. 1989
1.71±0.438			72±14.4	24±13.5	3.43±2.3	St. Clair River, ON	Jaagumagi 1988; et al. 1989
0.677±0.358						Trinity River, TX	Dickson et al. 1989
						Kishwaukee River Basin, IL	IEPA 1988a
0.25			83	7	2	Niagara River, NY	Jaagumagi 1988; et al. 1989
0.25			83	7	2	Niagara River, NY	Jaagumagi 1988; et al. 1989
1.6±0.707			82.5±12	14.5±13.4	2	St. Clair River, ON	Jaagumagi 1988; et al. 1989
1.9±0.271			64±13	31±12.3	4.5±2.65	St. Clair River, ON	Jaagumagi 1988; et al. 1989
0.435±0.377			58.3±14.2	28.8±12.3	11.5±3.11	Oakville Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
0.615±0.269						Trinity River, TX	Dickson et al. 1989
2.04±0.391			62.2±11.9	32.2±11	4.2±2.39	St. Clair River, ON	Jaagumagi 1988; et al. 1989
1.9±0.755			66.7±21.1	28±20.1	2.67±0.57	St. Clair River, ON	Jaagumagi 1988; et al. 1989
0.66±0.267						Trinity River, TX	Dickson et al. 1989
2.3±0.424			54.5±0.70	39.5±3.54	3	St. Clair River, ON	Jaagumagi 1988; et al. 1989
3.3±2.12			65±2.83	25±1.41	5	Midland Bay, Lake Huron, ON	Jaagumagi 1988; et al. 1989
0.661±0.26						Trinity River, TX	Dickson et al. 1989
0.25						Humber Bay, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
2.6			55	37	3	St. Clair River, ON	Jaagumagi 1988; et al. 1989
2.6			55	37	3	St. Clair River, ON	Jaagumagi 1988; et al. 1989
4.8			63	26	5	Midland Bay, Lake Huron, ON	Jaagumagi 1988; et al. 1989
4.8			63	26	5	Midland Bay, Lake Huron, ON	Jaagumagi 1988; et al. 1989
3.2			77	17	3	Midland Bay, Lake Huron, ON	Jaagumagi 1988; et al. 1989
0.97						Trinity River, TX	Dickson et al. 1989
1.06±0.7						Trinity River, TX	Dickson et al. 1989
0.505±0.12						Trinity River, TX	Dickson et al. 1989
0.885±0.12						Trinity River, TX	Dickson et al. 1989
2.45±3.57						St. Mary's River, ON	Jaagumagi 1988; et al. 1989
						Duluth-Superior Harbor, MN, WI	Bahnick et al. 1981
3.45±1.91			56.5±9.19	31±7.07	6±1.41	Midland Bay, Lake Huron, ON	Jaagumagi 1988; et al. 1989
1.3			30	52	11	Oakville Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
0.642±0.356	13.4±8.65		24.5±13.8	43.3±16.8	32.2±8.31	Galveston Bay, TX	Carr 1992
0.658±0.122						Trinity River, TX	Dickson et al. 1989
2.1			50	36	7	Midland Bay, Lake Huron, ON	Jaagumagi 1988; et al. 1989
2.1			50	36	7	Midland Bay, Lake Huron, ON	Jaagumagi 1988; et al. 1989
0.572±0.229						Trinity River, TX	Dickson et al. 1989
0.47±0.266			42.7±18.3	23.7±21.6	33.7±8.71	Trinity River, TX	Ingersoll 1992
0.91±0.464			35.5±15.6	45.8±9.46	12.8±1.89	Oakville Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
						United States	USEPA 1977 (As cited in SAIC 199

Appendix Ia. Summary of the available biological effects and related physicochemical data for

Chromium ($mg \cdot kg^{-1}$)		Hit	Analysis Type	Test Type	Endpoint Measured	Species	Life Stage
25		*	SBA		OMOE Open Water Disposal Criteria		
25		NE	COA		High abundance (33 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
25.4	±8.35	SG	COA		Low abundance (2536±1062 N/sq.m.)	Benthic invertebrates	
25.5	±8.43	NE	COA		High species (20.6±4.28 S/sq.m.)	Benthic species	
26		*	SLCA		OMOE Provincial SQGs - Lowest Effect Level		
26	±1.41	NE	COA		High abundance (67±8.49 N/sq.m.)	Amphipoda	
26.1	±10.2	NE	COA		High biotic integrity (45.6±3.32; AIBI)	Benthic invertebrates	
26.2		SG	COA	48-h	Significantly toxic (22% mortality)	*Chironomus tentans* (midge)	LAR
26.2	±9.69	NE	COA	10-d	Not toxic (5.8±7.93% mortality)	*Chironomus tentans* (midge)	LAR
26.5	±14.8	NE	COA		High abundance (2107±1030 N/sq.m.)	Chironomidae	
26.6	±18.8	SG	COA		Low abundance (0 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
26.7	±11.1	SG	COA		Low species (9.14±2.48 S/sq.m.)	Benthic species	
26.7	±19.2	NE	COA		High diversity (3.16±0.355; Shannon Weaver Index)	Benthic species	
26.7	±19.8	NC	COA		Low abundance (44.3±19.2 N/sq.m.)	Chironomidae	
26.7	±19.8	NC	COA		Low abundance (5.75±6.45 N/sq.m.)	Amphipoda	
26.7	±19.8	NC	COA		Low abundance (1±2 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
27.1		SG	COA		Low biotic integrity (38; AIBI)	Benthic invertebrates	
27.3		NC	COA		Low abundance (7839 N/sq.m.)	Benthic invertebrates	
27.3		NE	COA		High abundance (1484 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
27.3	±10.3	SG	COA		Low abundance (3569±2188 N/sq.m.)	Benthic invertebrates	
27.3	±2.25	NC	COA		Moderate abundance (28916±3965 N/sq.m.)	Benthic invertebrates	
27.5		NE	COA		High abundance (1835 N/sq.m.)	Chironomidae	
27.5		NC	COA		Low abundance (2937 N/sq.m.)	Benthic species	
27.6	±0.354	NC	COA		Moderate abundance (1320±107 N/sq.m.)	Chironomidae	
27.6	±0.354	NE	COA		High abundance (71±25.5 N/sq.m.)	Gastropoda	
27.7	±15.5	SG	COA		Low abundance (4±3.27 N/sq.m.)	Amphipoda	
27.8		NE	COA		High abundance (169 N/sq.m.)	Amphipoda	
27.9	±40.7	NE	COA	10-d	Not significantly toxic (8.18±8.74% mortality)	*Chironomus tentans* (midge)	LAR
28	±14.2	SG	COA		Low abundance (1611±794 N/sq.m.)	Benthic invertebrates	
28.3	±1.34	NC	COA	16-22-d	Significantly toxic (33.3% mortality)	*Daphnia magna* (water flea)	<24 h
28.7	±9.1	NE	COA		High diversity (2.75±0.298; Brillouin's Diversity Index)	Benthic species	
28.8	±10.8	NC	COA		Low biotic integrity (37±2.45; AIBI)	Freshwater fish	
29	±14	NE	COA		High abundance (491±221 N/sq.m.)	Benthic Invertebrates	
29	±14	NE	COA		High taxa (13.4±2.88 S)	Benthic species	
29	±14	NE	COA		High diversity (0.81±0.07 Shannon Weaver Index)	Benthic species	
29	±14	NE	COA		High abundance (195±155 N/sq.m.)	Emphemeroptera	
29.1	±4.06	NC	COA		Low abundance (18±16.4 N/sq.m.)	Hemiptera	
29.2		SG	COA	10-d	Significantly toxic (40% mortality)	*Chironomus tentans* (midge)	LAR
29.2	±9.07	NE	COA		High taxa (16.3±4.61 S)	Benthic species	
29.3	±2.52	SG	COA		Moderate abundance (10.3±4.04 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
29.6	±18.1	SG	COA		Low species richness (1.06±0.189; SRUs)	Benthic species	
29.7	±21.5	NC	COA		Low species richness (1.64±0.24; SRUs)	Benthic species	
29.7	±25.2	*	COA		Moderate species richness (1.53±0.284; SRUs)	Benthic species	
29.7	±40.3	NE	COA	48-h	Not toxic (4.82±5.91% mortality)	*Daphnia magna* (water flea)	NEO
29.7	±40.3	NE	COA	10-d	Not toxic (4±4.24% mortality)	*Daphnia magna* (water flea)	NEO
30		NE	COA		High abundance (720 N/sq.m.)	Chironomidae	
30		NE	COA		High abundance (582 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
30.3	±14	NE	COA		High diversity (2.31±0.177; Shannon Weaver Index)	Benthic species	
30.5	±30.4	NC	COA		Low diversity (1.45±0.262; Shannon Weaver Index)	Benthic species	
30.9	±8.22	NE	COA	28-d FT	Not significantly toxic (12.8±7.22% mortality)	*Hyalella azteca* (amphipod)	<2 mm
30.9	±8.22	NE	COA	28-d FT	Not significantly toxic (2±2.09% mortality)	*Oncorhynchus mykiss* (rainbow trout)	5 d
31	±12.9	SG	COA		Low abundance (53±33.9 N/0.00203 sq.m.)	Benthic invertebrates	
31.2	±4.74	NC	COA	28-d FT	Significantly toxic (11.5±4.95% mature males)	*Hyalella azteca* (amphipod)	<2 mm
31.4	±22.8	*	COA		Moderate abundance (91.7±30.5 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
31.4	±45	NE	COA		High biotic integrity (46±5.92; AIBI)	Benthic invertebrates	
31.5	±17.7	NE	COA		High abundance (6668±2109 N/sq.m.)	Chironomidae	
31.5	±17.7	NE	COA		High abundance (9094±2546 N/sq.m.)	Benthic invertebrates	
31.5	±7.5	NE	COA	14-d FT	Not toxic (22.8±9.95% mortality)	*Chironomus riparius* (midge)	<24 h
31.5	±7.5	NE	COA	16-22-d	Not toxic (10.5±1.37 offspring/brood; reproduction)	*Daphnia magna* (water flea)	<24 h
31.5	±7.5	NE	COA	16-22-d	Not toxic (20.4±2.88 offspring/brood; reproduction)	*Daphnia magna* (water flea)	<24 h
31.5	±7.5	NE	COA	16-22-d	Not toxic (33.4±3.21 offspring/brood; reproduction)	*Daphnia magna* (water flea)	<24 h
31.6	±5.23	NC	COA		Low abundance (142±139 N/sq.m.)	Chironomidae	
31.7	±9.28	NE	COA	28-d FT	Not significantly toxic (26.5±2.38% mature males)	*Hyalella azteca* (amphipod)	<2 mm
32.2	±8.13	SG	COA		Low abundance (5.4±8.05 N/sq.m.)	Amphipoda	

sediment-associated chromium (mg·kg^{-1} dry weight) in freshwater systems.

TOC (%)	AVS (µmol·g^{-1})	Unionized Ammonia (mg·L^{-1})	Sand (%)	Silt (%)	Clay (%)	Area	Reference
						Ontario	Fitchko 1989
1			42	43	13	Oakville Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
0.587±0.217						Trinity River, TX	Dickson et al. 1989
0.61±0.142						Trinity River, TX	Dickson et al. 1989
1						Ontario	Persaud et al. 1992
1.15±0.212			30.5±16.3	46.5±4.95	13.5±0.70	Oakville Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
0.617±0.129						Trinity River, TX	Dickson et al. 1989
0.59						Trinity River, TX	Dickson et al. 1989
						Trinity River, TX	Dickson et al. 1989
0.875±0.884			78±22.6	15±15.6	3.5±3.54	Toronto Outer Harbour	Jaagumagi 1988; et al. 1989
0.708±0.636			46.4±25.8	39.6±23.7	11.4±2.7	Oakville Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
0.654±0.144						Trinity River, TX	Dickson et al. 1989
2.05±1.88			57.7±28.9	30±20.1	9±7.55	Detroit River, MI	Jaagumagi 1988; et al. 1989
0.538±0.575			72±45.4	19.5±31.8	7.75±13.5	Port Weller Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
0.538±0.575			72±45.4	19.5±31.8	7.75±13.5	Port Weller Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
0.538±0.575			72±45.4	19.5±31.8	7.75±13.5	Port Weller Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
0.84						Trinity River, TX	Dickson et al. 1989
1.41	149 µg/g		74	12	14	Upper Clark Fork River, MT	Ingersoll et al. 1992b
1.41	149 µg/g		74	12	14	Upper Clark Fork River, MT	Ingersoll et al. 1992b
0.662±0.132						Trinity River, TX	Dickson et al. 1989
3.19±3.02	328±192 µg/g		75.7±6.03	11.7±5.51	12.7±0.58	Upper Clark Fork River, MT	Ingersoll et al. 1992b
5.03						St. Marys River, ON	Pope 1990
5.03						St. Marys River, ON	Pope 1990
1.56±0.205	348±281 µg/g		74.5±0.70	12	13.5±0.70	Upper Clark Fork River, MT	Ingersoll et al. 1992b
1.56±0.205	348±281 µg/g		74.5±0.70	12	13.5±1	Upper Clark Fork River, MT	Ingersoll et al. 1992b
0.92±0.664			38.6±25.4	46.6±23.2	12.3±2.69	Oakville Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
1.7	547 µg/g		75	12	13	Upper Clark Fork River, MT	Ingersoll et al. 1992b
0.422±0.214						Trinity River, TX	Dickson et al. 1989
0.93±0.616			37.6±24.4	45.9±21.4	16.3±2.49	Oakville Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
1.31±0.141	197±67.9 µg/	0.006±0.003	78±5.66	9±4.24	13±1.41	Upper Clark Fork River, MT	Ingersoll et al. 1992b
0.656±0.136						Trinity River, TX	Dickson et al. 1989
			9±5.57			Kishwaukee River Basin, IL	IEPA 1988a
2.12±0.59						Keweenaw Waterway, MI	Malueg et al. 1984a
2.12±0.59						Keweenaw Waterway, MI	Malueg et al. 1984a
2.12±0.59						Keweenaw Waterway, MI	Malueg et al. 1984a
2.12±0.59						Keweenaw Waterway, MI	Malueg et al. 1984a
4.12±3.09	248±224 µg/g		69.8±12.8	15.8±9.3	14.5±3.7	Upper Clark Fork River, MT	Ingersoll et al. 1992b
0.63						Trinity River, TX	Dickson et al. 1989
			16±16.6			Kishwaukee River Basin, IL	IEPA 1988a
1.4±0.361			19±8	59.3±11.4	14.3±0.57	Oakville Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
1.02±0.725			37.8±29.6	47.2±27.3	12.4±2.97	Oakville Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
3.52±3.38			44.7±37.6	45±33.2	7.33±5.03	Penetang Harbour, Lake Huron, ON	Jaagumagi 1988; et al. 1989
1.28±1.13			63.2±34.3	26±24.5	7.4±9.69	Toronto Outer Harbour	Jaagumagi 1988; et al. 1989
0.465±0.202						Trinity River, TX	Dickson et al. 1989
0.465±0.202						Trinity River, TX	Dickson et al. 1989
1.9			43	44	8	Detroit River, MI	Jaagumagi 1988; et al. 1989
1.9			43	44	8	Detroit River, MI	Jaagumagi 1988; et al. 1989
1.18±0.49			27.7±16	54.4±14.5	12.9±1.77	Oakville Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
3.63±4.77			51±50.9	40±45.3	7±7.07	Penetang Harbour, Lake Huron, ON	Jaagumagi 1988; et al. 1989
2.81±2.28	311±168 µg/	0.011±0.008	72.2±8.07	14.8±7.85	13±0.707	Upper Clark Fork River, MT	Ingersoll et al. 1992b
2.81±2.28	311±168 µg/	0.009±0.007	72.2±8.07	14.8±7.85	13±0.707	Upper Clark Fork River, MT	Ingersoll et al. 1992b
0.985±0.247	22±11.2		8.03±6.42	58.6±14.3	33.4±7.83	Galveston Bay, TX	Carr 1992
4.3±3.68	277±382 µg/	0.064±0.054	63.5±16.3	20±11.3	16.5±4.95	Upper Clark Fork River, MT	Ingersoll et al. 1992b
4.23±3.25						St. Mary's River, ON	Jaagumagi 1988; et al. 1989
0.416±0.202						Trinity River, TX	Dickson et al. 1989
1.58±1.87			78.5±6.36	14.5±10.6	2	Niagara River, NY	Jaagumagi 1988; et al. 1989
1.58±1.87			78.5±6.36	14.5±10.6	2	Niagara River, NY	Jaagumagi 1988; et al. 1989
3.5±2.63	261±195 µg/	0.039±0.06	68.8±11	17±8.85	14.2±2.93	Upper Clark Fork River, MT	Ingersoll et al. 1992b
3.5±2.63	261±195 µg/	0.038±0.065	68.8±11	17±8.85	14.2±2.93	Upper Clark Fork River, MT	Ingersoll et al. 1992b
3.5±2.63	261±195 µg/	0.038±0.065	68.8±11	17±8.85	14.2±2.93	Upper Clark Fork River, MT	Ingersoll et al. 1992b
3.5±2.63	261±195 µg/	0.038±0.065	68.8±11	17±8.85	14.2±2.93	Upper Clark Fork River, MT	Ingersoll et al. 1992b
1.47±0.665	415±548 µg/g		62±15.6	24.5±10.6	14.5±4.95	Milltown Reservoir, MT	Ingersoll et al. 1992b
3.09±2.53	252±121 µg/	0.007±0.001	71.5±9.15	15.5±8.89	13±0.816	Upper Clark Fork River, MT	Ingersoll et al. 1992b
3.85±2.77	203±152 µg/g		67.6±11.8	18±9.51	14.4±3.21	Upper Clark Fork River, MT	Ingersoll et al. 1992b

Appendix Ia. Summary of the available biological effects and related physicochemical data for

