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Ecological Risk Assessment of Open-Water Sediment Disposal to Support the Management of Freshwater Dredging Projects

Ministère du Développement durable, de l'Environnement,
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Foreword

In 1998, the St. Lawrence Vision 2000 Action Plan (Phase III) adopted a navigation component, with the goal of harmonizing commercial and recreational navigation practices in order to minimize their environmental impacts on the aquatic ecosystems of the St. Lawrence, specifically with reference to dredging and sediment management. The Navigation Consensus-building Committee (NCC) was established with the mandate to establish and implement a sustainable navigation strategy, including integrated management of dredged sediments (EC and MDDEP 2004). To date, the work carried out by the NCC's dredging working groups has helped improve communication and coordination among the various partners involved in or affected by dredging activities (governments, the shipping industry and related business sectors, community groups), in particular by reducing disparities arising from shared jurisdiction (federal and provincial) for certain areas, by improving the efficiency of the authorization processes, and by ensuring that public interests receive due consideration early in the process.

In Quebec, criteria for assessing the chemical quality of sediments constitute the first tool for assessing the potential environmental effects of chemical contamination present in sediments. The *Interim Criteria for Quality Assessment of St. Lawrence River Sediment* (SLC and MENV 1992) were introduced in 1992 and were used as a dredging management tool. These criteria were reviewed as part of the activities of the St. Lawrence Plan, culminating in the publication, in January 2008, of the *Criteria for the Assessment of Sediment Quality in Quebec and Application Frameworks: Prevention, Dredging and Remediation* (EC and MDDEP 2007). This document outlines the fundamental principles on which the new quality criteria are based, as well as the threshold values adopted for 33 chemicals and how these values should be applied in a particular management context. However, beyond applying the criteria, there was still work to be done to fully incorporate the various tools for assessing sediment quality. For instance, it was necessary to determine the type of studies required in each management context in order to complete the assessments of sediment contamination and its associated ecological risks. More detailed guidance was also required concerning the interpretation of results and how they should be used in the decision-making process. The ecological risk assessment approach described in this document thus provides additional information to help fine-tune the environmental analysis process for managing dredged sediments, thereby improving the decision-making process.

A detailed description of the approach followed during this study is presented in related articles are published in peer-reviewed journals (Desrosiers et al. 2008, 2010, 2012; Masson et al. 2010).

Résumé

Il est reconnu que les activités de dragage peuvent, entre autres, engendrer des modifications du régime hydrologique et avoir des effets négatifs sur les habitats fauniques. En présence de sédiments contaminés, une gestion inappropriée peut entraîner des risques significatifs pour l'environnement. La plupart des projets de dragage doivent par conséquent faire l'objet d'une évaluation environnementale avant leur réalisation afin d'assurer la protection de l'environnement et d'optimiser la gestion des sédiments. En fonction du degré de contamination, le cadre de gestion actuel prévoit le recours à des outils d'évaluation complémentaires pour juger du risque écotoxicologique associé aux sédiments contaminés. Ainsi, la démarche d'évaluation du risque écotoxicologique (ERE) qui a été élaborée permet d'affiner le cheminement de l'analyse concernant l'évaluation du rejet en eau libre des déblais de dragage et d'améliorer le processus de prise de décision. L'évaluation doit donner une réponse à la question suivant : « Dans le contexte d'un projet de dragage spécifique, est-il acceptable de rejeter les sédiments dragués en eau libre ? »

Cette démarche d'ERE a été établie à l'aide d'une étude écotoxicologique relative au fleuve Saint-Laurent comprenant la collecte et la caractérisation d'échantillons en milieu naturel, et utilisant des données de la littérature concernant le fleuve. Afin d'obtenir une caractérisation chimique, toxicologique et biologique des sédiments, deux campagnes d'échantillonnage ont été réalisées : la première à l'automne 2004 et la seconde à l'automne 2005. Lors de la sélection des stations d'échantillonnage, les zones de sédimentation (lacs fluviaux, zones portuaires, embouchures de tributaires, etc.) ont été favorisées, puisqu'il est reconnu que les plus fortes teneurs en contaminants sont associées à ces zones en raison de l'accumulation de particules fines qui s'y produit. La démarche d'ERE est constituée de deux étapes. L'étape 1 décrit la procédure de dépistage du niveau de contamination des sédiments qui compare les résultats des analyses chimiques aux critères d'évaluation de la qualité des sédiments établis pour le Québec. Lorsqu'elle est requise, l'étape 2 décrit la procédure d'évaluation de la toxicité des sédiments d'eau douce à partir d'essais de toxicité réalisés en laboratoire (mortalité et croissance de *Hyalella azteca* et *Chironomus riparius*). Selon les résultats de l'ERE, deux options de gestion sont possibles : 1) les sédiments peuvent être rejetés en eau libre ou être utilisés à d'autres fins, par exemple pour des usages bénéfiques tels que la création d'habitats fauniques ou le rechargement de plages, dans la mesure où le dépôt ne contribue pas à détériorer le milieu récepteur ou; 2) le dépôt en eau libre est proscrit et il faudra choisir une autre option de gestion. L'élaboration de la démarche d'ERE de même que ses règles d'application sont présentées dans le présent document.

Abstract

Dredging activities can have a number of potential impacts, such as changes in the hydrological regime and adverse effects on wildlife habitats. Inappropriate management of contaminated sediments may also lead to significant environmental risk. Most dredging projects must therefore undergo an environmental assessment before they are carried out, in order to protect the environment and optimize sediment management. Depending on the degree of contamination, the current management framework includes the use of additional assessment tools to evaluate the ecological risk associated with contaminated sediments. Hence, the ecological risk assessment (ERA) approach presented in this document fine-tunes the analytical process surrounding the assessment of open-water disposal of dredged material and improves the decision-making process. The assessment must answer the following question: “In the context of a specific dredging project, is open-water disposal of dredged sediments acceptable?”