Chromium (mg·kg^{-1})		Hit	Analysis Type	Test Type	Endpoint Measured	Species	Life Stage
32.3	±45.2	NE	COA		High abundance (567±173 N/sq.m.)	Gastropoda	
32.3	±8.06	SG	COA		Low abundance (55.2±32.3 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
32.5	±13.6	NE	COA		High biotic integrity (5.18±0.713; MBI)	Macroinvertebrates	
32.5	±3.52	NC	COA		Low abundance (0 N/sq.m.)	Gastropoda	
32.6	±14.2	NE	COA		High biotic integrity (47.9±4.36; AIBI)	Freshwater fish	
33.1	±10.8	NC	COA		Low abundance (533±162 N/sq.m.)	Chironomidae	
33.1	±9.09	NE	COA	16-22-d	Not significantly toxic (0% mortality)	*Daphnia magna* (water flea)	<24 h
33.5	±14.8	NE	COA		High abundance (705±196 N/sq.m.)	Gastropoda	
33.5	±47.6	NE	COA	10-d	Not toxic (9±3.66% mortality)	*Hyalella azteca* (amphipod)	<6 mm
33.5	±8.84	SG	COA		Low abundance (11.3±8.62 N/sq.m.)	Gastropoda	
33.6	±2.8	NE	COA		High abundance (3343±65.1 N/sq.m.)	Chironomidae	
33.7	±13.7	SG	COA		Moderate abundance (24.3±6.11 N/sq.m.)	Amphipoda	
34		NC	COA	14-d FT	Significantly toxic (2.8% sexual maturity)	*Hyalella azteca* (amphipod)	LAR
34	±13.1	NE	COA		High diversity (3.12±0.0721; Shannon Weaver Index)	Benthic species	
34	±15	NE	COA		High abundance (368±76.9 N/sq.m.)	Chironomidae	
34	±2.12	NE	COA		High abundance (86865±16330 N/sq.m.)	Benthic invertebrates	
34	±5.9	NE	COA		High taxa (15.8±2 S)	Benthic species	
34.1	±4.72	NC	COA		Low abundance (9.8±10.1 N/sq.m.)	Hemiptera	
34.1	±4.74	NC	COA		Low abundance (1.8±4.02 N/sq.m.)	Amphipoda	
34.2	±4.8	NE	COA	28-d FT	Not toxic (34±10.4% mature males)	*Hyalella azteca* (amphipod)	<2 mm
34.3	±4.26	NE	COA	14-d FT	Not significantly toxic (19.5±6.6% mortality)	*Chironomus riparius* (midge)	<24 h
34.5		SG	COA	28-d FT	Significantly toxic (52% mortality)	*Hyalella azteca* (amphipod)	<2 mm
34.5		SG	COA	28-d FT	Significantly toxic (77.5% mortality)	*Oncorhynchus mykiss* (rainbow trout)	5 d
34.5		NE	COA		High abundance (2071 N/sq.m.)	Chironomidae	
34.8		NC	COA	10-d ST	Significantly toxic (100% mortality)	*Chironomus tentans* (midge)	LAR
34.9	±4.53	NE	COA	16-22-d	Not toxic (23.2±2.65 offspring/brood; reproduction)	*Daphnia magna* (water flea)	<24 h
34.9	±6.1	NC	COA		Low abundance (38±27.6 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
35		NE	COA		High abundance (215 N/sq.m.)	Chironomidae	
35		NE	COA		High abundance (214 N/sq.m.)	Amphipoda	
35		NE	COA		High abundance (15 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
35	±23.6	NC	COA		Low abundance (536±569 N/sq.m.)	Chironomidae	
35	±23.6	NC	COA		Low abundance (87.3±93.7 N/sq.m.)	Amphipoda	
35	±23.6	NC	COA		Low abundance (1032±693 N/sq.m.)	Benthic invertebrates	
35	±4.32	NE	COA	28-d FT	Not toxic (9.29±8.64% mortality)	*Hyalella azteca* (amphipod)	<2 mm
35.3	±0.252	NE	COA		High abundance (333±189 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
35.6	±4.44	NE	COA	16-22-d	Not toxic (0% mortality)	*Daphnia magna* (water flea)	<24 h
35.6	±4.44	NE	COA	16-22-d	Not toxic (10.9±1.14 offspring/brood; reproduction)	*Daphnia magna* (water flea)	<24 h
35.6	±4.44	NE	COA	16-22-d	Not toxic (21.3±3.41 offspring/brood; reproduction)	*Daphnia magna* (water flea)	<24 h
35.6	±4.44	NE	COA	21-d FT	Not toxic (2.1±2.92% mortality)	*Oncorhynchus mykiss* (rainbow trout)	5 d
35.8	±5.71	SG	COA		Low abundance (12339±2051 N/sq.m.)	Benthic invertebrates	
36.3	±12.7	NE	COA		High abundance (405±68.6 N/sq.m.)	Hemiptera	VAR
36.3	±21.9	NE	COA	48-h	Not significantly toxic (3.9±3.26% mortality)	*Daphnia magna* (water flea)	JUV
36.4			TEL				
36.5	±23.3	NE	COA		High abundance (152±72.8 N/sq.m.)	Chironomidae	
36.7	±29.8	NC	COA		Low species richness (2.05±0.356; SRUs)	Benthic species	
37		NC	COA		Low biotic integrity (7.9; MBI)	Macroinvertebrates	
37.2	±3.04	SG	COA	28-d FT	Toxic (11.5±2.12% mature males)	*Hyalella azteca* (amphipod)	<2 mm
37.3	±16.6	NE	COA		High diversity (2.62±0.377; Shannon Weaver Index)	Benthic species	
37.3	±16.6	NE	COA		High species richness (1.95±0.306; SRUs)	Benthic species	
37.3	±2.83	NE	COA		High abundance (27 N/sq.m.)	Amphipoda	
37.4	±19.7	NE	COA	4-d	Not significantly toxic (8±6.3% mortality)	*Gammarus pseudolimnaeus* (Scud)	VAR
37.4	±19.7	NE	COA	4-d	Not significantly toxic (2±4.8% mortality)	*Procambarus* sp. (crayfish)	VAR
37.4	±19.7	NE	COA	4-d	Not significantly toxic (16.5±21.7% mortality)	*Hexagenia* sp. (mayfly)	VAR
37.4	±19.7	NE	COA	4-d	Not significantly toxic (13.5±13.8% mortality)	*Physa gyrina* (snail)	VAR
37.4	±19.7	NE	COA	4-d	Not significantly toxic (0% mortality)	*Truncilla donaciformis* (fawnfoot clam)	VAR
37.4	±19.7	NE	COA	4-d	Not significantly toxic (11±8.4% mortality)	*Sphaerium* sp. (fingernail clam)	VAR
37.4	±2.69	NE	COA		High abundance (130±19.8 N/sq.m.)	Hemiptera	
37.7	±14.4	NE	COA		High biotic integrity (6.02±0.47; MBI)	Macroinvertebrates	
37.7	±16.2	SG	COA		Low abundance (57.3±69.9 N/sq.m.)	Gastropoda	
37.9	±20.1	NE	COA		High species richness (2.14±0.516; SRUs)	Benthic species	
38.2	±19.8	SG	COA		Low abundance (495±275 N/sq.m.)	Chironomidae	
38.2	±19.8	SG	COA		Low abundance (6.78±9.85 N/sq.m.)	Gastropoda	
38.3	±2.93	SG	COA		Moderate abundance (1707±271 N/sq.m.)	Chironomidae	
38.4	±2.76	NE	COA		High abundance (116±58.5 N/sq.m.)	Gastropoda	

sediment-associated chromium (mg·kg^{-1} dry weight) in freshwater systems.

TOC (%)	AVS (µmol·g^{-1})	Unionized Ammonia (mg·L^{-1})	Sand (%)	Silt (%)	Clay (%)	Area	Reference
2.44±3.98			71.3±26.3	24.3±24.1	2.67±2.08	St. Mary's River, ON	Jaagumagi 1988; et al. 1989
3.91±2.71	283±209 µg/g		67.8±11.9	18±9.51	14 2±3.27	Upper Clark Fork River, MT	Ingersoll et al. 1992b
			19.6±18			Kishwaukee River Basin, IL	IEPA 1988a
1.37±0.739	249±256 µg/g		60.5±9.26	26.3±6.99	13.5±2.52	Milltown Reservoir, MT	Ingersoll et al. 1992b
			20.2±19.3			Kishwaukee River Basin, IL	IEPA 1988a
3.65±2.77	287±122 µg/g		70.7±11	16.7±10.5	12.7±0.57	Upper Clark Fork River, MT	Ingersoll et al. 1992b
4.59±2.6	292±241 µg/	0.054±0.078	64.3±10.3	21±7.79	14.8±3.5	Upper Clark Fork River, MT	Ingersoll et al. 1992b
1.72±1.67			81.5±10.6	14±11.3	1.5±0.707	Niagara River, NY	Jaagumagi 1988; et al. 1989
0.438±0.233						Trinity River, TX	Dickson et al. 1989
4.47±2.79	217±172 µg/g		66±13	19.5±10.3	14.5±3.7	Upper Clark Fork River, MT	Ingersoll et al. 1992b
1.03±0.4	268±336 µg/g		58±5.7	29±5.7	13	Milltown Reservoir, MT	Ingersoll et al. 1992b
3.03±1.07			42.3±21.9	45.7±19.1	8.33±2.89	Midland Bay, Lake Huron, ON	Jaagumagi 1988; et al. 1989
1.4	3.7	0.049	71.2	9.9	16.4	Saginaw River, MI	Ingersoll et al. 1992a
3.5±0.265			45.7±27.6	43.3±23.1	7.67±4.04	Midland Bay, Lake Huron, ON	Jaagumagi 1988; et al. 1989
1.4±0.265			19.6±8.82	60.8±11.4	13.4±1.52	Oakville Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
1.5±0.852	323±256 µg/g		56.3±4 93	29.3±4.04	14.3±2.31	Milltown Reservoir, MT	Ingersoll et al. 1992b
						DuPage River Basin, IL	IEPA 1988b
1.86±1.01	451±277 µg/g		57.4±8.91	28.4±6.5	14.4±2.97	Milltown Reservoir, MT	Ingersoll et al. 1992b
1.74±1.03	297±246 µg/g		59.6±8.26	27±6.28	13.4±2.19	Milltown Reservoir, MT	Ingersoll et al. 1992b
1.64±0.974	371±336 µg/	0.1±0.004	59.2±8.7	27.4±6.58	13.6±2.61	Milltown Reservoir, MT	Ingersoll et al. 1992b
1.77±0.929	381±302 µg/	0.015±0.013	58.2±8.18	27.8±5.98	14.2±2.71	Milltown Reservoir, MT	Ingersoll et al. 1992b
6.9	7.4 µg/g	0.102	52	28	20	Upper Clark Fork River, MT	Ingersoll et al. 1992b
6.9	7.4 µg/g	0.03	52	28	20	Upper Clark Fork River, MT	Ingersoll et al. 1992b
6.9	7.4 µg/g		52	28	20	Upper Clark Fork River, MT	Ingersoll et al. 1992b
0.2	1.2	0.846	98.2	1.5	2.3	Saginaw River, MI	Ingersoll et al. 1992a
1.97±0.879	357±330 µg/	0.015±0.012	59±8.86	26.8±6.06	14.4±2.97	Milltown Reservoir, MT	Ingersoll et al. 1992b
1.91±1.21	273±260 µg/g		55.8±13.5	30±9.63	14.8±3.5	Milltown Reservoir, MT	Ingersoll et al. 1992b
0.78			47	43	8	Port Weller Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
0.78			47	43	8	Port Weller Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
0.78			47	43	8	Port Weller Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
3.29±2.45			44.3±33.3	43.7±26.6	10.3±8.37	Penetang Harbour, Lake Huron, ON	Jaagumagi 1988; et al. 1989
3.29±2.45			44.3±33.3	43.7±26.6	10.3±8.37	Penetang Harbour, Lake Huron, ON	Jaagumagi 1988; et al. 1989
3.29±2.45			44.3±33.3	43.7±26.6	10.3±8.37	Penetang Harbour, Lake Huron, ON	Jaagumagi 1988; et al. 1989
1.9±0.914	337±300 µg/	0.013±0.011	55.6±10.1	29.6±7.14	15±3.32	Milltown Reservoir, MT	Ingersoll et al. 1992b
1.89±0.567	422±386 µg/g		55.3±5.86	29±3.61	16±2.65	Milltown Reservoir, MT	Ingersoll et al. 1992b
2.09±0.836	309±318 µg/	0.055±0.069	55.8±11.1	29±7.64	15.3±3.5	Milltown Reservoir, MT	Ingersoll et al. 1992b
2.09±0.836	309±318 µg/	0.055±0.069	55.8±11.1	29±7.64	15.3±3.5	Milltown Reservoir, MT	Ingersoll et al. 1992b
2.09±0.836	309±318 µg/	0.055±0.069	55.8±11.1	29±7.64	15.3±3.5	Milltown Reservoir, MT	Ingersoll et al. 1992b
2.09±0.836	309±318 µg/	0.018±0.017	55.8±11.1	29±7.64	15.3±3.5	Milltown Reservoir, MT	Ingersoll et al. 1992b
2.2±0.95	347±368 µg/g		55±13.7	29.8±9.5	15.5±4.2	Milltown Reservoir, MT	Ingersoll et al. 1992b
2.25±1.18	287±194 µg/g		67±9.9	19.5±10.6	13.5±0.70	Upper Clark Fork River, MT	Ingersoll et al. 1992b
2±0.6						Keweenaw Waterway, MI	Malueg et al. 1984a
1.18±1.31			52	39	9	Humber Bay, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
5.13±5.95			62.3±29	28±25.5	6.33±5.86	Bay of Quinte, ON	Jaagumagi 1988; et al. 1989
						DuPage River Basin, IL	IEPA 1988b
2.55±0.17	252±258 µg/	0.02±0.023	46.5±9.19	35±7.07	18.5±2.12	Milltown Reservoir, MT	Ingersoll et al. 1992b
0.81±0.576			47.7±44	39±30.2	12.3±14	Port Weller Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
0.81±0.576			47.7±44	39±30.2	12.3±14	Port Weller Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
2.31±0.516	436±518 µg/g		45.5±7.78	36±5.66	19±1.41	Milltown Reservoir, MT	Ingersoll et al. 1992b
			82.1±31.3	9.8±15.9	5.6±10.2	Upper Mississippi River, MS	Marking et al. 1981
			82.1±31.3	9.8±15.9	5.6±10.2	Upper Mississippi River, MS	Marking et al. 1981
			82.1±31.3	9.8±15.9	5.6±10.2	Upper Mississippi River, MS	Marking et al. 1981
			82.1±31.3	9.8±15.9	5.6±10.2	Upper Mississippi River, MS	Marking et al. 1981
			82.1±31.3	9.8±15.9	5.6±10.2	Upper Mississippi River, MS	Marking et al. 1981
			82.1±31.3	9.8±15.9	5.6±10.2	Upper Mississippi River, MS	Marking et al. 1981
1.99±0.969	50±27.6 µg/g		51±15.6	32.5±10.6	16.5±4.95	Milltown Reservoir, MT	Ingersoll et al. 1992b
						DuPage River Basin, IL	IEPA 1988b
2.45±2.18			40±39	44.7±34.9	11.7±8.74	Niagara River, NY	Jaagumagi 1988; et al. 1989
3.58±0.985			41.4±23.9	44.7±18.5	10.4±6.8	Midland Bay, Lake Huron, ON	Jaagumagi 1988; et al. 1989
3.28±0.977			40±22.8	45.8±17.5	10.7±6.63	Midland Bay, Lake Huron, ON	Jaagumagi 1988; et al. 1989
3.28±0.977			40±22.8	45.8±17.5	10.7±6.63	Midland Bay, Lake Huron, ON	Jaagumagi 1988; et al. 1989
2.76±0.39	331±228 µg/g		49.7±8.5	33.3±5.77	16.7±3.51	Milltown Reservoir, MT	Ingersoll et al. 1992b
2.6±0.628	454±368 µg/g		49±8.19	34±5.29	17±3.61	Milltown Reservoir, MT	Ingersoll et al. 1992b

Appendix Ia. Summary of the available biological effects and related physicochemical data for

Chromium ($mg \cdot kg^{-1}$)		Hit	Analysis Type	Test Type	Endpoint Measured	Species	Life Stage
38.9	±17.2	NC	COA		Moderate abundance (3304±588 N/sq.m.)	Benthic invertebrates	
38.9	±21.9	NC	COA		Low abundance (15.9±13.3 N/sq.m.)	Gastropoda	
39	±13.9	SG	COA		Low abundance (232±180 N/sq.m.)	Chironomidae	
39	±13.9	SG	COA		Low abundance (2186±1451 N/sq.m.)	Benthic invertebrates	
39.3		SG	COA	14-d FT	Significantly toxic (53% mortality)	*Chironomus riparius* (midge)	<24 h
39.3		SG	COA	16-22-d	Significantly toxic (14.3 offspring/brood; reproduction)	*Daphnia magna* (water flea)	<24 h
39.3	±45.8	NC	COA		Low abundance (4804±1517 N/sq.m.)	Benthic invertebrates	
39.8	±23.8	SG	COA		Moderate diversity (2.43±0.239; Shannon Weaver Index)	Benthic species	
39.9	±7.64	NE	COA		High abundance (57732±15882 N/sq.m.)	Benthic invertebrates	
40.1	±20.3	SG	COA		Low abundance (4.25±5.04 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
40.2	±45.5	*	COA		Low abundance (21.4±26.5 N/sq.m.)	Amphipoda	
40.2	±45.5	*	COA		Low abundance (1.71±3.15 N/sq.m.)	Gastropoda	
40.3	±11.6	*	COA		Low abundance (104±76.6 N/sq.m.)	Amphipoda	
40.3	±11.6	*	COA		Low abundance (261±321 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
40.5	±23.1	*	COA		Low abundance (0 N/sq.m.)	Amphipoda	
40.6	±19.7	*	COA		Low abundance (819±241 N/sq.m.)	Benthic invertebrates	
40.8	±49.9	SG	COA		Low abundance (113±118 N/sq.m.)	Chironomidae	
42		*	COA		Low diversity (1.37; Shannon Weaver Index)	Benthic species	
42.4	±28.1	NE	COA		High diversity (3.02±0.251; Shannon Weaver Index)	Benthic species	
43	±17	NE	COA		High abundance (4818±791 N/sq.m.)	Amphipoda	
43.4	±22.5	SG	COA		Low taxa (8.4±0.55 S)	Benthic species	
44.4	±29.7	NE	COA		High species richness (2.67±0.178; SRUs)	Benthic species	
44.6	±57.2	NE	COA		High abundance (341±80.6 N/sq.m.)	Gastropoda	
45	±14.1	NE	COA		High abundance (65.5±16.3 N/sq.m.)	Gastropoda	
45	±14.1	NE	COA		High abundance (1667±26.2 N/sq.m.)	Benthic invertebrates	
45	±35.4	NC	COA		Moderate abundance (4150±254 N/sq.m.)	Benthic invertebrates	
45.8	±25.8	NC	COA		Low diversity (2.22±0.064; Shannon Weaver Index)	Benthic species	
45.8	±25.8	NC	COA		Low taxa (5.5±0.707 S/sq.m.)	Benthic species	
46		NE	COA		High abundance (2771 N/sq.m.)	Chironomidae	
47.9	±36	SG	COA		Low abundance (33.7±45.2 N/sq.m.)	Gastropoda	
47.9	±65.3	NE	COA		High diversity (2.53±0.585; Shannon Weaver Index)	Benthic species	
49.3	±18.2	NC	COA		Moderate species (2.83±0.753 S/0.044 sq.m.)	Benthic species	
49.6	±30.1	NC	COA		Low abundance (0 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
49.6	±39	*	COA		Low abundance (2121±1924 N/sq.m.)	Benthic invertebrates	
49.8	±38.8	*	COA		Low abundance (64.8±75.7 N/sq.m.)	Amphipoda	
50	±24.5	NE	COA	10-d ST	Not toxic (13.5±4.73% mortality)	*Hyalella azteca* (amphipod)	
50	±25	*	SQG		Moderately Polluted - USEPA Region 5 Harbour Classification		
50.1	±25	*	COA		Low abundance (0 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
50.9	±25.7	NC	COA		Significantly toxic (19.2±4.45% deformities)	Chironomidae	
50.9	±25.7	NC	COA		Low abundance (176±163 N/sq.m.)	Chironomidae	
51	±20.2	NC	COA		Low abundance (21.5±27.8 N/0.044 sq.m.)	Benthic invertebrate	
51.4	±41.1	*	COA		Moderate diversity (2.56±0.39; Shannon Weaver Index)	Benthic species	
52	±22.4	NE	COA		High diversity (0.348±0.057; Shannon Weaver Index)	Benthic species	
53		NE	COA		High abundance (15830 N/sq.m.)	Benthic invertebrates	
54.2	±24.4	NE	COA	10-d ST	Not significantly toxic (43.3±22.5% mortality)	*Chironomus tentans* (midge)	LAR
54.3	±24.3	NE	COA	14-d FT	Not significantly toxic (13.1±5.95% sexual maturity)	*Hyalella azteca* (amphipod)	LAR
54.4	±24.6	NE	COA	14-d FT	Not significantly toxic (7.61±5.17% mortality)	*Hyalella azteca* (amphipod)	LAR
54.4	±24.6	NE	COA	28-d FT	Not significantly toxic (5.03±3.41% mortality)	*Hyalella azteca* (amphipod)	LAR
54.4	±24.6	NE	COA	14-d FT	Not significantly toxic (11.5±8.37% mortality)	*Chironomus riparius* (midge)	24 h
54.4	±24.6	NE	COA	28-d FT	Not significantly toxic (82.2±11.3% sexual maturity)	*Hyalella azteca* (amphipod)	LAR
54.5		NE	COA		Not significantly toxic (2% deformities)	Chironomidae	
54.5		NE	COA		High abundance (2204 N/sq.m.)	Chironomidae	
54.5		NE	COA		High abundance (7152 N/sq.m.)	Benthic invertebrates	
55		NE	SBA		Sediment Quality Criteria - No Effects Threshold		
55		*	SLCA		Sediment Quality Criteria - Minimal Effects Threshold		
55	±12.7	SG	COA		Low diversity (0.1±0.085; Shannon Weaver Index)	Benthic species	
55.5	±40.6	*	COA		Low species richness (1.61±0.445; SRUs)	Benthic species	
56	±39.6	NE	COA	48-h	Not significantly toxic (0.85±1.2% mortality)	*Daphnia magna* (water flea)	JUV
56	±39.6	NE	COA	48-h	Not significantly toxic (0.85±1.2% mortality)	*Daphnia magna* (water flea)	JUV
56	±39.6	NE	COA	10-d	Not significantly toxic (16.7±4.74% mortality)	*Hexagenia limbata* (mayfly)	NYM
58.5	±45.1	NE	COA	48-h	Not significantly toxic (2.49±1.77% mortality)	*Daphnia magna* (water flea)	JUV
59.7	±28.8	SG	COA		Low taxa (6.67±2.5 S)	Benthic species	
60.8	±44.3	*	COA		Low abundance (0.8±1.79 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
61	±60.7	NE	COA		High abundance (10577±2385 N/sq.m.)	Benthic invertebrates	

sediment-associated chromium (mg·kg^{-1} dry weight) in freshwater systems.

TOC (%)	AVS (µmol·g^{-1})	Unionized Ammonia (mg·L^{-1})	Sand (%)	Silt (%)	Clay (%)	Area	Reference
1.87±0.896	3.4±0.265	0.053±0.007	66.1±17.5	9.63±3.41	22.6±14.1	Saginaw River, MI	Ingersoll et al. 1992a
3.67±2.32			39.3±31.7	47.6±25.4	11.4±8.3	Penetang Harbour, Lake Huron, ON	Jaagumagi 1988; et al. 1989
2.55±2.03			42±42.3	44.3±35.5	11.3±9.29	Niagara River, NY	Jaagumagi 1988; et al. 1989
2.55±2.03			42±42.3	44.3±35.5	11.3±9.29	Niagara River, NY	Jaagumagi 1988; et al. 1989
2.67	69.5 µg/g	0.003	40	40	20	Milltown Reservoir, MT	Ingersoll et al. 1992b
2.67	69.5 µg/g	0.003	40	40	20	Milltown Reservoir, MT	Ingersoll et al. 1992b
1.13±1.91			69±37.2	20.2±25.3	7.33±12.3	Toronto Harbour, ON	Jaagumagi 1988; et al. 1989
3.62±1.23			39.3±24.3	45.3±18.2	11.8±7.78	Midland Bay, Lake Huron, ON	Jaagumagi 1988; et al. 1989
4.99±2.7	216±295 µg/g		56±5.66	27.5±0.70	16.5±4.95	Upper Clark Fork River, MT	Ingersoll et al. 1992b
3.43±0.932			38.8±24	47±18.3	11.1±6.94	Midland Bay, Lake Huron, ON	Jaagumagi 1988; et al. 1989
1.54±1.51			60.2±32.6	28.5±23.4	7.67±8.69	Toronto Outer Harbour	Jaagumagi 1988; et al. 1989
1.54±1.51			60.2±32.6	28.5±23.4	7.67±8.69	Toronto Outer Harbour	Jaagumagi 1988; et al. 1989
2.64±1.67			50±38.1	38.8±31	9±8.91	Niagara River, NY	Jaagumagi 1988; et al. 1989
2.64±1.67			50±38.1	38.8±31	9±8.91	Niagara River, NY	Jaagumagi 1988; et al. 1989
3.67±1.15			36.5±21.3	47.3±15.9	12.2±7.47	Midland Bay, Lake Huron, ON	Jaagumagi 1988; et al. 1989
3.46±0.86			36.6±21.8	48.5±16.5	11.4±6.72	Midland Bay, Lake Huron, ON	Jaagumagi 1988; et al. 1989
1.54±1.66			59.8±36.4	29±26.1	8±9.67	Toronto Outer Harbour	Jaagumagi 1988; et al. 1989
7.4			53	28	18	St. Mary's River, ON	Jaagumagi 1988; et al. 1989
3.89±2.74			38.4±31.8	48.5±25	11.6±8.35	Penetang Harbour, Lake Huron, ON	Jaagumagi 1988; et al. 1989
1.07±0.467			29±12.7	57.5±12	10±1.41	Humber Bay, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
			31.5±23.9			Kishwaukee River Basin, IL	IEPA 1988a
3.98±2.95			39.3±34.3	47.6±26.8	12.1±8.88	Penetang Harbour, Lake Huron, ON	Jaagumagi 1988; et al. 1989
4.53±6.05			47.5±53	43.5±44.5	8±8.49	Penetang Harbour, Lake Huron, ON	Jaagumagi 1988; et al. 1989
1.09±0.438			25.5±30.4	55±17	18±14.1	Port Weller Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
1.09±0.438			25.5±30.4	55±17	18±14.1	Port Weller Harbour, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
2.2±1.13			31.5±24.7	49.5±13.4	15±12.7	Toronto Outer Harbour	Jaagumagi 1988; et al. 1989
5.39±0.509						St. Marys River, ON	Pope 1990
5.39±0.509						St. Marys River, ON	Pope 1990
1.9	13.5		33.6	7.9	55.8	Buffalo River, NY	Ingersoll et al. 1992a
6.2±4.19			37.8±17.3	51±18.3	10.3±6.6	St. Mary's River, ON	Jaagumagi 1988; et al. 1989
1.36±1.86			69.1±39.9	20.6±27.4	8.57±12.9	Toronto Harbour, ON	Jaagumagi 1988; et al. 1989
0.695±0.508	0.083±0.145		9.12±8.71			Mississippi River, LA, MS	EMAP 1991
7.36±5.94			52.8±28.8	36.8±24.9	7.6±5.9	Bay of Quinte, ON	Jaagumagi 1988; et al. 1989
5.77±4.2			45±20.4	45.5±19.6	8±6.26	St. Mary's River, ON	Jaagumagi 1988; et al. 1989
5.73±4.25			48.2±26.7	43.2±23.9	7.83±6.46	St. Mary's River, ON	Jaagumagi 1988; et al. 1989
4.78±6.85			49.8±46.7			Everett, WA	Johnson and Norton 1989
						United States	USEPA 1977 (As cited in SAIC 199
4.78±2.76			28.4±27.3	56.4±22.5	13.4±7.81	Penetang Harbour, Lake Huron, ON	Jaagumagi 1988; et al. 1989
2.23±1.49	4.35±2.67	0.059±0.031	63.2±25.2	9.72±5.04	25.8±19.2	Saginaw River, MI	Ingersoll et al. 1992a
2.23±1.49	4.35±2.67	0.06±0.03	63.2±25.2	9.72±5.04	25.8±19.2	Saginaw River, MI	Ingersoll et al. 1992a
0.49±0.3	0.041±0.061		6.3±5.79			Mississippi River, LA, MS	EMAP 1991
5.45±4.59			47.2±29.7	46.2±25.4	5.8±4.6	St. Mary's River, ON	Jaagumagi 1988; et al. 1989
0.729±0.457	0.036±0.052		10.2±8.27			Mississippi River, LA, MS	EMAP 1991
2.1			52	39	9	Humber Bay, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
2.38±1.29	4.98±2.43	0.33±0.29	56.4±18.4	11.7±3.12	30.3±15.4	Saginaw River, MI	Ingersoll et al. 1992a
2.18±1.54	4.48±2.96	0.055±0.034	60.9±24.9	10.3±5.24	28±18.7	Saginaw River, MI	Ingersoll et al. 1992a
2.05±1.34	4.2±2.57	0.05±0.03	57.3±26.5	9.67±4.38	31.4±23.7	Saginaw River, MI	Ingersoll et al. 1992a
2.05±1.34	4.2±2.57	0.05±0.03	57.3±26.5	9.67±4.38	31.4±23.7	Saginaw River, MI	Ingersoll et al. 1992a
2.05±1.34	4.2±2.57	0.05±0.03	57.3±26.5	9.67±4.38	31.4±23.7	Saginaw River, MI	Ingersoll et al. 1992a
2.05±1.34	4.2±2.57	0.05±0.03	57.3±26.5	9.67±4.38	31.4±23.7	Saginaw River, MI	Ingersoll et al. 1992a
1.05		0.024	57.4	13.2	29.2	Saginaw River, MI	Ingersoll et al. 1992a
1.05		0.024	57.4	13.2	29.2	Saginaw River, MI	Ingersoll et al. 1992a
1.05		0.024	57.4	13.2	29.2	Saginaw River, MI	Ingersoll et al. 1992a
						St. Lawrence River	EC/MINVEQ 1992
						St. Lawrence River	EC/MINVEQ 1992
0.936±0.87	0.3		12.1±14.6			Mississippi River, LA, MS	EMAP 1991
6.19±4.41			54.4±24.4	38±22.6	6.8±6.65	St. Mary's River, ON	Jaagumagi 1988; et al. 1989
5.85±7.42						Little Grizzly Creek System, CA	Malueg et al. 1984b
5.85±7.42						Little Grizzly Creek System, CA	Malueg et al. 1984b
5.85±7.42						Little Grizzly Creek System, CA	Malueg et al. 1984b
2.08±0.53						Keweenaw Waterway, MI	Malueg et al. 1984a
						DuPage River Basin, IL	IEPA 1988b
6.6±4.86			47.2±29.7	46.2±25.4	5.8±4.6	St. Mary's River, ON	Jaagumagi 1988; et al. 1989
1.95±1.96			66±26.2	24.3±19.6	5.33±4.04	Toronto Outer Harbour	Jaagumagi 1988; et al. 1989

Appendix Ia. Summary of the available biological effects and related physicochemical data for

Chromium (mg·kg^{-1})		Hit	Analysis Type	Test Type	Endpoint Measured	Species	Life Stage
61.5	±10.8	NC	COA		Moderate species (10.7±1.63 S/0.044 sq.m)	Benthic species	
62	±11.1	NE	COA		High abundance (950±222 N/0.044 sq.m.)	Benthic invertebrate	
62	±29.7	NC	COA		Low abundance (0 N/sq.m.)	Gastropoda	
62.5	±23.3	NE	COA		High abundance (301±88.4 N/sq.m.)	Chironomidae	
62.9	±30.3	SG	COA		Low abundance (806±434 N/sq.m.)	Benthic invertebrates	
64.2	±12.5	NC	COA		Low diversity (0.492±0.073; Shannon Weaver Index)	Benthic species	
64.3	±91.5	*	COA		Moderate abundance (255±91.4 N/sq.m.)	Gastropoda	
64.7	±64.2	NC	COA		Low abundance (731±621 N/sq.m.)	Benthic invertebrates	
65		NE	COA	30-d	Not significantly toxic (645% increase in number)	*Lumbriculus variegatus* (oligochaete)	ADT
65.3	±42.3	NE	COA	10-d	Not significantly toxic (12.1±11.4% mortality)	*Hexagenia limbata* (mayfly)	NYM
65.3	±82.9	NE	COA		High abundance (102±46.8 N/sq.m.)	Amphipoda	
66.3	±66.3	NE	COA		High species richness (2±0.573; SRUs)	Benthic species	
68	±16.7	NC	COA		Low abundance (163±46.8 N/sq.m.)	Chironomidae	
69	±7.07	NE	COA		High abundance (277±79.2 N/0.044 sq.m.)	Benthic invertebrate	
69.6	±85.4	*	COA		Low species richness (0.655±0.021; SRUs)	Benthic species	
70		NE	COA	28-d FT	Not significantly toxic (7.5% mortality)	*Hyalella azteca* (amphipod)	LAR
70	±49.2	*	COA		Low abundance (6.67±6.11 N/sq.m.)	Gastropoda	
71.8	±31.2	*	COA		Low abundance (107±119 N/sq.m.)	Chironomidae	
72.1		*	SBA		Texas Water Commission Screening Levels		
72.3	±10.7	NC	COA		Low diversity (0.5±0.07; Shannon Weaver Index)	Benthic species	
73		NC	COA		Moderate abundance (3686 N/sq.m.)	Benthic invertebrates	
74		NG	COA		Low species (0 S/0.044 sq.m.)	Benthic species	
74		NE	COA		High species (8 S/0.044 sq.m.)	Benthic species	
74	±25.5	NC	COA		Significantly toxic (15.5±2.12% deformities)	Chironomidae	
74.4	±17.1	NC	COA		Significantly toxic (16.6±4.83% deformities)	Chironomidae	
74.8		NC	COA	10-d ST	Significantly toxic (50% mortality)	*Hexagenia limbata* (mayfly)	LAR
>75		*	SQG		Heavily Polluted - USEPA Region 5 Harbour Classification		
75.2	±38.7	SG	COA		Low abundance (295±146 N/0.044 sq.m.)	Benthic invertebrate	
75.3	±10.5	NC	COA		Low abundance (51.5±15 N/0.044 sq.m.)	Benthic invertebrate	
75.3	±10.5	NC	COA		Low species (6.5±2.65 S/0.044 sq.m.)	Benthic species	
77		NC	COA		Low species (5 S/0.044 sq.m)	Benthic species	
77		NE	COA		High abundance (378 N/sq.m.)	Gastropoda	
77	±8.49	*	COA		Low abundance (265±35.4 N/sq.m.)	Benthic invertebrates	
78.8	±85.1	NE	COA		High abundance (385±41.4 N/sq.m.)	Chironomidae	
81	±18	NE	COA	10-d ST	Not significantly toxic (3.08±3.84% mortality)	*Lumbriculus variegatus* (oligochaete)	ADT
81	±18	NE	COA	10-d ST	Not significantly toxic (6.15±7.40% mortality)	*Chironomus riparius* (midge)	LAR
81	±18	NE	COA	10-d ST	Not significantly toxic (5.77±6.07% mortality)	*Pimephales promelas* (fathead minnow)	FRY
81	±18	NE	COA	30-d	Not significantly toxic (18.8±5.05% mortality)	*Pimephales promelas* (fathead minnow)	FRY
81.5	±18.7	NE	COA	10-d ST	Not significantly toxic (19.2±12.4% mortality)	*Hexagenia limbata* (mayfly)	LAR
82.3	±18.1	SG	COA	30-d	Significantly toxic (402±40.6% increase in number)	*Lumbriculus variegatus* (oligochaete)	ADT
83.5	±61.3	*	COA		Low abundance (16±17.2 N/sq.m.)	Amphipoda	
84	±57.1	NE	COA		High diversity (0.797±0.067; Shannon Weaver Index)	Benthic species	
84.3	±5.51	NE	COA		High abundance (6616±538 N/sq.m.)	Benthic invertebrates	
85		SG	COA		Low diversity (0.82; Shannon Weaver Index)	Benthic species	
85		NE	COA		High abundance (5346 N/sq.m.)	Chironomidae	
85		NE	COA		High abundance (1179 N/sq.m.)	Amphipoda	
85		NE	COA		High abundance (6991 N/sq.m.)	Benthic invertebrates	
86	±2.83	NE	COA		High diversity (0.835±0.106; Shannon Weaver Index)	Benthic species	
86.8	±78.8	*	COA		Moderate abundance (582±142 N/sq.m.)	Benthic invertebrates	
86.9	±33.1	*	COA		Low abundance (0 N/sq.m.)	Chironomidae	
86.9	±33.1	NE	COA		High abundance (11106±2958 N/sq.m.)	Benthic species	
86.9	±46.8	SG	COA	48-h	Significantly toxic (100% mortality)	*Daphnia magna* (water flea)	JUV
86.9	±46.8	SG	COA	48-h	Significantly toxic (100% mortality)	*Daphnia magna* (water flea)	JUV
86.9	±46.8	SG	COA	10-d	Significantly toxic (98.3±3.1% mortality)	*Hexagenia limbata* (mayfly)	NYM
88		NE	COA		High abundance (645 N/0.044 sq.m.)	Benthic invertebrate	
88		NE	COA		High species (26 S/0.044 sq.m.)	Benthic species	
88.6			**PEL**				
90.3	±96.1	NE	COA		High abundance (14±7.94 N/sq.m.)	Gastropoda	
92.5	±110	NE	COA		High abundance (21306±2196 N/sq.m.)	Benthic invertebrates	
92.5	±6.36	SG	COA	48-h	Significantly toxic (53.4±23.6% mortality)	*Daphnia magna* (water flea)	JUV
93.8	±24.8	NE	COA		High abundance (16865±2383 N/sq.m.)	Benthic invertebrates	
94.5	±36	NE	COA		High diversity (2.87±0.035; Shannon Weaver Index)	Benthic species	
94.5	±36	NE	COA		High taxa (9.33±0.577 S/sq.m.)	Benthic species	
95	±64.2	*	COA		Low abundance (35.5±31 N/sq.m.)	Chironomidae	

sediment-associated chromium (mg·kg^{-1} dry weight) in freshwater systems.

TOC (%)	AVS (µmol·g^{-1})	Unionized Ammonia (mg·L^{-1})	Sand (%)	Silt (%)	Clay (%)	Area	Reference
1.26±0.469	1.28±1.7		24.9±8.36			Lake Pontchartrain, LA	EMAP 1991
1.33±0.662	0.753±0.332		23.1±7.62			Lake Pontchartrain, LA	EMAP 1991
10.3±5.56			39±28.9	47.8±26	10.8±5.85	Bay of Quinte, ON	Jaagumagi 1988; et al. 1989
3.48±0.46	5.23±0.813		32.7±20.9	9.85±0.35	55.2±22.9	Saginaw River, MI	Ingersoll et al. 1992a
2.6±2.09	5.3±3.87	0.065±0.047	60.4±35.4	9.8±7.2	29±26.3	Saginaw River, MI	Ingersoll et al. 1992a
1.57±0.85	0.543±0.325		25.7±8.22			Lake Pontchartrain, LA	EMAP 1991
2.05±2.66			65.3±39.4	23±29.5	8.33±11.8	Toronto Harbour, ON	Jaagumagi 1988; et al. 1989
2.5±2.77			60.7±33.9	27.5±24.1	9.17±9.35	Detroit River, MI	Jaagumagi 1988; et al. 1989
4.37			2.8	39.1	58.1	Lower Fox River & Green Bay, WI	Call et al. 1991
2.15±0.58						Keweenaw Waterway, MI	Malueg et al. 1984a
1.12±0.828			74±26.9	19.3±21.4	4±3.46	Detroit River, MI	Jaagumagi 1988; et al. 1989
2.5±2.77			60.7±33.9	27.5±24.1	9.17±9.35	Detroit River, MI	Jaagumagi 1988; et al. 1989
10.8±3.94			38.3±19.7	49.7±15.4	10.3±5.51	Bay of Quinte, ON	Jaagumagi 1988; et al. 1989
1.45±0.145	0.205±0.134		21.5±1.34			Mississippi River, LA, MS	EMAP 1991
2.18±2.72			69.5±38.9	22.5±29	5±5.66	Toronto Outer Harbour	Jaagumagi 1988; et al. 1989
1.7	2.6	0.072	10	9.5	74.3	Buffalo River, NY	Ingersoll et al. 1992a
2.62±2.38			48.3±46.5	35.3±30.1	13.3±16.4	Toronto Harbour, ON	Jaagumagi 1988; et al. 1989
8.2±3.1			41±16.6	48.3±16.9	9.75±6.99	St. Mary's River, ON	Jaagumagi 1988; et al. 1989
						United States	Davis 1987 (As cited in Dickson et a
1.33±0.95	0.327±0.196		24.5±8.64			Vermillion Bay, LA	EMAP 1991
2.6	5.6		27.3	9	55.6	Saginaw River, MI	Ingersoll et al. 1992a
0.316	0.13		5.5			Mississippi River, LA, MS	EMAP 1991
1.35	0.11		20.5			Mississippi River, LA, MS	EMAP 1991
1.95±0.353	5.4±0.424	0.095±0.03	14±9.83	7.55±3.46	76.7±13.7	Buffalo River, NY	Ingersoll et al. 1992a
3.11±0.436	5.13±1		22.6±15.6	9.58±0.39	64.9±17.4	Saginaw River, MI	Ingersoll et al. 1992a
6.7			9.2	63.6	27.2	Lower Fox River & Green Bay, WI	Call et al. 1991
						United States	USEPA 1977 (As cited in SAIC 199
1.24±0.979	1.08±1.77		28.8±18			Lake Pontchartrain, LA	EMAP 1991
2.55±2.56	0.635±0.637		26.4±7.99			Vermillion Bay, LA	EMAP 1991
2.55±2.56	0.635±0.637		26.4±7.99			Vermillion Bay, LA	EMAP 1991
3.02	0.26		26.4			Lake Pontchartrain, LA	EMAP 1991
1.8	4.2		7.5	3.7	92	Buffalo River, NY	Ingersoll et al. 1992a
13±1.41			27±2.83	58.5±2.12	13.5±0.70	Bay of Quinte, ON	Jaagumagi 1988; et al. 1989
4.24±5.87			68±32.9	24±27.9	5.25±5.97	Bay of Quinte, ON	Jaagumagi 1988; et al. 1989
4.53±2.41			33.4±28.7	35.3±16.7	31.3±18	Lower Fox River & Green Bay, WI	Call et al. 1991
4.53±2.41			33.4±28.7	35.3±16.7	31.3±18	Lower Fox River & Green Bay, WI	Call et al. 1991
4.53±2.41			33.4±28.7	35.3±16.7	31.3±18	Lower Fox River & Green Bay, WI	Call et al. 1991
4.53±2.41			33.4±28.7	35.3±16.7	31.3±18	Lower Fox River & Green Bay, WI	Call et al. 1991
4.35±2.42			35.4±29	33±15	31.6±18.8	Lower Fox River & Green Bay, WI	Call et al. 1991
4.54±2.52			36±28.4	35±17.4	29±16.8	Lower Fox River & Green Bay, WI	Call et al. 1991
7.89±5.97			50.8±30.6	39.3±25.1	8.33±5.96	Bay of Quinte, ON	Jaagumagi 1988; et al. 1989
0.672±0.485	1.83±2.48		29.4±26.8			Lake Pontchartrain, LA	EMAP 1991
3.05±0.09	4.75±1.2		12.8±4.58	9.77±0.29	76.7±4.56	Saginaw River, MI	Ingersoll et al. 1992a
5			46	38	7	Toronto Harbour, ON	Jaagumagi 1988; et al. 1989
8.8			10	75	14	Penetang Harbour, Lake Huron, ON	Jaagumagi 1988; et al. 1989
8.8			10	75	14	Penetang Harbour, Lake Huron, ON	Jaagumagi 1988; et al. 1989
8.8			10	75	14	Penetang Harbour, Lake Huron, ON	Jaagumagi 1988; et al. 1989
4.45±2.48	3.01±2.05		35.8±5.35			Vermillion Bay, LA	EMAP 1991
5.34±5.72			62.8±31.5	29.8±26.1	5.75±5.68	Bay of Quinte, ON	Jaagumagi 1988; et al. 1989
8±2.83						St. Marys River, ON	Pope 1990
8±2.83						St. Marys River, ON	Pope 1990
0.94±0.23						Little Grizzly Creek System, CA	Malueg et al. 1984b
0.94±0.23						Little Grizzly Creek System, CA	Malueg et al. 1984b
0.94±0.23						Little Grizzly Creek System, CA	Malueg et al. 1984b
2.7	4.46		39.5			Vermillion Bay, LA	EMAP 1991
2.7	4.46		39.5			Vermillion Bay, LA	EMAP 1991
2.79±3.16			77±16.5	18±14	3±1.73	Bay of Quinte, ON	Jaagumagi 1988; et al. 1989
3.85±1.77			58±53.7	30.5±37.5	11.5±14.8	Toronto Harbour, ON	Jaagumagi 1988; et al. 1989
2.45±0.92						Keweenaw Waterway, MI	Malueg et al. 1984a
1.85±0.26	6.28±4.01		7.47±2.34	6.67±2.46	83.3±7.88	Buffalo River, NY	Ingersoll et al. 1992a
8.75±2.93						St. Marys River, ON	Pope 1990
8.75±2.93						St. Marys River, ON	Pope 1990
3.24±2.3			41.3±40.6	40.8±26.8	15.5±14.1	Toronto Harbour, ON	Jaagumagi 1988; et al. 1989

Appendix Ia. Summary of the available biological effects and related physicochemical data for

Chromium ($mg \cdot kg^{-1}$)		Hit	Analysis Type	Test Type	Endpoint Measured	Species	Life Stage
95	±64.2	*	COA		Low abundance (0 N/sq.m.)	Amphipoda	
95	±64.2	*	COA		Low species richness (0.918±0.417; SRUs)	Benthic species	
95.5	±75.7	NE	COA		High species (18.5±0.707 S/0.044 sq.m)	Benthic species	
99.7	±76.2	*	COA		Low abundance (42.1±71.4 N/sq.m.)	Chironomidae	
99.7	±76.2	*	COA		Low abundance (33.3±45.6 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
100		*	SBA		WDNR Interim Criteria for In-Water Disposal of Dredged Sediments		
100		*	SLCA		Sediment Quality Criteria - Toxic Effects Threshold		
100		*	SSBA		Sediment Quality Guidelines		
100		*	SBA		USEPA Region VI Proposed Guidelines for Sediment Disposal		
102	±23.3	*	COA		Low abundance (110±69.4 N/sq.m.)	Benthic Invertebrates	
102	±23.3	*	COA		Low taxa (3.8±1.92 S)	Benthic species	
102	±23.3	*	COA		Low diversity (0.38±0.25 Shannon Weaver Index)	Benthic species	
102	±23.3	*	COA		Low abundance (0 N/sq.m.)	Emphemeroptera	
102	±66.7	NE	COA		High species richness (2.92±0.216; SRUs)	Benthic species	
102	±86.6	SG	COA		Low abundance (20.8±27 N/sq.m.)	Gastropoda	
102	±99.3	SG	COA		Low abundance (6050±2566 N/sq.m.)	Benthic invertebrates	
103	±71.6	*	COA		Low abundance (0.571±1.51 N/sq.m.)	Gastropoda	
105	±84.7	*	COA		Low abundance (154±102 N/sq.m.)	Chironomidae	
106	±74.4	SG	COA		Low abundance (9±6.75 N/sq.m.)	Amphipoda	
106	±91.3	NE	COA		Not significantly toxic (5.5±2.56% deformities)	Chironomidae	
109	±19.6	*	COA	48-h	Significantly toxic (97.1±2.85% mortality)	*Daphnia magna* (water flea)	JUV
110		*	SLCA		OMOE Provincial SQGs - Severe effect level		
112	±67.9	*	COA		Moderate diversity (2.14±0.351; Shannon Weaver Index)	Benthic species	
113		NC	COA	14-d FT	Toxic (36.2% mortality)	*Hyalella azteca* (amphipod)	LAR
113		NE	COA	14-d FT	Not significantly toxic (4.2% sexual maturity)	*Hyalella azteca* (amphipod)	LAR
120		*	SBA		OMOE Unrestricted Land Use Criteria		
120		*	SBA		OMOE Restricted Land Use Criteria		
121	±112	SG	COA		Low abundance (65±41.7 N/sq.m.)	Chironomidae	
129	±105	NE	COA	28-d FT	Not significantly toxic (99.7±0.716% sexual maturity)	*Hyalella azteca* (amphipod)	LAR
129	±105	NE	COA	10-d ST	Not significantly toxic (5.36±3% mortality)	*Chironomus tentans* (midge)	LAR
133	±121	NE	COA	14-d FT	Not toxic (20.9±11.5% mortality)	*Hyalella azteca* (amphipod)	LAR
133	±121	SG	COA	14-d FT	Significantly toxic (0.35±0.7% sexual maturity)	*Hyalella azteca* (amphipod)	LAR
135	±159	SG	COA		Low abundance (1494±533 N/sq.m.)	Benthic invertebrates	
136	±91.2	NE	COA		High abundance (24.5±13.4 N/sq.m.)	Ephemeroptera, Plecoptera, Tricoptera	
142	±224	*	COA		Low abundance (684±529 N/sq.m.)	Benthic invertebrates	
143	±115	*	COA	28-d FT	Significantly toxic (20±5.1% mortality)	*Hyalella azteca* (amphipod)	LAR
143	±259	NC	COA		Low abundance (0 N/sq.m.)	Gastropoda	
150		*	COA	48-h	Significantly toxic (22% mortality)	*Chironomus tentans* (midge)	LAR
158	±211	NE	COA		High abundance (16.3±2.31 N/sq.m.)	Gastropoda	
164	±241	*	COA		Low species richness (1.07±0.192; SRUs)	Benthic species	
165	±240	NE	COA		High diversity (2.23±0.387; Shannon Weaver Index)	Benthic species	
165	±49.5	*	COA		Low species richness (0.69±0.0141; SRUs)	Benthic species	
170	±280	*	COA		Moderate abundance (6908±2982 N/sq.m.)	Benthic invertebrates	
170	±42.4	NE	COA		High abundance (5177±779 N/sq.m.)	Benthic invertebrates	
178	±257	*	COA		Low abundance (160±316 N/sq.m.)	Amphipoda	
179	±256	*	COA		Low abundance (28.3±20.1 N/sq.m.)	Chironomidae	
180	±197	NE	COA		Not significantly toxic (3±4.24% deformities)	Chironomidae	
200		*	COA		Low diversity (1.39; Shannon Weaver Index)	Benthic species	
282	±321	NE	COA		High taxa (11.3±0.577 S)	Benthic species	
287	±262	NE	COA		High diversity (0.833±0.057; Shannon Weaver Index)	Benthic species	
309	±529	*	COA		Low taxa (4.14±2.54 S)	Ephemeroptera, Plecoptera, Tricoptera	
309	±529	*	COA		Low taxa (1.14±1.46 S)	Ephemeroptera, Plecoptera, Tricoptera	
315	±236	NE	COA	48-h	Not significantly toxic (1.68±2.04% mortality)	*Daphnia magna* (water flea)	JUV
319		*	COA	14-d FT	Significantly toxic (90% mortality)	*Hyalella azteca* (amphipod)	LAR
319		*	COA	28-d FT	Significantly toxic (90% mortality)	*Hyalella azteca* (amphipod)	LAR
319		*	COA	14-d FT	Significantly toxic (85.5% mortality)	*Chironomus riparius* (midge)	24 h
319		*	COA	28-d FT	Significantly toxic (37.5% sexual maturity)	*Hyalella azteca* (amphipod)	LAR
379	±300	NE	COA		High abundance (1401±208 N/sq.m.)	Benthic Invertebrates	
426	±344	NE	COA	48-h	Not significantly toxic (3.33±2.8% mortality)	*Daphnia magna* (water flea)	JUV
426	±344	NE	COA	10-d	Not significantly toxic (10±3.61% mortality)	*Hexagenia limbata* (mayfly)	NYM
473	±446	SG	COA		Low abundance (626±201 N/sq.m.)	Benthic invertebrates	
570	±361	*	COA		Low taxa (7.67±1.15 S)	Benthic species	
705	±389	SG	COA		Low diversity (0.54±0.099; Shannon Weaver Index)	Benthic species	
797	±728	NE	COA		High abundance (22±6 N/0.0225 sq.m.)	Chironomidae	

sediment-associated chromium (mg·kg^{-1} dry weight) in freshwater systems.

TOC (%)	AVS (µmol·g^{-1})	Unionized Ammonia (mg·L^{-1})	Sand (%)	Silt (%)	Clay (%)	Area	Reference
3.24±2.3			41.3±40.6	40.8±26.8	15.5±14.1	Toronto Harbour, ON	Jaagumagi 1988; et al. 1989
3.24±2.3			41.3±40.6	40.8±26.8	15.5±14.1	Toronto Harbour, ON	Jaagumagi 1988; et al. 1989
0.43±0.345	0.4±0.226		33.3±36.7			Lake Pontchartrain, LA	EMAP 1991
3.63±3.18			61.9±30.3	26±21	9.86±8.57	Detroit River, MI	Jaagumagi 1988; et al. 1989
3.63±3.18			61.9±30.3	26±21	9.86±8.57	Detroit River, MI	Jaagumagi 1988; et al. 1989
						Great Lakes	Sullivan et al. 1985
						St. Lawrence River	EC/MINVEQ 1992
1						Canada	Hart et al. 1988
						United States	Pavlou and Weston 1983
2.2±0.64						Keweenaw Waterway, MI	Malueg et al. 1984a
2.2±0.64						Keweenaw Waterway, MI	Malueg et al. 1984a
2.2±0.64						Keweenaw Waterway, MI	Malueg et al. 1984a
2.2±0.64						Keweenaw Waterway, MI	Malueg et al. 1984a
8.49±6.27			50±34.6	40.3±28.2	8.25±6.75	Bay of Quinte, ON	Jaagumagi 1988; et al. 1989
2.5±2.47	24.7±51.4		31±28	6.43±2.94	58.1±26.6	Buffalo River, NY	Ingersoll et al. 1992a
2.68±2.82	29.7±58.2		37.8±28.3	5.94±3.23	52.1±27.5	Buffalo River, NY	Ingersoll et al. 1992a
3.86±2.94			55±27.9	31.3±20.1	10.7±7.93	Detroit River, MI	Jaagumagi 1988; et al. 1989
2.49±2.47	23.7±51.7		28.1±29	5.97±3.01	62.1±28.8	Buffalo River, NY	Ingersoll et al. 1992a
4.79±3.02			50.8±29	33.6±20.3	13±8.19	Detroit River, MI	Jaagumagi 1988; et al. 1989
2.55±2.63	27±54.4		32.4±29.6	5.81±2.9	57.7±29.1	Buffalo River, NY	Ingersoll et al. 1992a
2.38±0.54						Keweenaw Waterway, MI	Malueg et al. 1984a
1						Ontario	Persaud et al. 1992
4.14±3.44			60.5±32.1	27.3±23	10±9.06	Detroit River, MI	Jaagumagi 1988; et al. 1989
2	5.8	0.052	34.6	7	54	Buffalo River, NY	Ingersoll et al. 1992a
2	5.8	0.052	34.6	7	54	Buffalo River, NY	Ingersoll et al. 1992a
						Ontario	Fitchko 1989
						Ontario	Fitchko 1989
2.3±0.84	6.4±5.37		37±29.7	8±1.91	50.7±30	Saginaw River, MI	Ingersoll et al. 1992a
3.3±3.14	36±69.9	0.076±0.024	22.9±15.3	7.84±1.98	65.1±16.8	Buffalo River, NY	Ingersoll et al. 1992a
3.3±3.14	36±69.9	0.214±0.203	22.9±15.3	7.84±1.98	65.1±16.8	Buffalo River, NY	Ingersoll et al. 1992a
3.63±3.52	43.6±72.3	0.08±0.02	20±15.9	8.05±2.22	67.9±18	Buffalo River, NY	Ingersoll et al. 1992a
3.63±3.52	43.6±78.3	0.08±0.02	20±15.9	8.05±2.22	67.9±18	Buffalo River, NY	Ingersoll et al. 1992a
2.3±1.41	7.6±7.17		61.5±12.2	7.17±2.11	26.1±11.2	Saginaw River, MI	Ingersoll et al. 1992a
6.28±8.09			61.5±46	30.5±37.5	7±8.49	Bay of Quinte, ON	Jaagumagi 1988; et al. 1989
2.1±3.2			55.5±57.3	35.5±47.4	7.5±9.19	Humber Bay, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
3.7±3.47	44.4±77.7	0.077±0.028	26.2±15.5	7.43±2.02	62.8±18.4	Buffalo River, NY	Ingersoll et al. 1992a
1.77±2.09			41±35.8	45.5±28.2	11±8.17	Humber Bay, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
0.18						Trinity River, TX	Dickson et al. 1989
2.72±2.83			33.5±26.2	54±21.2	11.5±3.54	Humber Bay, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
2.31±2.27			39.1±32	47.6±25.5	11.1±7.04	Humber Bay, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
2.31±2.27			31±24.1	54.1±19	12.6±6.19	Humber Bay, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
6.15±2.19			56±7.07	30.5±4.95	11±1.41	Detroit River, MI	Jaagumagi 1988; et al. 1989
2.07±2.19			30±26.4	54.2±20.7	13±7.31	Humber Bay, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
6.15±2.19			56±7.07	30.5±4.95	11±1.41	Detroit River, MI	Jaagumagi 1988; et al. 1989
2.37±2.48			42.5±36.7	44.3±28.8	11.5±8.26	Humber Bay, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
2.34±2.45			37.3±34.1	48.9±27.3	11.4±7.55	Humber Bay, Lake Ontario, ON	Jaagumagi 1988; et al. 1989
1.55±0.78	8.5±9.9	0.043±0.03	68.6±0.91	5.95±0.07	19.6±0.28	Saginaw River, MI	Ingersoll et al. 1992a
4.6			61	27	10	Detroit River, MI	Jaagumagi 1988; et al. 1989
22.7±6.59						Phillips Chain of Lakes, WI	Malueg et al. 1984b
21.7±5.76						Phillips Chain of Lakes, WI	Malueg et al. 1984b
2.75±2.71						River Adige, Italy	Duzzin et al. 1988
2.75±2.71						River Adige, Italy	Duzzin et al. 1988
20.4±5.82						Phillips Chain of Lakes, WI	Malueg et al. 1984b
2.1	15.5	0.064	69.2	6	19.8	Saginaw River, MI	Ingersoll et al. 1992a
2.1	15.5	0.064	69.2	6	19.8	Saginaw River, MI	Ingersoll et al. 1992a
2.1	15.5	0.063	69.2	6	19.8	Saginaw River, MI	Ingersoll et al. 1992a
2.1	15.5	0.064	69.2	6	19.8	Saginaw River, MI	Ingersoll et al. 1992a
19.2±7.13						Phillips Chain of Lakes, WI	Malueg et al. 1984b
19.3±5.78						Phillips Chain of Lakes, WI	Malueg et al. 1984b
19.3±5.78						Phillips Chain of Lakes, WI	Malueg et al. 1984b
19.5±5.75						Phillips Chain of Lakes, WI	Malueg et al. 1984b
15.9±2.36						Phillips Chain of Lakes, WI	Malueg et al. 1984b
14.6±0.495						Phillips Chain of Lakes, WI	Malueg et al. 1984b
						Palestine Lake, IN	Wentsel et al. 1977a

Appendix Ia. Summary of the available biological effects and related physicochemical data for

Chromium ($mg \cdot kg^{-1}$)		Hit	Analysis Type	Test Type	Endpoint Measured	Species	Life Stage
980		*	COA	48-h	Significantly toxic (16.7% mortality)	*Daphnia magna* (water flea)	JUV
1014	±821	NC	COA		Low abundance (3357±2112 N/sq.m.)	Benthic invertebrates	
1156	±505	NE	COA	5-d	Not significantly toxic (58.8±1.98% avoidance)	*Chironomus tentans* (midge)	LAR
1412		NE	COA		High abundance (493917 N/sq.m.)	Benthic invertebrates	
1721	±375	SG	COA	5-d	Significantly toxic (86.53±11.8% avoidance)	*Chironomus tentans* (midge)	LAR
1954	±215	*	COA		Low abundance (0.2 N/0.0225 sq.m.)	Chironomidae	

sediment-associated chromium ($mg \cdot kg^{-1}$ dry weight) in freshwater systems.

TOC (%)	AVS ($\mu mol \cdot g^{-1}$)	Unionized Ammonia ($mg \cdot L^{-1}$)	Sand (%)	Silt (%)	Clay (%)	Area	Reference
14.2						Phillips Chain of Lakes, WI	Malueg et al. 1984b
9.2±2.29	41.5±21.4		23.9±14.3	5.9±0.636	66.1±15.1	Indiana Harbor, IN	Ingersoll et al 1992a
						Palestine Lake, IN	Wentsel et al. 1977b
12.3	31.7		14.2	9.9	72.1	Indiana Harbor, IN	Ingersoll et al. 1992a
						Palestine Lake, IN	Wentsel et al. 1977b
						Palestine Lake, IN	Wentsel et al 1977a

Appendix Ib. Summary of the available biological effects and related physicochemical data for

Chromium (mg·kg^{-1})		Hit	Analysis Type	Test Type	Endpoint Measured	Species	Life Stage	TOC (%)
0.6		NC	COA	20-d	Significantly toxic (-0.004 mg/day growth)	*Hyalella azteca* (amphipod)	JUV	0.185
1.32	±1.34	NE	COA	10-d	Not toxic (7.2±2.77% mortality)	*Palaemonetes pugio* (grass shrimp)	ADT	
1.32	±1.34	NE	COA	10-d	Not toxic (10.6±5.18% mortality)	*Nereis virens* (sandworm)	ADT	
1.32	±1.34	NE	COA	10-d	Not toxic (0.2±0.447% mortality)	*Mercenaria mercenaria* (hard clams)	ADT	
2.46		NE	COA		Not toxic (species richness or abundance)	Benthic species		0.12
2.74	±1.82	NE	COA		Not sig. toxic (mean ET50;burrowing time 0.704±0.17h)	*Mya arenaria* (clam)	1-2 cm	
4		NC	COA		Significantly toxic (10% mortality)	*Mysidopsis bahia* (mysid shrimp)		0.009
4.2	±2.43	NE	COA		High species richness (14.9±2.04; SRUs)	Benthic species		
4.5	±2.12	NC	COA	10-d	Significantly toxic (15.6% mortality)	*Ampelisca abdita* (amphipod)		0.438±0.442
5.6		NC	COA	10-d	Significantly toxic (18% mortality)	*Ampelisca abdita* (amphipod)		0.028
6		NE	COA	48-h	Not significantly toxic (0% mortality)	*Arbacia punctulata* (sea urchin)		0.03
6	±2.94	NE	COA		Not significantly toxic (0.75±1.5% mortality)	*Mysidopsis bahia* (mysid shrimp)		0.54±0.385
6.25		*	EqPA		Chronic Marine EqP Threshold	Aquatic biota		1
6.4		SG	COA		Low species richness (5.16; SRUs)	Benthic species		
6.64	±2.47	NE	COA		High diversity (4.15±0.59; SDUs)	Benthic species		
6.67	±5.03	NE	COA		Not significantly toxic (15±13.2% mortality)	Sheepshead minnow	ADT	0.05±0.0242
6.8		SG	COA		Low diversity (1.16; SDUs)	Benthic species		
7.5	±3.54	NE	COA	10-d	Not significantly toxic (3.3±3.25% mortality)	*Ampelisca abdita* (amphipod)		0.643±0.455
7.63	±8.94	NE	COA	10-d	Not significantly toxic (15.3±11.0% mortality)	*Hyalella azteca* (amphipod)	JUV	0.648±0.641
8		SG	COA		Significantly toxic (mean ET50; burrowing time 1.4 h)	*Mya arenaria* (clam)	1-2 cm	
8.02	±2.45	SG	COA		Moderate species richness (9.05±1.33; SRUs)	Benthic species		
8.37	±2.30	NE	COA	10-d	Not toxic (2±1% mortality)	*Palaemonetes pugio* (grass shrimp)	ADT	
8.37	±2.30	NE	COA	10-d	Not toxic (9.6±4.22% mortality)	*Nereis virens* (sandworm)	ADT	
8.37	±2.30	NE	COA	10-d	Not toxic (0% mortality)	*Mercenaria mercenaria* (hard clams)	ADT	
8.43	±2.49	NE	COA	10-d	Not toxic (4.8±2.28% mortality)	*Palaemonetes pugio* (grass shrimp)	ADT	
8.43	±2.49	NE	COA	10-d	Not toxic (15.8±4.21% mortality)	*Nereis virens* (sandworm)	ADT	
8.43	±2.49	NE	COA	10-d	Not toxic (1.2±0.447% mortality)	*Mercenaria mercenaria* (hard clams)	ADT	
9.32	±3.27	SG	COA		Moderate diversity (2.3±0.2; SDUs)	Benthic species		
9.5	±9.19	NC	COA	10-d	Significantly toxic (11.6±3.39% mortality)	*Ampelisca abdita* (amphipod)		0.111±0.045
10		NE	COA		High density (24 N/0.1 sq.m.)	Echinodermata	VAR	
10.6	±10.6	NC	COA	20-d	Significantly toxic (55±19.5% mortality)	*Streblospio benedicti* (polychaete worm)		0.349±0.11
11.8	±3.7	NE	COA	48-h	Least toxic (15.1±3.1% abnormality)	Oyster	LAR	
11.8	±4.61	NE	COA	48-h	Not significantly toxic (86.6±4.25% normal development)	*Arbacia punctulata* (sea urchin)		0.181±0.162
12		NE	COA	10-d	Not significantly toxic (11.3% mortality)	*Leptocheirus plumulosus* (amphipod)	JUV	
12	±9.7	NE	COA	10-d	Not significantly toxic (7.2±1.92% mortality)	*Lepidactylus dytiscus* (amphipod)		0.56±0.47
12	±9.7	NE	COA	10-d	Not significantly toxic (99.6±0.55% reburial)	*Lepidactylus dytiscus* (amphipod)		0.56±0.47
12	±9.7	NE	COA	20-d	Not significantly toxic (<10% emergence)	*Lepidactylus dytiscus* (amphipod)		0.56±0.47
12	±9.7	NE	COA	20-d	Not significantly toxic (<10% emergence)	*Hyalella azteca* (amphipod)	JUV	0.56±0.47
13.2	±14	NE	COA	10-d	Not significantly toxic (27.5±10.6% mortality)	*Mysidopsis bahia* (mysid)		0.237±0.311
13.3	±11.5	NE	COA	96-h	Not significantly toxic (9.78±5.33% mortality)	*Mysidopsis bahia* (mysid)		0.24±0.247
13.5	±12.3	NE	COA	96-h	Not significantly toxic (4±2.83% mortality)	*Menidia beryllina* (silverside)		0.243±0.264
13.5	±6.11	SG	COA	10-d	Significantly toxic (28.3±6.11% mortality)	*Mysidopsis bahia* (mysid)		0.246±0.051
14	±14.1	NC	COA	10-d	Significantly toxic (46±49.3% mortality)	*Ampelisca abdita* (amphipod)		0.22±0.298
14.2	±11.9	NE	COA	10-d	Not significantly toxic (11.6±4.33% mortality)	*Ampelisca abdita* (amphipod)		0.267±0.25
14.7	±12.5	NE	COA	10-d	Not significantly toxic (5.4±3.6% mortality)	Amphipod	ADT	0.129±0.054
15	±4.24	NE	COA	20-d	Not significantly toxic (25.4±17.1% mortality)	*Leptocheirus plumulosus* (amphipod)	JUV	
15.3	±2.0	NE	COA	96-h	Not toxic (0-10% mortality)	*Palaemonetes pugio* (shrimp)		
15.4	±8.41	NE	COA		High abundance (155±49.5 N/0.00203 sq.m.)	Oligochaeta		1.2±1.27
15.8	±11.9	NE	COA	48-h	Not significantly toxic (91.3±2.91% normal development)	*Arbacia punctulata* (sea urchin)		0.299±0.251
16.1	±8.17	NE	COA	1-h	Least toxic (83±4.26% fertilization)	*Arbacia punctulata* (sea urchin)	GAM	
16.2	±8.1	NE	COA	10-d	Least toxic (12.5±4.5% mortality)	*Rhepoxynius abronius* (amphipod)	ADT	
17.4	±12.2	*	COA	48-h	Significantly toxic (25.4±7.66% mortality)	*Arbacia punctulata* (sea urchin)		0.344±0.242
17.6	±13	NE	COA		High abundance (154±30.2 N/0.00203 sq.m.)	Benthic invertebrates		0.47±0.27
17.7		NE	COA	10-d	Not significantly toxic (1% mortality)	*Streblospio benedicti* (polychaete worm)		1.38
17.7		NC	COA	20-d	Significantly toxic (-0.003 mg/day growth)	*Lepidactylus dytiscus* (amphipod)		1.38
17.7	±7.3	SG	COA	48-h	Moderately toxic (23±2.3% abnormality)	Oyster	LAR	
17.7	±7.3	SG	COA	10-d	Moderately toxic (26±5.2% mortality)	*Rhepoxynius abronius* (amphipod)	ADT	
17.9	±0.173	NE	COA	10-d	Not toxic (2.4±1.52% mortality)	*Palaemonetes pugio* (grass shrimp)	ADT	
17.9	±0.173	NE	COA	10-d	Not toxic (0% mortality)	*Mercenaria mercenaria* (hard clams)	ADT	
17.9	±0.173	NE	COA	10-d	Not toxic (2.8±2.17% mortality)	*Nereis virens* (sandworm)	ADT	
18	±4.93	NE	COA	10-d	Not significantly toxic (8±2% mortality)	*Crassostrea virginica* (oyster)	ADT	6.73±2.04
18	±4.93	NE	COA	10-d	Not significantly toxic (22±5.29% mortality)	*Arenicola cristata* (lugworm)	ADT	6.73±2.04
18	±4.93	NE	COA	10-d	Not significantly toxic (3.3±3.06% mortality)	*Penaeus duorarum* (pink shrimp)	ADT	6.73±2.04
19	±12.7	NE	COA		High density (299±40.3 N/0.1 sq.m.)	Arthropoda	VAR	

sediment-associated chromium ($mg \cdot kg^{-1}$ dry weight) in marine and estuarine systems.

AVS ($\mu mol \cdot g^{-1}$)	Total Sulphide ($mg \cdot kg^{-1}$)	TFS ($mg \cdot L^{-1}$)	Unionized Ammonia ($mg \cdot L^{-1}$)	Sand (%)	Silt (%)	Clay (%)	Area	Reference
0.5				100			Chesapeake Bay, VA, MD	Hall et al. 1992
				85.7±13.7	5.33±9.24	5.33±7.51	Matagorda Ship Channel, TX	Espey, Huston & Assoc. 1985a
				85.7±13.7	5.33±9.24	5.33±7.51	Matagorda Ship Channel, TX	Espey, Huston & Assoc. 1985a
				85.7±13.7	5.33±9.24	5.33±7.51	Matagorda Ship Channel, TX	Espey, Huston & Assoc. 1985a
				0.5		0.5	Georgetown Harbor, SC	Van Dolah et al. 1984
							Chesapeake Bay,MD,VI; Delaware Bay,DE	Phelps 1990
0.004				94.8			Mobile Bay, AL	EMAP 1991
				94.9±3.91			Charleston Harbor, SC	Winn et al. 1989
0.421±0.593				40.5±24.2			Apalachee Bay, FL	EMAP 1991
9 µg/g							Savannah River Entrance Channel, GA	Windom 1995
19 µg/g							Savannah River Entrance Channel, GA	Windom 1995
0.405±0.417				51.6±25.1			Apalachee Bay, FL	EMAP 1991
							United States	Bolton et al. 1985
				98.1			Charleston Harbor, SC	Winn et al. 1989
				97.1±2.34			Charleston Harbor, SC	Winn et al. 1989
							Houston Ship Channel, TX	Crocker et al. 1991
				98.1			Charleston Harbor, SC	Winn et al. 1989
0.39±0.41				62.7±28.3			Apalachee Bay, FL	EMAP 1991
1.74±1.39				87.4±16.6	9.77±13.7	2.8±3.02	Chesapeake Bay, VA, MD	Hall et al. 1992
							Chesapeake Bay,MD,VI; Delaware Bay,DE	Phelps 1990
				97.7±1.07			Charleston Harbor, SC	Winn et al. 1989
				24.7±21	41.4±5.4	34±16.5	Freeport Harbor, TX	Espey, Huston & Assoc. 1985b
				24.7±21	41.4±5.4	34±16.5	Freeport Harbor, TX	Espey, Huston & Assoc. 1985b
				24.7±21	41.4±5.4	34±16.5	Freeport Harbor, TX	Espey, Huston & Assoc. 1985b
							Sabine-Neches Waterway, TX	Espey, Huston & Assoc. 1983a
							Sabine-Neches Waterway, TX	Espey, Huston & Assoc. 1983a
							Sabine-Neches Waterway, TX	Espey, Huston & Assoc. 1983a
				97.5±0.67			Charleston Harbor, SC	Winn et al. 1989
0.03				87.9±3.46			Mississippi Sound, MS	EMAP 1991
							Massachusetts Bay, MA	Gilbert et al. 1976
1.07±0.66				94.7±3.73	3.25±2.57	2.03±1.38	Chesapeake Bay, VA, MD	Hall et al. 1992
							Commencement Bay, WA	Tetra Tech 1985
182±208 µg/g							Brunswick Harbor Entrance, GA	Windom 1995
				4.9	44.4	50.8	Curtis Creek, Baltimore, MD	McGee et al. 1993
1.5±1.13				89.5±12.1	7.68±10.2	2.82±2.14	Chesapeake Bay, VA, MD	Hall et al. 1992
1.5±1.13				89.5±12.1	7.68±10.2	2.82±2.14	Chesapeake Bay, VA, MD	Hall et al. 1992
1.5±1.13				89.5±12.1	7.68±10.2	2.82±2.14	Chesapeake Bay, VA, MD	Hall et al. 1992
1.5±1.13				89.5±12.1	7.68±10.2	2.82±2.14	Chesapeake Bay, VA, MD	Hall et al. 1992
74.6±85.5 µg/g							Savannah River Entrance Channel, GA	Windom 1995
91.9±73.4 µg/g							Savannah River Entrance Channel, GA	Windom 1995
90.5±78.4 µg/g							Savannah River Entrance Channel, GA	Windom 1995
127±24 µg/g							Savannah River Entrance Channel, GA	Windom 1995
3.84±5.43				59.6±49.8			Mobile Bay, AL	EMAP 1991
102±71.1 µg/g							Savannah River Entrance Channel, GA	Windom 1995
							Houston Ship Channel, TX	Crocker et al. 1991
				5.8±1.27	43.6±1.13	50.7±0.212	Curtis Creek, Baltimore, MD	McGee et al. 1993
							Duwamish River, WA	Lee and Mariani 1977
4.27±3.85				48.6±14.1	30.2±5.06	21.2±19.2	Galveston Bay, TX	Carr 1992
114±68.4 µg/g							Savannah River Entrance Channel, GA	Windom 1995
0.131±0.169							Tampa Bay, FL	Long 1993
							Commencement Bay, WA	Tetra Tech 1985
129.5±59.3 µg/g							Savannah River Entrance Channel, GA	Windom 1995
9.07±2.94				32.8±5.56	35.6±13	31.7±9.67	Galveston Bay, TX	Carr 1992
3.25				68.6	25.4	6	Chesapeake Bay, VA, MD	Hall et al. 1992
3.25				68.6	25.4	6	Chesapeake Bay, VA, MD	Hall et al. 1992
							Commencement Bay, WA	Tetra Tech 1985
							Commencement Bay, WA	Tetra Tech 1985
				25.3±0.917	25.3±2.06	49.4±2.15	Galveston Harbor, TX	Grieco 1984
				25.3±0.917	25.3±2.06	49.4±2.15	Galveston Harbor, TX	Grieco 1984
				25.3±0.917	25.3±2.06	49.4±2.15	Galveston Harbor, TX	Grieco 1984
							Gulfport Mississippi Channel, MS	Parrish 1987b
							Gulfport Mississippi Channel, MS	Parrish 1987b
							Gulfport Mississippi Channel, MS	Parrish 1987b
							Massachusetts Bay, MA	Gilbert et al. 1976

Appendix Ib. Summary of the available biological effects and related physicochemical data for

Chromium (mg·kg^{-1})		Hit	Analysis Type	Test Type	Endpoint Measured	Species	Life Stage	TOC (%)
19.7	±11.3	SG	COA	10-d	Highly toxic (78.5±19.5% mortality)	*Rhepoxynius abronius* (amphipod)	ADT	
19.7	±4.45	NE	COA		High abundance (156±22.9 N/0.00203 sq.m.)	Polychaeta		0.916±0.661
19.9	±0.4	NE	COA	96-h	Not toxic (0% mortality)	*Palaemonetes pugio* (shrimp)		
20	±1.97	NE	COA		High abundance (27.3±10 N/0.00203 sq.m.)	Mollusca		1.64±0.4
20	±5	NE	COA	10-d	Not toxic (6±2.35% mortality)	*Palaemonetes pugio* (grass shrimp)	ADT	
20	±5	NE	COA	10-d	Not toxic (2.6±2.97% mortality)	*Nereis virens* (sandworm)	ADT	
20	±5	NE	COA	10-d	Not toxic (1.8±1.79% mortality)	*Mercenaria mercenaria* (hard clams)	ADT	
20.5	±14.3	NE	COA	7-d	Not significantly toxic (7.5±5% mortality)	Sheepshead minnow	ADT	0.137±0.056
20.8	±7.97	NE	COA	10-d	Not significantly toxic (15±7.21% mortality)	*Hyalella azteca* (amphipod)	JUV	
21.2	±7.65	NE	COA	10-d	Not toxic (6.67±4.51% mortality)	*Corophium volutator* (amphipod)	ADT/JUV	0.24±0.18
21.2	±7.65	NE	COA	10-d	Not toxic (2.33±1.53% mortality)	*Rhepoxynius abronius* (amphipod)	ADT/JUV	0.24±0.18
21.2	±7.65	NE	COA	10-d	Not toxic (100% reburial)	*Rhepoxynius abronius* (amphipod)	ADT/JUV	0.24±0.18
21.5	±3.51	NE	COA		High species richness (24.5±3.7 S/0.00203 sq.m.)	Benthic species		1.36±0.66
21.5	±3.51	NE	COA		High abundance (359±92.8 N/0.00203 sq.m.)	Benthic species		1.36±0.66
21.6	±8.67	NE	COA		High abundance (16.2±6.19 N/0.00203 sq.m.)	Copepoda		0.844±0.524
22	±13.5	NE	COA		Low species (11.2±1.94 S/0.00203 sq.m.)	Benthic species		0.642±0.356
22.2	±9	SG	COA	48-h	Highly toxic (44.5±19% abnormality)	Oyster	LAR	
22.3	±10.7	NE	COA		High species richness (102±4.73 S/0.1 sq.m.)	Benthic species	VAR	
23.4	±10.3	SG	COA		Moderate abundance (58.3±16.2 N/0.00203 sq.m.)	Oligochaeta		0.632±0.274
23.5	±3.54	NG	COA	10-d	Significantly toxic (22.7±0.778% mortality)	Amphipod	ADT	0.248±0.132
23.7	±15.6	NE	COA	48-h	Not toxic (98.1±1.79% normal development)	*Arbacia punctulata* (sea urchin)	EMB	0.748±0.476
23.7	±22.3	NE	COA	10-d	Not significantly toxic (4.7±4.29% mortality)	Amphipod	ADT	0.144±0.123
24.6	±29.9	SG	COA	1-h	Moderately toxic (44.5±17.8% fertilization)	*Arbacia punctulata* (sea urchin)	GAM	
24.8	±22.5	NE	COA	10-d	Not significantly toxic (9±1.73% mortality)	*Nereis virens* (polychaetes)	ADT	
24.8	±22.5	NE	COA	10-d	Not significantly toxic (13±3.61% mortality)	*Penaeus duorarum* (pink shrimp)	ADT	
24.8	±22.5	NE	COA	10-d	Not significantly toxic (11.3±3.21% mortality)	*Crassostrea virginica* (oyster)	ADT	
25	±14.4	NE	COA	1-h	Not toxic (92.5±6.9% fertilization)	*Arbacia punctulata* (sea urchin)	EMB	0.77±0.448
26		NE	COA	96-h	Not significantly toxic (4% mortality)	*Citharichthys stigmaeus* (sanddab)		
26		NE	COA	10-d	Not significantly toxic (14% mortality)	*Acanthomysis sculpta* (mysid)	ADT	
26		NE	COA	20-d	Not significantly toxic (13% mortality)	*Neanthes arenaceodentata* (polychaete)		
26		NE	COA	20-d	Not significantly toxic (1% mortality)	*Macoma nasuta* (clam)		
26		NE	COA	96-h	Not significantly toxic (0% mortality)	*Acanthomysis sculpta* (mysid)	ADT	
26		NE	COA	96-h	Not significantly toxic (18% mortality)	*Acartia tonsa* (copepod)	ADT	
26.4	±5.08	NE	COA	10-d	Not significantly toxic (0% mortality)	*Crassostrea virginica* (oyster)	ADT	2.5±2.83
26.4	±5.08	NE	COA	10-d	Not significantly toxic (5±1.41% mortality)	*Penaeus duorarum* (pink shrimp)	ADT	2.5±2.83
26.4	±5.08	NE	COA	10-d	Not significantly toxic (8% mortality)	*Arenicola cristata* (lugworm)	ADT	2.5±2.83
26.7	±12	NC	COA		Low abundance (0.3±0.651 N/0.00203 sq.m.)	Amphipoda		0.859±0.487
27		*	AETA	15-min	1986 Puget Sound AET	Microtox (*Photobacterium phosphoreum*)		15
28	±40.1	NE	COA	10-d	Not significantly toxic (0.33±0.516% mortality)	*Palaemonetes pugio* (grass shrimp)		0.493±0.448
28	±40.1	NE	COA	20-d	Not significantly toxic (2.33±1.03% mortality)	*Palaemonetes pugio* (grass shrimp)		0.493±0.448
28	±40.1	NE	COA	20-d	Not toxic (0.367±0.085 mg/day growth)	*Palaemonetes pugio* (grass shrimp)		0.493±0.448
28	±40.1	NE	COA	20-d	Not toxic (0.002±0.0005 mg/day growth)	*Streblospio benedicti* (polychaete worm)		0.493±0.448
28.7	±7.68	NE	COA	10-d	Not significantly toxic (46.7±11.4% mortality)	*Nereis virens* (polychaete)	ADT	
28.7	±7.68	NE	COA	10-d	Not significantly toxic (5.3±2.31% mortality)	*Crassostrea virginica* (oyster)	ADT	
28.7	±7.68	NE	COA	10-d	Not significantly toxic (14±9.17% mortality)	*Penaeus duorarum* (pink shrimp)	ADT	
29		NE	COA		High density (4135 N/0.1 sq.m.)	Annelida	VAR	
29.1	±15.7	SG	COA		Low abundance (1.86±2.59 N/0.00203 sq.m.)	Mollusca		0.784±0.421
29.1	±25.4	NE	COA	10-d	Not significantly toxic (1±1.41% mortality)	*Crassostrea virginica* (oyster)	ADT	7.31±3.86
29.1	±25.4	NE	COA	10-d	Not significantly toxic (4±2.83% mortality)	*Penaeus duorarum* (pink shrimp)	ADT	7.31±3.86
29.1	±25.4	NE	COA	10-d	Not significantly toxic (5±1.41% mortality)	*Nereis virens* (polychaete)	ADT	7.31±3.86
29.2	±16	SG	COA		Low species richness (10±3.73 S/0.00203 sq.m.)	Benthic species		0.795±0.425
29.2	±16	SG	COA		Low abundance (89±60.7 N/0.00203 sq.m.)	Benthic species		0.795±0.425
29.6	±15.6	NE	COA		High abundance (191±70.1 N/0.1 sq.m.)	Echinoderm		
29.8	±11.3	NE	COA		Not significantly toxic (1.03±2.05% mortality)	*Mysidopsis bahia* (mysid shrimp)		0.824±0.915
30	±44.4	SG	COA	10-d	Significantly toxic (29.8±11.1% mortality)	*Streblospio benedicti* (polychaete worm)		0.316±0.12
30	±44.4	NE	COA	20-d	Not significantly toxic (0.003±0.002 mg/day growth)	*Lepidactylus dytiscus* (amphipod)		0.316±0.12
30.5	±16.1	SG	COA		Low abundance (4.21±5.66 N/0.00203 sq.m.)	Oligochaeta		0.895±0.453
31	±12.9	SG	COA		Low abundance (53±33.9 N/0.00203 sq.m.)	Benthic invertebrates		0.985±0.247
31	±16.5	SG	COA		Low abundance (41.7±21.8 N/0.00203 sq.m.)	Polychaeta		0.848±0.427
31.1	±33.2	NE	COA		Not significantly toxic (EC50; 0.133±0.103 mg dry wt/mL)	Microtox (*Photobacterium phosphoreum*)		
31.3	±16.1	*	COA	10-d	Significantly toxic (82.8±22.2% mortality)	*Leptocheirus plumulosus* (amphipod)	JUV	
32.3	±17.5	SG	COA		Moderate abundance (56.2±23 N/0.1 sq.m.)	Echinoderm		
32.7	±17.1	SG	COA		Low abundance (2.05±1.58 N/0.00203 sq.m.)	Copepoda		0.879±0.47
33	±13.9	NC	COA		Significantly toxic(mean ET50,burrowing time3.79±3.29 h)	*Mya arenaria* (clam)	1-2 cm	

sediment-associated chromium ($mg \cdot kg^{-1}$ dry weight) in marine and estuarine systems.

AVS ($\mu mol \cdot g^{-1}$)	Total Sulphide ($mg \cdot kg^{-1}$)	TFS ($mg \cdot L^{-1}$)	Unionized Ammonia ($mg \cdot L^{-1}$)	Sand (%)	Silt (%)	Clay (%)	Area	Reference
							Commencement Bay, WA	Tetra Tech 1985
2.94±2.3				46.7±11	40.7±7.63	12.6±4.37	Galveston Bay, TX	Carr 1992
							Newport, RI.	Lee and Mariani 1977
1.39±0.17				57.4±5.75	34.7±5.09	7.94±0.68	Galveston Bay, TX	Carr 1992
							Sabine-Neches Waterway, TX	Espey, Huston & Assoc. 1983b
							Sabine-Neches Waterway, TX	Espey, Huston & Assoc. 1983b
							Sabine-Neches Waterway, TX	Espey, Huston & Assoc. 1983b
							Houston Ship Channel, TX	Crocker et al. 1991
				7.98±3.2	41.1±3.19	50.9±0.665	Curtis Creek, Baltimore, MD	McGee et al. 1993
				92.4±6.83			Black Rock Beach, NS	Nicol and Doe 1990
				92.4±6.83			Black Rock Beach, NS	Nicol and Doe 1990
				92.4±6.83			Black Rock Beach, NS	Nicol and Doe 1990
1.2±0.39				52.4±11	38.6±8.89	8.98±2.16	Galveston Bay, TX	Carr 1992
1.2±0.39				52.4±11	38.6±8.89	8.98±2.16	Galveston Bay, TX	Carr 1992
4.73±3.1				33.3±17.9	48.2±15.9	18.5±9.96	Galveston Bay, TX	Carr 1992
13.4±8.65				24.5±13.8	43.3±16.8	32.2±8.31	Galveston Bay, TX	Carr 1992
							Commencement Bay, WA	Tetra Tech 1985
							Massachusetts Bay, MA	Gilbert et al. 1976
7.93±5.6				33.2±11.8	46.8±10.9	20±10.4	Galveston Bay, TX	Carr 1992
							Houston Ship Channel, TX	Crocker et al. 1991
4.16±3.45				35.2±18.4	46.9±16.4	17.8±9.16	Galveston Bay, TX	Carr 1992
							Houston Ship Channel, TX	Crocker et al. 1991
0.622±1.12							Tampa Bay, FL	Long 1993
							Pensacola Harbor & Bay, FL	EG&G Bionomics 1980
							Pensacola Harbor & Bay, FL	EG&G Bionomics 1980
							Pensacola Harbor & Bay, FL	EG&G Bionomics 1980
6.37±6.58				31.7±18.8	49.2±16	19.1±9.19	Galveston Bay, TX	Carr 1992
							San Diego Bay, CA	Salazar and Salazar 1985
							San Diego Bay, CA	Salazar and Salazar 1985
							San Diego Bay, CA	Salazar and Salazar 1985
							San Diego Bay, CA	Salazar and Salazar 1985
							San Diego Bay, CA	Salazar and Salazar 1985
							Pensacola Naval Air Station, FL	Parrish 1988a
							Pensacola Naval Air Station, FL	Parrish 1988a
							Pensacola Naval Air Station, FL	Parrish 1988a
7.67±8.12				28.1±19.5	54.3±17.7	17.6±7.44	Galveston Bay, TX	Carr 1992
							Puget Sound, WA	Bellar et al. 1986
2.32±2.25				91.1±11.5	6.53±9.51	2.42±2.16	Chesapeake Bay, VA, MD	Hall et al. 1992
2.32±2.25				91.1±11.5	6.53±9.51	2.42±2.16	Chesapeake Bay, VA, MD	Hall et al. 1992
2.32±2.25				91.1±11.5	6.53±9.51	2.42±2.16	Chesapeake Bay, VA, MD	Hall et al. 1992
2.32±2.25				91.1±11.5	6.53±9.51	2.42±2.16	Chesapeake Bay, VA, MD	Hall et al. 1992
							Bayou La Batre, AB	Parrish 1987a
							Bayou La Batre, AB	Parrish 1987a
							Bayou La Batre, AB	Parrish 1987a
							Massachusetts Bay, MA	Gilbert et al. 1976
8.29±8.13				25.7±17.7	55.8±16.9	18.5±7.14	Galveston Bay, TX	Carr 1992
				29±32.2			Pascagoula Naval Station, MS	Parrish 1990
				29±32.2			Pascagoula Naval Station, MS	Parrish 1990
				29±32.2			Pascagoula Naval Station, MS	Parrish 1990
8.56±8.14				25.3±17.9	56±17.2	18.7±7.17	Galveston Bay, TX	Carr 1992
8.56±8.14				25.3±17.9	56±17.2	18.7±7.17	Galveston Bay, TX	Carr 1992
				32.6			Southern California	Word and Mearns 1979
1.69±0.92				8.82±7.13			Chandeleur Sound, LA	EMAP 1991
2.13±2.46				95.5±3.71	2.76±2.48	1.7±1.4	Chesapeake Bay, VA, MD	Hall et al. 1992
2.13±2.46				95.5±3.71	2.76±2.48	1.7±1.4	Chesapeake Bay, VA, MD	Hall et al. 1992
7.85±8.8				25.9±20.4	57.6±17.3	16.6±5.67	Galveston Bay, TX	Carr 1992
22±11.2				8.03±6.42	58.6±14.3	33.4±7.83	Galveston Bay, TX	Carr 1992
9.21±8.61				22.6±17.7	58.2±17.4	19.1±7.64	Galveston Bay, TX	Carr 1992
1.23±1.96							Tampa Bay, FL	Long 1993
				12.9±8.13	35.6±9.26	51.6±1.38	Curtis Creek, Baltimore, MD	McGee et al. 1993
				49			Southern California	Word and Mearns 1979
9.63±9.65				25.5±20.1	57.7±17.5	16.8±5.37	Galveston Bay, TX	Carr 1992
							Chesapeake Bay,MD,VI; Delaware Bay,DE	Phelps 1990

Appendix Ib. Summary of the available biological effects and related physicochemical data for

Chromium (mg·kg^{-1})		Hit	Analysis Type	Test Type	Endpoint Measured	Species	Life Stage	TOC (%)
33.3	±17.7	NC	COA	48-h	Significantly toxic (48.1±1.91% mortality)	*Mulinia lateralis* (coot clam)	LAR	
33.4	±42.2	NE	COA	20-d	Not significantly toxic (0.001±0.001 mg/day growth)	*Hyalella azteca* (amphipod)	JUV	0.555±0.471
33.5	±10.5	NC	COA	10-d	Significantly toxic (11.5±4.86% mortality)	*Ampelisca abdita* (amphipod)		0.811±0.711
33.6	±35.9	NE	COA	10-d	Not significantly toxic (9.5±6.15% mortality)	*Ampelisca abdita* (amphipod)	SUBADT	
34	±10.8	NE	COA	10-d	Not toxic (4.5±3.02% emergence)	*Rhepoxynius abronius* (amphipod)	ADT	2.66±2.15
34.6	±11.5	SG	COA	48-h	Toxic (4.65±16.1% normal development)	*Arbacia punctulata* (sea urchin)	EMB	1.06±0.449
35.7	±16.5	*	COA	20-d	Significantly toxic (94.6±9.35% mortality)	*Leptocheirus plumulosus* (amphipod)	JUV	
35.7	±18.5	NE	COA	10-d	Not significantly toxic (2.87±4.05% mortality)	*Ampelisca abdita* (amphipod)		0.517±0.166
36.2	±11.2	NE	COA	10-d	Not toxic (5.21±3.61% emergence)	*Corophium volutator* (amphipod)	ADT	3.18±2.1
37.5	±12.9	NE	COA	10-d	Not toxic (8.9±2.99% mortality)	*Corophium volutator* (amphipod)	ADT	2.8±1.96
37.5	±12.9	NE	COA	10-d	Not toxic (97.2±2.84% reburial)	*Rhepoxynius abronius* (amphipod)	ADT	2.8±1.96
37.8	±13.4	SG	COA		Significantly toxic (7.6±1.32% mortality)	*Mysidopsis bahia* (mysid shrimp)		0.624±0.197
38.1	±36.3	NC	COA		Moderate species richness (72±3.3 S/0.1 sq.m.)	Benthic species		
38.5	±26.3	NE	COA	10-d	Not significantly toxic (10.6±3.17% mortality)	*Ampelisca abdita* (amphipod)		1.19±1.04
38.5	±26.3	NE	COA	10-d	Not significantly toxic (8±2.92% mortality)	*Mysidopsis bahia* (mysid)		1.19±1.04
39.1	±13.1	SG	COA	1-h	Toxic (20.4±18.9% fertilization)	*Arbacia punctulata* (sea urchin)	EMB	1.26±0.47
39.9	±12.7	NE	COA	10-d	Not toxic (7.9±5.12% mortality)	*Rhepoxynius abronius* (amphipod)	ADT	2.64±2.14
39.9	±27.1	NE	COA	96-h	Not toxic (9.61±2.7% mortality)	*Menidia beryllina* (silverside)		1.22±1.13
40.1	±23.5	NE	COA		Not significantly toxic (0.789±1.59% mortality)	*Mysidopsis bahia* (mysid shrimp)		0.827±0.757
40.2	±27.3	NE	COA	96-h	Not toxic (5.43±3.21% mortality)	*Mysidopsis bahia* (mysid)		1.21±1.13
40.7	±30.9	NE	COA		High abundance (148±58 N/0.1 sq.m.)	Arthropods		
41.9	±34	NE	COA	10-d	Not significantly toxic (5.28±3.04% mortality))	*Ampelisca abdita* (amphipod)	ADT	
42	±11	NE	COA		Sediments populated by feral clams	*Macoma balthica* (bivalve)		0.45
42	±39.8	NC	COA		Moderate abundance (75.6±12.7 N/0.1 sq.m.)	Benthic species		
42.5	±37	NE	COA	48-h	Not significantly toxic (38.5±2.9% mortality)	*Mulinia lateralis* (coot clam)	LAR	
43	±26.9	NC	COA	10-d	Significantly toxic (15.8±3.89% mortality)	*Ampelisca abdita* (amphipod)		0.647±0.606
45.8	±21.5	NE	COA		High density (201±33.4 N/0.1 sq.m.)	Mollusca	VAR	
46.3	±43.3	SG	COA		Moderate abundance (72.6±6.8 N/0.1 sq.m.)	Arthropods		
47.6	±2.7	SG	COA	96-h	Highly toxic (>50% mortality)	*Palaemonetes pugio* (shrimp)		
48.1	±22.8	NE	COA	10-d	Not significantly toxic (5.8±3.7% mortality)	*Rhepoxynius abronius* (amphipod)		2.83±0.459
48.1	±22.8	NE	COA	10-d	Not significantly toxic (3.4±1.82% mortality)	*Nereis virens* (polychaete)		2.83±0.459
48.1	±22.8	NE	COA	28-d	Not significantly toxic (11.2±3.7% mortality)	*Nereis virens* (polychaete)		2.83±0.459
48.1	±22.8	NE	COA	28-d	Not significantly toxic (3.52±1.66% mortality)	*Macoma nasuta* (clam)		2.83±0.459
48.3	±51.9	*	COA	10-d	Significantly toxic (55±22.6% mortality)	*Hyalella azteca* (amphipod)	JUV	0.338±0.133
49.3	±21.9	NE	COA		Not significantly toxic (0% mortality)	*Mysidopsis bahia* (mysid shrimp)		0.934±0.657
49.6	±24.1	SG	COA		Low density (1316±768 N/0.1 sq.m.)	Annelida	VAR	
50.2	±26.1	NE	COA	10-d	Not significantly toxic (5.3±3.06% mortality)	*Penaeus duorarum* (pink shrimp)	ADT	5.2±2.6
50.2	±26.1	NE	COA	10-d	Not significantly toxic (8±6% mortality)	*Arenicola cristata* (lugworm)	ADT	5.2±2.6
50.2	±26.1	NE	COA	10-d	Not significantly toxic (0% mortality)	*Crassostrea virginica* (oyster)	ADT	5.2±2.6
51		NE	COA		Most toxic (100 TU/g)	Microtox (*Photobacterium phosphoreum*)		0.4
51.2	±18.3	NE	COA	10-d	Not significantly toxic (1.4±1.73% mortality)	*Ampelisca abdita* (amphipod)		0.998±0.602
51.2	±42.9	NE	COA		High Abundance (6±1.41 N/0.00203 sq.m.)	Amphipoda		0.955±0.629
51.6	±3.05	NE	COA	10-d	Not toxic (0.36±0.261% emergence/d)	*Rhepoxynius abronius* (amphipod)	ADT	0.42±0.13
51.7	±2.56	NE	COA	48-h	Not toxic (4.54±5.37% mortality)	*Crassostrea gigas* (oyster)	LAR	0.443±0.127
51.9	±2.42	NE	COA	48-h	Not significantly toxic (7.76±1.89% abnormality)	*Crassostrea gigas* (oyster)	LAR	0.425±0.128
52.3			TEL					
52.8	±2.36	NE	COA	10-d	Significantly toxic (41.75±38.9% mortality)	*Rhepoxynius abronius* (amphipod)	ADT	0.45±0.058
53.8	±14.2	NE	COA		Not significantly toxic (1.58±1.82% mortality)	*Mysidopsis bahia* (mysid shrimp)		0.944±0.54
53.8	±14.2	NE	COA	10-d	Not significantly toxic (2.33±4.65% mortality)	*Ampelisca abdita* (amphipod)		0.944±0.54
54		*	COA	10-d	Significantly toxic (55% mortality)	*Hyalella azteca* (amphipod)	JUV	
54	±83.5	NC	COA		Low abundance (57.6±13.6 N/0.1 sq.m.)	Benthic species		
54.7		SG	COA	96-h	Toxic (16% mortality)	*Mysidopsis bahia* (mysid)		1.8
54.9		SG	COA	10-d	Highly toxic (23% emergence)	*Corophium volutator* (amphipod)	ADT	3.5
54.9		SG	COA	10-d	Highly toxic (30.5% emergence)	*Rhepoxynius abronius* (amphipod)	ADT	3.5
55.3	±21	SG	COA		Significantly toxic (12.2±4.42% mortality)	*Mysidopsis bahia* (mysid shrimp)		0.939±0.335
56	±1.41	NE	COA	10-d	Not significantly toxic (7.8±1.63% mortality)	*Ampelisca abdita* (amphipod)		1.15±1.04
56	±18.2	SG	COA		Significantly toxic (13.4±9.11% mortality)	*Mysidopsis bahia* (mysid shrimp)		0.778±0.482
56.7		SG	COA	96-h	Toxic (22.3% mortality)	*Menidia beryllina* (silverside)		1.71
57.5	±15.2	NE	COA	10-d	Not significantly toxic (1.35±1.45% mortality)	*Ampelisca abdita* (amphipod)		0.961±0.537
57.9	±44.9	*	COA	1-h	Most toxic (1.96±3.19% fertilization)	*Arbacia punctulata* (sea urchin)	GAM	
58		NE	COA		High abundance (208 N/0.1 sq.m.)	Echinoderms	VAR	
58.6	±129	NE	COA		Not significantly toxic (mean ET50;	*Mya arenaria* (clam)	1-2 cm	
59		*	AETA		1986 Puget Sound AET	Benthic species		15
59	±27.2	NC	COA		Low density (7.14±4.23 N/0.1 sq.m.)	Rhynchocoela	VAR	

sediment-associated chromium ($mg \cdot kg^{-1}$ dry weight) in marine and estuarine systems.

AVS ($\mu mol \cdot g^{-1}$)	Total Sulphide ($mg \cdot kg^{-1}$)	TFS ($mg \cdot L^{-1}$)	Unionized Ammonia ($mg \cdot L^{-1}$)	Sand (%)	Silt (%)	Clay (%)	Area	Reference
2.82±2.65							Tampa Bay, FL	Long 1993
2.68±2.3				89.3±11.9	7.84±10	2.9±2.01	Chesapeake Bay, VA, MD	Hall et al. 1992
1.87±0.973				7.38±5.97			Chandeleur Sound, LA	EMAP 1991
1.56±2.73							Tampa Bay, FL	Long 1993
				25.7±41.3	39.1±18.7	52.6±25.5	Burrard Inlet, BC	McLeay et al. 1991
14.5±9.2				15.9±12.9	64±13.4	20.1±7.68	Galveston Bay, TX	Carr 1992
				14.9±8.59	33.1±9.68	51.9±1.45	Curtis Creek, Baltimore, MD	McGee et al. 1993
2.02±1.38				15.8±22.5			Chandeleur Sound, LA	EMAP 1991
				27.6±47.4	24.8±21.6	46.7±35.3	Burrard Inlet, BC	McLeay et al. 1991
				23±37.5	40.3±16.4	51.2±22.3	Burrard Inlet, BC	McLeay et al. 1991
				23±37.5	40.3±16.4	51.2±22.3	Burrard Inlet, BC	McLeay et al. 1991
2.07±1.19				11.3±17.1			Chandeleur Sound, LA	EMAP 1991
							Southern California	Word and Mearns 1979
7864±11973 µg/g							Brunswick Harbor Entrance, GA	Windom 1995
7864±11973 µg/g							Brunswick Harbor Entrance, GA	Windom 1995
14.6±10.1				13±11.1	70.5±11.6	16.4±5.2	Galveston Bay, TX	Carr 1992
				27.5±40.1	46.9±8.02	42.7±13.2	Burrard Inlet, BC	McLeay et al. 1991
9172±13363 µg/g							Brunswick Harbor Entrance, GA	Windom 1995
1.46±1.58				24.4±36.1			Mississippi Sound, MS	EMAP 1991
9096±13378 µg/g							Brunswick Harbor Entrance, GA	Windom 1995
				59.4			Southern California	Word and Mearns 1979
				50.2±34.3	23.6±17.2	26.2±20.5	Narragansett Bay, RI	Munns et al. 1991
							Fraser River Estuary, BC	McGreer 1982
							Southern California	Word and Mearns 1979
2.64±3.04							Tampa Bay, FL	Long 1993
0.16±0.184				18.3±3.77			Galveston Bay, TX	EMAP 1991
							Massachusetts Bay, MA	Gilbert et al. 1976
				49.4			Southern California	Word and Mearns 1979
							Los Angeles, CA	Lee and Mariani 1977
16.3±9.58				51.6±31.4	20.2±15	28.2±17.4	Wilmington Harbor, NC	Ward et al. 1992
16.3±9.58				51.6±31.4	20.2±15	28.2±17.4	Wilmington Harbor, NC	Ward et al. 1992
16.3±9.58				51.6±31.4	20.2±15	28.2±17.4	Wilmington Harbor, NC	Ward et al. 1992
16.3±9.58				51.6±31.4	20.2±15	28.2±17.4	Wilmington Harbor, NC	Ward et al. 1992
2.9±3.11				94.7±3.84	3.3±2.72	2.03±1.44	Chesapeake Bay, VA, MD	Hall et al. 1992
0.457±0.53				16.1±4.68			Galveston Bay, TX	EMAP 1991
							Massachusetts Bay, MA	Gilbert et al. 1976
							Bayou Casotte, MS	Parrish 1988b
							Bayou Casotte, MS	Parrish 1988b
							Bayou Casotte, MS	Parrish 1988b
				24.6	29.1	46.1	Gulf of Mexico	Chapman et al. 1991
2.01±1.82				7.47±3.87			Mississippi Sound, MS	EMAP 1991
7.21±8.2				36.4±20.8	46.9±9.5	16.7±11.3	Galveston Bay, TX	Carr 1992
				33.8±15.6	35.8±11.1	30.3±10.1	Gulf of Mexico	Chapman et al. 1991
				31.4±13.3	37.1±10.5	31.1±9.29	Gulf of Mexico	Chapman et al. 1991
				31.4±12.3	36.6±9.79	31.6±8.73	Gulf of Mexico	Chapman et al. 1991
				22.2±5.33	37.4±11.3	39.7±5.97	Gulf of Mexico	Chapman et al. 1991
1.93±1.58				20±9.22			Matagorda Bay, TX	EMAP 1991
1.93±1.58				20±9.22			Matagorda Bay, TX	EMAP 1991
				24.5	22.1	53.4	Curtis Creek, Baltimore, MD	McGee et al. 1993
							Southern California	Word and Mearns 1979
7095 µg/g							Brunswick Harbor Entrance, GA	Windom 1995
				9.5	44.9	45.6	Burrard Inlet, BC	McLeay et al. 1991
				9.5	44.9	45.6	Burrard Inlet, BC	McLeay et al. 1991
2.63±2.3				9.68±5.01			Mississippi Sound, MS	EMAP 1991
2.31±0.651				7.7±4.05			Mississippi River, MS, LA	EMAP 1991
1.21±0.778				22.8±8.31			Galveston Bay, TX	EMAP 1991
6562 µg/g							Brunswick Harbor Entrance, GA	Windom 1995
1.17±0.64				20±8.77			Galveston Bay, TX	EMAP 1991
3.95±5.94							Tampa Bay, FL	Long 1993
10				43.2			Palos Verdes, CA	Ferraro et al. 1991
							Chesapeake Bay,MD,VI; Delaware Bay,DE	Phelps 1990
							Puget Sound, WA	Bellar et al. 1986
							Massachusetts Bay, MA	Gilbert et al. 1976

Appendix Ib. Summary of the available biological effects and related physicochemical data for

Chromium (mg·kg^{-1})		Hit	Analysis Type	Test Type	Endpoint Measured	Species	Life Stage	TOC (%)
59.9	±9.2	*	COA	48-h	Significantly toxic (14.6±18.9% normal development)	*Arbacia punctulata* (sea urchin)		1.99±0.562
60		*	COA	48-h	Significant increase in burrowing time	*Macoma balthica* (bivalve)		1.13
60		NE	SQO		Sediment Quality Objectives	Aquatic biota		
60.4	±26.1	*	COA		Low density (2.76±2.7 N/0.1 sq.m.)	Echinodermata	VAR	
60.6	±27.3	SG	COA		Low density (39.1±30.8 N/0.1 sq.m.)	Mollusca	VAR	
60.8	±8.77	NE	COA	10-d	Not toxic (10.5±4.09% mortality)	*Rhepoxynius abronius* (amphipod)	ADT	3.4±4.39
60.8	±8.77	NE	COA	10-d	Not toxic (5.83±5.48% emergence)	*Rhepoxynius abronius* (amphipod)	ADT	3.4±4.39
60.8	±8.77	NE	COA	10-d	Not toxic (96.3±4.19% reburial)	*Rhepoxynius abronius* (amphipod)	ADT	3.4±4.39
61.2	±25.9	*	COA		Low density (61.1±36.6 N/0.1 sq.m.)	Arthropoda	VAR	
62		NE	COA		High density (4.4 N/0.1 sq.m.)	Echinoderm		0.9
62		NE	COA		High density (88 N/0.1 sq.m.)	Foraminifera (sponge)		0.9
62		NE	COA		High density (132 N/0.1 sq.m.)	Ophiuroidea (brittle star)		0.9
62	±139	NE	COA		High species richness (96.3±22.3 S/0.1 sq.m.)	Benthic species		
62	±25.7	*	COA		Low species richness (57.2±12.2 S/0.1 sq.m.)	Benthic species	VAR	
62.8	±63.7	NE	COA	20-d	Not significantly toxic (16.5±9.19% mortality)	*Streblospio benedicti* (polychaete worm)		0.783±0.845
66	±36.4	NE	COA		Not significantly toxic (1.07±1.85% mortality)	*Mysidopsis bahia* (mysid shrimp)		0.968±0.815
66.5	±13.4	NE	COA		Not significantly toxic (0% mortality)	*Mysidopsis bahia* (mysid shrimp)		1.3±0.83
67.5	±5.9	NE	COA	96-h	Not toxic (0% mortality)	*Palaemonetes pugio* (shrimp)		
67.8	±36.4	NE	COA		Not toxic (3.21±0.239 diversity index)	Benthic species		0.464±0.143
69.5	±20.5	SG	COA		Significantly toxic (32.9±36.4% mortality)	*Mysidopsis bahia* (mysid shrimp)		1.07±0.924
69.5	±37.9	NE	COA		Least toxic (6.32±3.68 TU/g)	Microtox (*Photobacterium phosphoreum*)		0.47±0.149
70		NE	COA		High density (22 N/0.1 sq.m.)	Rhynchocoela	VAR	
73	±124	NE	COA	10-d	Not significantly toxic (23.2% mortality)	*Grandidierella japonica* (amphipod)	JUV	
73.8	±19	NE	COA	10-d	Not significantly toxic (17.2±7.99% mortality)	*Ampelisca abdita* (amphipod)	SUBADT	1.14±0.352
76.4	±19.3	NC	COA	10-d	Significantly toxic (25.9±5.01% mortality)	*Ampelisca abdita* (amphipod)	SUBADT	1.21±0.444
76.4	±44.3	NE	COA	10-d	Not significantly toxic (9±3.96% mortality)	*Rhepoxynius abronius* (amphipod)	ADT	0.471±0.18
77.4	±45.8	*	COA		Significantly toxic (EC50; 0.021±0.012 mg dry wt/mL)	Microtox (*Photobacterium phosphoreum*)		
80	±5.66	SG	COA	10-d	Significantly toxic (15.9±6.65% mortality)	*Ampelisca abdita* (amphipod)		1.22±0.717
81.3	±46.4	SG	COA	10-d	Highly toxic (2.5±1.05% emergence/d)	*Rhepoxynius abronius* (amphipod)	ADT	0.5±0.155
81.4	±88.5	SG	COA	10-d	Significantly toxicity (51.7% mortality)	*Grandidierella japonica* (amphipod)	JUV	
81.9	±20.9	NE	COA		Not significantly toxic (EC50; 0.125±0.084 mg dry wt/mL)	Microtox (*Photobacterium phosphoreum*)		1.25±0.468
83.5	±24.7	NC	COA	48-h	Significantly toxic (96±1.66% normal development)	*Mulinia lateralis* (bivalve)	LAR	2.52±0.997
83.6	±22.1	NE	COA	48-h	Not significantly toxic (6.48±10.1% mortality)	*Mulinia lateralis* (bivalve)	LAR	1.46±0.733
83.6	±22.1	NE	COA	48-h	Not significantly toxic (99±0.862% normal development)	*Mulinia lateralis* (bivalve)	LAR	1.46±0.733
83.8	±25.5	NE	COA	10-d	Not significantly toxic (17.2±6.07% mortality)	*Ampelisca abdita* (amphipod)	SUBADT	1.71±0.9
84.3	±18.5	NE	COA	10-d	Not significantly toxic (19.3±5.86% mortality)	*Ampelisca abdita* (amphipod)	SUBADT	1.82±0.225
85.3	±32.7	NE	COA		Not significantly toxic (EC50; 0.096±0.101 mg dry wt/mL)	Microtox (*Photobacterium phosphoreum*)		1.37±0.759
85.6	±14.7	NC	COA	48-h	Significantly toxic (68.5±11.4% mortality)	*Mulinia lateralis* (bivalve)	LAR	2.33±0.364
86		NE	COA	48-h	Least toxic (17.3% mortality)	Mussel	LAR	1.25
86	±6.7	NE	COA	96-h	Not toxic (10% mortality)	*Palaemonetes pugio* (shrimp)		
86.9	±37.1	NE	COA		Not significantly toxic (EC50; 0.074±0.043 mg dry wt/mL)	Microtox (*Photobacterium phosphoreum*)		1.27±0.626
87		NE	COA	10-d	Not significantly toxic (0.5±0.778% mortality)	*Ampelisca abdita* (amphipod)		1.24±0.948
87	±9.19	NG	COA	48-h	Moderately toxic (57.1±13.6% mortality)	Mussel	LAR	1.14±0.33
87.3	±22.1	*	COA		Sediments devoid of feral clams	*Macoma balthica* (bivalve)		1.95
87.4	±33.1	NE	COA	48-h	Not significantly toxic (99.3±0.827% normal development)	*Mulinia lateralis* (bivalve)	LAR	1.43±0.738
87.5	±41.7	NE	COA		High abundance (54.6±5.09 N/0.1 sq.m.)	Amphipod	VAR	
87.5	±41.7	NE	COA		High abundance (30.7±0.99 N/0.1 sq.m.)	Phoxocephalids	VAR	
88.2	±82.7	NE	COA	48-h	Least toxic (23.3±7.3% abnormal)	Bivalve	LAR	
89	±4.55	NE	COA	4-wk	Least toxic (116±4.3 young produced)	*Tigriopus californicus* (copepod)	ADT	1.23±0.09
89.5	±34.6	SG	COA		Significantly toxic (EC50; 0.008±0.001 mg dry wt/mL)	Microtox (*Photobacterium phosphoreum*)		2.19±1.27
90		*	COA	24-h	Significantly toxic (avoidance)	*Macoma balthica* (bivalve)		3.09
90.1	±11.5	NE	COA	48-h	Least toxic (18±8.01% abnormal)	Mussel	LAR	1.2±0.38
90.6	±15.8	SG	COA	10-d	Significantly toxic (39.5±18.2% mortality)	*Ampelisca abdita* (amphipod)	SUBADT	2.06±0.523
92.6	±15.5	NC	COA		Significantly toxic (EC50; 0.014±0.006 mg dry wt/mL)	Microtox (*Photobacterium phosphoreum*)		2.51±0.45
93	±31.8	NE	COA	48-h	Not significantly toxic (12.3±14.5% mortality)	*Mulinia lateralis* (bivalve)	LAR	1.53±0.743
93.7	±35.8	*	COA	10-d	Significantly toxic (32.1±16.9% mortality)	*Ampelisca abdita* (amphipod)	SUBADT	
94		NE	COA		High density (93.4 N/0.1 sq.m.)	Echinoderm		1.3±0.77
94.2	±17.4	NE	COA		Not significantly toxic (EC50; 0.061±0.038 mg dry wt/mL)	Microtox (*Photobacterium phosphoreum*)		1.8±0.477
94.7	±14.7	NE	COA	48-h	Not significantly toxic (98.3±2.22% normal development)	*Mulinia lateralis* (bivalve)	LAR	2.18±0.531
95	±13.2	SG	COA	48-h	Moderately toxic (25.1±6.61% abnormal)	Mussel	LAR	1.26±0.17
95.3	±23.6	NE	COA	10-d	Least toxic (13.6±7.76% mortality)	Amphipod	ADT	1.4±0.79
96	±52.3	SG	COA	48-h	Highly toxic (59.4±15.5% mortality)	*Crassostrea gigas* (oyster)	LAR	0.5±0.18
97.1	±15.4	NE	COA	48-h	Not significantly toxic (15.2±9.21% mortality)	*Mulinia lateralis* (bivalve)	LAR	2.17±0.663
97.1	±22.5	NE	COA	10-d	Least Toxic (4.63±2.91% avoidance)	Amphipod	ADT	1.44±0.74

sediment-associated chromium ($mg \cdot kg^{-1}$ dry weight) in marine and estuarine systems.

AVS ($\mu mol \cdot g^{-1}$)	Total Sulphide ($mg \cdot kg^{-1}$)	TFS ($mg \cdot L^{-1}$)	Unionized Ammonia ($mg \cdot L^{-1}$)	Sand (%)	Silt (%)	Clay (%)	Area	Reference
14009±13434 µg/g							Brunswick Harbor Entrance, GA	Windom 1995
							Strait of Georgia, BC	McGreer 1979
							Burrard Inlet, BC	Swain and Nijman 1991
							Massachusetts Bay, MA	Gilbert et al. 1976
							Massachusetts Bay, MA	Gilbert et al. 1976
				39.9±51.5	34.6±8.34	51.3±27.5	Howe Sound, BC	McLeay et al. 1991
				39.9±51.5	34.6±8.34	51.3±27.5	Howe Sound, BC	McLeay et al. 1991
				39.9±51.5	34.6±8.34	51.3±27.5	Howe Sound, BC	McLeay et al. 1991
							Massachusetts Bay, MA	Gilbert et al. 1976
	24	0.33	0.015	38	53.7	8.3	Palos Verdes, CA	Swartz et al 1985a
	24	0.33	0.015	38	53.7	8.3	Palos Verdes, CA	Swartz et al. 1985a
	24	0.33	0.015	38	53.7	8.3	Palos Verdes, CA	Swartz et al. 1985a
							Southern California	Word and Mearns 1979
							Massachusetts Bay, MA	Gilbert et al. 1976
4.83±2.23				83.7±21.4	13.1±17.4	3.2±3.96	Chesapeake Bay, VA, MD	Hall et al. 1992
8.23±3.4				19.8±4.69			Mobile Bay, AL	EMAP 1991
1.14±1				25.3±20.9			Mississippi River, MS, LA	EMAP 1991
							Norwalk River, CT	Lee and Mariani 1977
				28.8±11.4	37.8±9.61	29.9±11.4	Gulf of Mexico	Chapman et al. 1991
1.41±1.93				10.5±7.94			Mississippi River, MS, LA	EMAP 1991
				29.2±12	38.7±9.67	28.3±10.6	Gulf of Mexico	Chapman et al. 1991
							Massachusetts Bay, MA	Gilbert et al. 1976
							Southern California	Anderson et al. 1988
6.09±5.38				34.3±29.2	31±26.5	32.9±17	Long Island Sound, NY, CT	Bricker et al. 1993
2.41±2.32				42.9±29.8	40.3±21.4	16±9.54	Long Island Sound, NY, CT	Bricker et al. 1993
				32.6±12.5	38±9.47	24.3±9.91	Gulf of Mexico	Chapman et al. 1991
6.13±8.06							Tampa Bay, FL	Long 1993
0.235±0.276				28.1±17			Mississippi River, MS, LA	EMAP 1991
				24.7±4.7	39.4±8.91	29.6±13.3	Gulf of Mexico	Chapman et al. 1991
							Southern California	Anderson et al. 1988
1.98±2.25				32.8±30.5	46.5±21.2	19.8±10.9	Long Island Sound, NY, CT	Bricker et al. 1993
35±42.9				22.4±22.6	46.8±15.6	30.6±7.42	Long Island Sound, NY, CT	Bricker et al. 1993
4.2±6.27				36.7±29.4	44.1±20.4	18.4±10.3	Long Island Sound, NY, CT	Bricker et al. 1993
4.2±6.27				36.7±29.4	44.1±20.4	18.4±10.3	Long Island Sound, NY, CT	Bricker et al. 1993
15.3±23.2				31±23.5	37.1±21.8	30.6±13.8	Long Island Sound, NY, CT	Bricker et al. 1993
6.3±6.2				23±19.7	48±14.9	28.4±5.2	Long Island Sound, NY, CT	Bricker et al. 1993
6.93±6.83				47±30.3	30.6±21	20.3±14.7	Long Island Sound, NY, CT	Bricker et al. 1993
26.8±9.42				23.9±10.2	33.5±9.38	23.7±4.83	Long Island Sound, NY, CT	Bricker et al. 1993
				18.6	37.7	43.8	San Francisco Bay, CA	Chapman et al. 1987a
							Stamford, CT	Lee and Mariani 1977
6.21±5.41				49.8±30.2	27.4±20.3	20.4±15.8	Long Island Sound, NY, CT	Bricker et al. 1993
8.51±4.76				17.5±3.49			Mobile Bay, AL	EMAP 1991
				23.3	32.6	44	San Francisco Bay, CA	Chapman et al. 1987a
							Fraser River Estuary, BC	McGreer 1982
14.7±24				43.8±31	31.2±21	22.5±15.1	Long Island Sound, NY, CT	Bricker et al. 1993
	55.5±64.3			29.5±19.4			Palos Verdes, CA	Ferraro et al. 1991
	55.5±64.3			29.5±19.4			Palos Verdes, CA	Ferraro et al. 1991
							San Francisco Bay, CA	Long and Morgan 1990
				15.2	36.2	48.7	San Francisco Bay, CA	Chapman et al. 1987a
12±11.3				50.3±29.8	35.8±21.1	13.6±8.13	Long Island Sound, NY, CT	Bricker et al. 1993
							Strait of Georgia, BC	McGreer 1979
				20.5	34.4	45.1	San Francisco Bay, CA	Chapman et al. 1987a
20.1±13.9				22.3±9.32	39.6±11.2	26.3±5.56	Long Island Sound, NY, CT	Bricker et al. 1993
29.6±15.7				20.2±11	40.4±12	28.1±7.08	Long Island Sound, NY, CT	Bricker et al. 1993
15.7±23.2				38.6±28.5	34.8±20.4	24±14.3	Long Island Sound, NY, CT	Bricker et al. 1993
7.63±9.44							Tampa Bay, FL	Long 1993
	226						Palos Verdes, CA	Swartz et al. 1986
8.52±10.7				20±13.1	48.3±10.3	30.3±3.96	Long Island Sound, NY, CT	Bricker et al. 1993
19.1±12.5				19.8±10.4	43.1±11.6	28.8±6.03	Long Island Sound, NY, CT	Bricker et al. 1993
				14.7	36.3	49	San Francisco Bay, CA	Chapman et al. 1987a
				20.1	40.4	39.4	San Francisco Bay, CA	Chapman et al. 1987a
				24.3±6.01	39±9.31	27.8±15.9	Gulf of Mexico	Chapman et al. 1991
18.4±19.9				18.2±12	48.6±9.29	31.6±4.65	Long Island Sound, NY, CT	Bricker et al. 1993
				18.7	40.3	41	San Francisco Bay, CA	Chapman et al. 1987a

Appendix Ib. Summary of the available biological effects and related physicochemical data for

Chromium (mg·kg^{-1})		Hit	Analysis Type	Test Type	Endpoint Measured	Species	Life Stage	TOC (%)
97.5	±66.7	NC	COA	48-h	Highly toxic (92.4±4.5% abnormal)	Bivalve	LAR	
98.7	±31.1	NE	COA	48-h	Not significantly toxic (17±16.1% mortality)	*Mulinia lateralis* (bivalve)	LAR	1.99±1
99.9	±32.8	SG	COA	10-d	Significantly toxic (30.5±16.4% mortality)	*Ampelisca abdita* (amphipod)	SUBADT	1.62±0.777
101	±21.8	NE	COA		Not significantly toxic (EC50; 0.167±0.198 mg dry wt/mL)	Microtox (*Photobacterium phosphoreum*)		1.59±1.09
101	±33.7	NE	COA	48-h	Not significantly toxic (98.1±2.78% normal development)	*Mulinia lateralis* (bivalve)	LAR	2.12±1.04
107	±96.4	SG	COA	48-h	Significantly toxic (51.3±6.95% mortality)	*Mulinia lateralis* (bivalve)	LAR	1.8±1.71
108		*	COA	10-d	Significantly toxic (29% mortality)	*Lepidactylus dytiscus* (amphipod)		0.185
108		*	COA	10-d	Significantly toxic (89.7% reburial)	*Lepidactylus dytiscus* (amphipod)		0.185
108		*	COA	20-d	Significantly toxic (>10% emergence)	*Lepidactylus dytiscus* (amphipod)		0.185
108		*	COA	20-d	Significantly toxic (>10% emergence)	*Hyalella azteca* (amphipod)	JUV	0.185
109	±36.2	SG	COA	10-d	Significantly toxic (34.9±15.7% mortality)	*Ampelisca abdita* (amphipod)	SUBADT	2.23±1.04
109	±7.78	NE	COA	10-d	Not significantly toxic (21.5±6.36% mortality)	*Ampelisca abdita* (amphipod)	SUBADT	2.35±1.05
110	±53.5	*	COA	48-h	Significantly toxic (32.6±14.2% abnormality)	*Crassostrea gigas* (oyster)	LAR	0.567±0.153
113	±44.4	SG	COA		Significantly toxic (EC50; 0.016±0.007 mg dry wt/mL)	Microtox (*Photobacterium phosphoreum*)		2.26±0.936
116	±23.5	NE	COA	10-d	Not significantly toxic (5.6±2.2% mortality)	*Arenicola cristata* (lugworm)	ADT	
116	±23.5	NE	COA	10-d	Not significantly toxic (6.5±5.14% mortality)	*Penaeus duorarum* (pink shrimp)	ADT	
116	±23.5	NE	COA	10-d	Not significantly toxic (0.7±0.82% mortality)	*Crassostrea virginica* (oyster)	ADT	
116	±27.1	SG	COA	10-d	Moderately toxic (28.3±7.51% mortality)	Amphipod	ADT	2.01±0.98
117	±19.8	NE	COA	10-d	Not significantly toxic (23±4.24% mortality)	*Ampelisca abdita* (amphipod)	SUBADT	2.46±1.22
117	±31.9	SG	COA		Significantly toxic (EC50; 0.013±0.006 mg dry wt/mL)	Microtox (*Photobacterium phosphoreum*)		2.79±0.744
117	±46.5	SG	COA	48-h	Significantly toxic (68.1±13.3% mortality)	*Mulinia lateralis* (bivalve)	LAR	2.68±0.955
121	±25	NE	COA	48-h	Not significantly toxic (29.8±13.9% mortality)	*Mulinia lateralis* (bivalve)	LAR	2.65±1.07
124	±55.4	SG	COA	48-h	Significantly toxic (91.5±7.28% normal development)	*Mulinia lateralis* (bivalve)	LAR	2.6±0.899
128	±23.7	NE	COA	48-h	Not significantly toxic (96.6±3.79% normal development)	*Mulinia lateralis* (bivalve)	LAR	2.87±0.94
129	±24.3	SG	COA	10-d	Significantly toxic (38±13% mortality)	*Ampelisca abdita* (amphipod)	SUBADT	2.92±0.937
130	±18.3	SG	COA	48-h	Highly toxic (92.3±5.5% mortality)	Mussel	LAR	2.87±1.32
130	±18.3	SG	COA	4-wk	Moderately toxic (94.9±10.1 young produced)	*Tigriopus californicus* (copepod)	ADT	2.87±1.07
134		SG	COA	10-d	Most toxic (95% mortality)	Amphipod	ADT	4.03
134		SG	COA	10-d	Highly toxic (37% avoidance)	Amphipod	ADT	4.03
134	±94.2	NC	COA	48-h	Significantly toxic (55.7±22.7% abnormal)	Bivalve	LAR	
137	±17.4	SG	COA		Significantly toxic (EC50; 0.011±0.006 mg dry wt/mL)	Microtox (*Photobacterium phosphoreum*)		3.28±0.363
140		SG	COA	48-h	Highly toxic (66.8% abnormal)	Mussel	LAR	3.59
142	±43.9	NE	COA		High density (20.9±0.23 N/0.1 sq.m.)	Amphipod		1.7±0.61
142	±43.9	NE	COA		High density (11.2±1.64 N/0.1 sq.m.)	Phoxocephalid		1.7±0.61
142	±86.5	NC	COA	10-d	Highly toxic (67±11.8% mortality)	*Rhepoxynius abronius* (amphipod)	ADT	
143	±11.9	SG	COA	48-h	Significantly toxic (75.1±10.6% mortality)	*Mulinia lateralis* (bivalve)	LAR	3.32±0.317
145	±88.6	NE	COA	14-d	Not toxic (increased growth rate)	*Chromadorina germanica* (nematode)		
146	±308	*	COA		Low abundance (35.3±15.8 N/0.1 sq.m.)	Arthropods		
150	±85.9	NE	COA	48-h	Not significantly toxic (31.9±15.5% abnormal)	Bivalve	LAR	
152	±55.1	SG	COA	48-h	Significantly toxic (88.5±8.39% normal development)	*Mulinia lateralis* (bivalve)	LAR	2.65±1.05
155	±102	NC	COA	10-d	Significantly toxic (42.9±19.2% mortality)	*Rhepoxynius abronius* (amphipod)	ADT	
156	±131	NE	COA		High species richness (80.8±13.7 S/0.1 sq.m.)	Macro benthos		2.1±1.15
156	±131	NE	COA		High density (54.5±9.91 N/0.1 sq.m.)	Amphipod		2.1±1.15
156	±131	NE	COA		High density (34.3±5.67 N/0.1 sq.m.)	Phoxocephalid		2.1±1.15
156	±131	NE	COA		High density (111±32 N/0.1 sq.m.)	Crustacea		2.1±1.15
157	±321	*	COA		Low species richness (51.2±8.6 S/0.1 sq.m.)	Benthic species		
160	±54.5	NE	COA		High species richness (70.9±16.9 S/0.1 sq.m.)	Benthic species	VAR	
160	±85.4	*	COA	14-d	Significantly toxic (reduced growth rate)	*Chromadorina germanica* (nematode)		
160			PEL					
163	±117	NC	COA	10-d	Moderately toxic (33.8±4.7% mortality)	*Rhepoxynius abronius* (amphipod)	ADT	
164	±91.4	NC	COA	48-h	Moderately toxic (59.4±11.3% abnormal)	Bivalve	LAR	
183	±23.4	NE	COA		High abundance (944±101 N/0.1 sq.m.)	Benthic species	VAR	
192	±104	NE	COA	10-d	Not significantly toxic (11.9±5.36% mortality)	*Rhepoxynius abronius* (amphipod)	VAR	
195	±93.9	NE	COA	10-d	Least toxic (18±6.6% mortality)	*Rhepoxynius abronius* (amphipod)	ADT	
197	±136	SG	COA		Low abundance (415±94.4 N/0.1 sq.m.)	Benthic species	VAR	
201	±349	*	COA		Low abundance (6.1±7.2 N/0.1 sq.m.)	Echinoderm		
203	±97.3	NE	COA	10-d	Not significantly toxic (18.4±6.8% mortality)	*Rhepoxynius abronius* (amphipod)	ADT	
211	±95.5	*	COA		Low abundance (2.4±5.65 N/0.1 sq.m.)	Echinoderms	VAR	
227	±94.3	*	COA		Low abundance (13.8±11.4 N/0.1 sq.m.)	Amphipod	VAR	
227	±94.3	*	COA		Low abundance (5.87±7.06 N/0.1 sq.m.)	Phoxocephalids	VAR	
255		NE	COA		Not toxic (<3% mortality)	*M. elongata* (mysid)		
260		*	AETA		1988 Puget Sound AET	Benthic community		
260		*	SQG		Chemical Criteria	Benthic community		1
260	±58.5	NE	COA	10-d	Not significantly toxic (23.3±6.14% mortality)	*Rhepoxynius abronius* (amphipod)	ADT	1.65±0.738

sediment-associated chromium ($mg \cdot kg^{-1}$ dry weight) in marine and estuarine systems.

AVS ($\mu mol \cdot g^{-1}$)	Total Sulphide ($mg \cdot kg^{-1}$)	TFS ($mg \cdot L^{-1}$)	Unionized Ammonia ($mg \cdot L^{-1}$)	Sand (%)	Silt (%)	Clay (%)	Area	Reference
							San Francisco Bay, CA	Long and Morgan 1990
21.6±24.5				34.3±26.5	37.6±18.2	25.8±13.6	Long Island Sound, NY, CT	Bricker et al. 1993
17.7±26.3				43.6±30.1	34.4±19.2	19.4±12.8	Long Island Sound, NY, CT	Bricker et al. 1993
4.67±6.99				30.3±35.6	41.5±23.7	26.1±13.6	Long Island Sound, NY, CT	Bricker et al. 1993
22.9±23.7				34±26.6	35.3±17.3	25.9±13.2	Long Island Sound, NY, CT	Bricker et al. 1993
29.6±46.3				61.7±45.4	27.7±34.6	10.5±11.2	Long Island Sound, NY, CT	Bricker et al. 1993
6.41				98.8	0.8	0.4	Chesapeake Bay, VA, MD	Hall et al. 1992
6.41				98.8	0.8	0.4	Chesapeake Bay, VA, MD	Hall et al. 1992
6.41				98.8	0.8	0.4	Chesapeake Bay, VA, MD	Hall et al. 1992
6.41				98.8	0.8	0.4	Chesapeake Bay, VA, MD	Hall et al. 1992
25.4±25.5				34±26.2	36.1±16.6	24.4±12.7	Long Island Sound, NY, CT	Bricker et al. 1993
10.5±13.4				15±20.1	57.7±9.83	26.9±10.7	Long Island Sound, NY, CT	Bricker et al. 1993
				21.8±4.26	41±10.3	25.3±18.5	Gulf of Mexico	Chapman et al. 1991
43.4±35.6				20.8±21	49.9±16.3	27.6±9.62	Long Island Sound, NY, CT	Bricker et al. 1993
							Pascagoula Mississippi Channel, MS	Parrish 1987c
							Pascagoula Mississippi Channel, MS	Parrish 1987c
							Pascagoula Mississippi Channel, MS	Parrish 1987c
				7.47	53.8	38.7	San Francisco Bay, CA	Chapman et al. 1987a
40±55.1				3.7±4.1	59.8±6.79	36.4±2.83	Long Island Sound, NY, CT	Bricker et al. 1993
37.6±25.3				21.7±14.8	41.5±12.6	30.3±9.23	Long Island Sound, NY, CT	Bricker et al. 1993
31.8±24.4				28.5±25.1	33.7±15.5	26±10.7	Long Island Sound, NY, CT	Bricker et al. 1993
32.1±28.4				24.2±21.5	40.6±14.9	32.1±11.7	Long Island Sound, NY, CT	Bricker et al. 1993
36.8±34.9				21.3±17.2	52.7±12.2	25.8±8.98	Long Island Sound, NY, CT	Bricker et al. 1993
35.1±25.9				22±17.7	39.2±13.1	32.9±9.51	Long Island Sound, NY, CT	Bricker et al. 1993
34.6±23.9				23.9±17.5	37.1±11.6	32.5±9.93	Long Island Sound, NY, CT	Bricker et al. 1993
				5.8	70.1	24	San Francisco Bay, CA	Chapman et al. 1987a
				5.8±2.5	70.1±26.4	24.1±24.3	San Francisco Bay, CA	Chapman et al. 1987a
				3.3	88.5	8.1	San Francisco Bay, CA	Chapman et al. 1987a
				3.3	88.5	8.1	San Francisco Bay, CA	Chapman et al. 1987a
							San Francisco Bay, CA	Long and Morgan 1990
45.3±21.4				19.4±6.9	38.5±8.7	35±7.15	Long Island Sound, NY, CT	Bricker et al. 1993
				4.6	85.3	10.1	San Francisco Bay, CA	Chapman et al. 1987a
	318±222						Palos Verdes, CA	Swartz et al. 1986
	318±222						Palos Verdes, CA	Swartz et al. 1986
							San Francisco Bay, CA	Long and Morgan 1990
42.3±19.2				17.5±3.03	36.4±8.55	34.3±1.53	Long Island Sound, NY, CT	Bricker et al. 1993
							Hudson-Raritan Bay, NY	Tietjen and Lee 1984
				41.3			Southern California	Word and Mearns 1979
							San Francisco Bay, CA	Long and Morgan 1990
38.1±38.9				20.6±18.3	56.6±10.8	22.7±9.8	Long Island Sound, NY, CT	Bricker et al. 1993
							San Francisco Bay, CA	Long and Morgan 1990
	196±219	0.463±0.16	0.01±0.005	26.6±10.2	61.4±11.1	12±6.88	Palos Verdes, CA	Swartz et al. 1985a
	196±219	0.463±0.16	0.01±0.005	26.6±10.2	61.4±11.1	12±6.88	Palos Verdes, CA	Swartz et al. 1985a
	196±219	0.463±0.16	0.01±0.005	26.6±10.2	61.4±11.1	12±6.88	Palos Verdes, CA	Swartz et al. 1985a
	196±219	0.463±0.16	0.01±0.005	26.6±10.2	61.4±11.1	12±6.88	Palos Verdes, CA	Swartz et al. 1985a
							Southern California	Word and Mearns 1979
	186±145			20.8±12.5			Palos Verdes, CA	Ferraro et al. 1991
							Hudson-Raritan Bay, NY	Tietjen and Lee 1984
							San Francisco Bay, CA	Long and Morgan 1990
							San Francisco Bay, CA	Long and Morgan 1990
	145±27.8			23.1±6.74			Palos Verdes, CA	Ferraro et al. 1991
	308±371			19.5±12.1			Palos Verdes, CA	Ferraro et al. 1991
							San Francisco Bay, CA	Long and Morgan 1990
	407±456			17.3±14.8			Palos Verdes, CA	Ferraro et al. 1991
				57.4			Southern California	Word and Mearns 1979
							San Francisco Bay, CA	Long and Morgan 1990
	351±379			16.1±8.02			Palos Verdes, CA	Ferraro et al. 1991
	393±397			16.2±8.78			Palos Verdes, CA	Ferraro et al. 1991
	393±397			16.2±8.78			Palos Verdes, CA	Ferraro et al. 1991
							San Diego Bay, CA	Salazar et al. 1980
							Puget Sound, WA	PTI 1988
							Puget Sound, WA	WDE 1989
	182±135			23.2±21.8	33.5±9.86	43.2±13.4	Oakland Harbor, CA	Word et al. 1988

Appendix Ib. Summary of the available biological effects and related physicochemical data for

Chromium (mg·kg^{-1})		Hit	Analysis Type	Test Type	Endpoint Measured	Species	Life Stage	TOC (%)
270		*	AETA	10-d	1988 Puget Sound AET	*Rhepoxynius abronius* (amphipod)	ADT	
270	±63.6	NE	COA	10-d	Not significantly toxic (10.4±6.3% mortality)	*Nephtys caecoides* (polychaete)		1.63±0.657
270	±63.6	NE	COA	10-d	Not significantly toxic (0.2±0.6% mortality)	*Macoma nasuta* (clam)		1.63±0.657
280		*	AETA	48-h	San Francisco Bay AET	Oyster; mussel	LAR	
293	±459	NE	COA		High abundance (88.9±35.4 N/0.1 sq.m.)	Benthic species		
297	±302	NE	COA		High density (594±688 N/0.1 sq.m.)	Mollusca		2.53±1.27
297	±302	NE	COA	10-d	Not significantly toxic (8±5.7% mortality)	*Rhepoxynius abronius* (amphipod)	ADT	2.53±1.27
300		NE	COA		Not toxic (<3% mortality)	*Protothaca staminea* (clam)		
300		NE	COA		Not toxic (<3% mortality)	*Citharichthys stigmaeus* (sanddab)		
300		NE	COA		Not toxic (<3% mortality)	*Neanthes arenaceodentata* (polychaete)		
300		NE	COA		Not toxic (<3% mortality)	*M. elongata* (mysid)		
305		NE	COA		High density (2177 N/0.1 sq.m.)	Polychaeta		3.2
308	±78.4	SG	COA	10-d	Significantly toxic (35.3±2.5% mortality)	*Rhepoxynius abronius* (amphipod)	ADT	1.56±0.07
310		*	AETA		California AET Values	Benthic species		
310		*	AETA		Southern California AET Values	Benthic species		
335	±180	NE	COA	48-h	Least toxic; TLm	*Fundulus heteroclitus* (mummichog)		
335	±180	NE	COA	48-h	Least toxic; TLm	*Leiostomus xanthurus* (spot)		
335	±226	NE	COA	10-d	Not significantly toxic (9.5±5.45% mortality)	*Rhepoxynius abronius* (amphipod)	ADT	2.42±0.88
369		*	COA	14-d	Significantly toxic (100% mortality)	*Nereis virens* (sandworm)		
370		*	AETA	10-d	San Francisco Bay AET	*Rhepoxynius abronius* (amphipod)	ADT	
383	±215	*	COA		Low density (0.2±0.14 N/0.1 sq.m.)	Echinoderm		2.64±0.77
416		*	COA		Low species richness (19.2 S/0.1 sq.m.)	Benthic species	VAR	
482	±330	SG	COA		Low density (365±178 N/0.1 sq.m.)	Polychaeta		3.15±1.32
523	±279	*	COA		Low density (0.03±0.08 N/0.1 sq.m.)	Echinoderm		3.53±0.742
523	±279	*	COA		Low density (0.233±0.367 N/0.1 sq.m.)	Foraminifera (sponge)		3.53±0.742
523	±279	*	COA		Low density (0.3±0.8 N/0.1 sq.m.)	Ophiuroidea (brittle star)		3.53±0.742
527	±120	*	COA		Low density (5.3±3.7 N/0.1 sq.m.)	Amphipod		3.13±0.15
527	±120	*	COA		Low density (0.13±0.23 N/0.1 sq.m.)	Phoxocephalid		3.13±0.15
669	±173	*	COA		Low density (21.5±11.6 N/0.1 sq.m.)	Mollusca		4±0.265
669	±173	*	COA	10-d	Significantly toxic (21±6.3% mortality)	*Rhepoxynius abronius* (amphipod)	ADT	4±0.265
682	±144	*	COA		Low species richness (26±11.5 S/0.1 sq.m.)	Macro benthos		3.95±0.238
682	±144	*	COA		Low density (1±1.2 N/0.1 sq.m.)	Amphipod		3.95±0.238
682	±144	*	COA		Low density (0 N/0.1 sq.m.)	Phoxocephalid		3.95±0.238
682	±144	*	COA		Low density (8.7±6.01 N/0.1 sq.m.)	Crustacea		3.95±0.238
1646	±1628	*	COA	48-h	Most toxic; TLm	*Fundulus heteroclitus* (mummichog)		
1646	±1628	*	COA	48-h	Most toxic; TLm	*Leiostomus xanthurus* (spot)		

sediment-associated chromium (mg·kg⁻¹ dry weight) in marine and estuarine systems.

AVS (μmol·g⁻¹)	Total Sulphide (mg·kg⁻¹)	TFS (mg·L⁻¹)	Unionized Ammonia (mg·L⁻¹)	Sand (%)	Silt (%)	Clay (%)	Area	Reference
							Puget Sound, WA	PTI 1988
	196±131			23.4±19.5	33.1±8.9	43.4±12	Oakland Harbor, CA	Word et al. 1988
	196±131			23.4±19.5	33.1±8.9	43.4±12	Oakland Harbor, CA	Word et al. 1988
							San Francisco Bay, CA	Long and Morgan 1990
							Southern California	Word and Mearns 1979
	275±858	0.47±0.137	0.009±0.002	22.5±11.7	63.3±9.75	14.2±7.21	Palos Verdes, CA	Swartz et al 1985a
	275±585	0.47±0.137	0.009±0.002	22.5±11.7	63.3±9.75	14.2±7.21	Palos Verdes, CA	Swartz et al. 1985a
							San Diego Bay, CA	Salazar et al. 1980
							San Diego Bay, CA	Salazar et al. 1980
							San Diego Bay, CA	Salazar et al. 1980
							San Diego Bay, CA	Salazar et al. 1980
	443	0.65	0.006	23.6	56.5	19.9	Palos Verdes, CA	Swartz et al. 1985a
	249±112			24.1±6.74	31.6±6.75	44.3±3.69	Oakland Harbor, CA	Word et al. 1988
							California	Becker et al. 1990
							Southern California	Becker et al. 1990
							Baltimore Harbor, MD	Tsai et al. 1979
							Baltimore Harbor, MD	Tsai et al. 1979
	946±719						Palos Verdes, CA	Swartz et al. 1986
							Black Rock Harbor, CT	Simmers et al. 1984
							San Francisco Bay, CA	Long and Morgan 1990
	1090±700						Palos Verdes, CA	Swartz et al. 1986
	1164			10.6			Palos Verdes, CA	Ferraro et al. 1991
	1381±1098	6.99±8.97	0.009±0.004	21.1±9.96	64.4±7.44	14.6±5.43	Palos Verdes, CA	Swartz et al. 1985a
	1451±1003	7.04±8.92	0.008±0.006	18.7±6.03	64.9±6.69	16.5±4.79	Palos Verdes, CA	Swartz et al. 1985a
	1451±1003	7.04±8.92	0.008±0.006	18.7±6.03	64.9±6.69	16.5±4.79	Palos Verdes, CA	Swartz et al. 1985a
	1451±1003	7.04±8.92	0.008±0.006	18.7±6.03	64.9±6.69	16.5±4.79	Palos Verdes, CA	Swartz et al. 1985a
	1573±247						Palos Verdes, CA	Swartz et al. 1986
	1573±247						Palos Verdes, CA	Swartz et al. 1986
	2277±301	13.6±8.45	0.009±0.003	20±6.32	63.2±4.74	16.8±1.59	Palos Verdes, CA	Swartz et al. 1985a
	2277±301	13.6±8.45	0.009±0.003	20±6.32	63.2±4.74	16.8±1.59	Palos Verdes, CA	Swartz et al. 1985a
	2035±542	10.3±9.5	0.008±0.005	17.6±7.12	64.6±4.77	17.8±2.48	Palos Verdes, CA	Swartz et al. 1985a
	2035±542	10.3±9.5	0.008±0.005	17.6±7.12	64.6±4.77	17.8±2.48	Palos Verdes, CA	Swartz et al. 1985a
	2035±542	10.3±9.5	0.008±0.005	17.6±7.12	64.6±4.77	17.8±2.48	Palos Verdes, CA	Swartz et al. 1985a
	2035±542	10.3±9.5	0.008±0.005	17.6±7.12	64.6±4.77	17.8±2.48	Palos Verdes, CA	Swartz et al. 1985a
							Baltimore Harbor, MD	Tsai et al. 1979
							Baltimore Harbor, MD	Tsai et al. 1979