This ERA approach was developed using an ecotoxicological study on the St. Lawrence River that collected and characterized samples in the natural environment and incorporated data from literature on the St. Lawrence River. In order to obtain a chemical, toxicological and biological characterization of the sediments, two sampling campaigns were carried out: the first in the fall of 2004 and the second in the fall of 2005. During selection of the sampling stations, areas of sedimentation (fluvial lakes, port areas, mouths of tributaries, etc.) were preferred, since these areas are recognized as having the highest contaminant levels due to the accumulation of fine particles. The ERA approach is composed of two tiers. Tier 1 describes the procedure for detecting the level of sediment contamination by comparing the results of the chemical analyses to the sediment quality assessment criteria established for Quebec. Tier 2, when required, describes the procedure for assessing the toxicity of freshwater sediments based on laboratory toxicity tests (mortality and growth of *Hyalella azteca* and *Chironomus riparius*). Depending on the results of the ERA, there are two possible management options: (1) the sediments can be disposed of in open water or can be used for other purposes, for example for beneficial uses such as the creation of wildlife habitats or beach replenishment, provided that the receiving environment is not thereby adversely affected; or (2) open-water disposal is prohibited and another management option must be chosen. This document presents the process by which the ERA approach was developed as well as rules for applying this approach.

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1.0 ERA Application Context

The ecological risk assessment (ERA) approach presented in this document was developed for the freshwater portion of the St. Lawrence River, which extends from Cornwall, Ontario, to the eastern tip of Île d'Orléans, and includes Lake Saint-François, Lake Saint-Louis and Lake Saint-Pierre. Because these three fluvial lakes have a relatively low average depth—5.7 m, 3.4 m and 2.7 m respectively; because the fluvial section between Montréal and Sorel is also shallow; and because there are rapids at several locations, excavation work has been carried out since the mid-19th century to deepen and widen the navigation channel. Specifically, between 1844 and 1999, the depth of the navigation channel between Montréal and Québec was increased from 4.0 m to 11.3 m, and the width was expanded from 46 m to 240 m. At the same time, numerous locks were constructed in the rapids sections. This helped make the St. Lawrence River one of the largest navigable waterways in the world—one that allows ocean-going vessels to reach the heart of the North American continent (SLC 1996; EC and MDDEP 2007).

Sedimentation in the St. Lawrence is a natural and dynamic process, generally determined by the quantity of suspended solids carried by the St. Lawrence River and the hydrodynamic conditions near the shores and in the water column. The flow rate of the St. Lawrence River varies with the seasons, resulting in significant fluctuations in water levels, primarily between Lake Saint-Louis and Lake Saint-Pierre. These fluctuations in levels, combined with strong currents, contribute to significant bank and shoreline erosion, primarily in the Île de Boucherville, Île de Verchères, Île de Contrecœur and Sorel Delta sector. Consequently, this erosion generates very high sediment inputs of more than 4600 tonnes per year, compared to just under 2300 tonnes per year originating from the tributaries (Rondeau et al. 2000). The largest quantities of materials are carried during spring freshets, when the flow rates and hydraulic forces that govern the mechanisms of erosion are at their maximum. This particle load circulating in the St. Lawrence tends to settle in various locations as a function of the prevailing hydrodynamic conditions. Protected areas, particularly navigation channels and port areas constructed to provide shelter to commercial vessels or pleasure craft during berthing, are some of the main areas where sediments tend to accumulate in the St. Lawrence River. These areas must therefore be regularly maintained in order to preserve their intended uses. In addition, as a result of industrial development, the expansion of agriculture and the population explosion in the last century, the St. Lawrence has received significant inputs of organic and inorganic contaminants. These contaminants combine with sediment inputs from the Great Lakes and from the tributaries of the St. Lawrence River and eventually end up in the sedimentary basins of the fluvial lakes as well as in the sediment accumulation areas.

Approximately 400 000–600 000 m³ of sediments are dredged annually from the St. Lawrence system, including the St. Lawrence River, the St. Lawrence Estuary, Chaleur Bay and the area around the Magdalen Islands (EC 1993; MTQ 2003). Approximately 45% of this volume is dredged in the freshwater section of the St. Lawrence River (Lalancette 2001). This mainly involves maintenance dredging work carried out to keep the navigation channels and ports open for vessels. Other types of work may also require dredging, such as excavation during marine engineering work (e.g. wharf construction) or remediation of a contaminated aquatic environment. However, dredging activities can have a number of potential impacts, such as changes in the hydrological regime and adverse effects on wildlife habitats. In the case of contaminated sediments, inappropriate management can lead to significant ecological risks.

Most dredging projects must therefore undergo an environmental assessment before they are carried out in order to protect the environment and optimize the management of dredged sediments. In addition, it is important to ensure that these projects are all dealt with in the same way, regardless of the federal or provincial regulatory framework to which they may be subject.

In Quebec, outside the area covered by the *Disposal at Sea Regulations*,[2] the first tool for assessing the chemical quality of sediments consisted of a series of quality criteria developed for metals (cadmium, chromium, copper, mercury, nickel, lead and zinc), one metalloid (arsenic) and various organic contaminants: polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), dioxins and furans, organochlorine pesticides, and nonylphenol and its ethoxylates. The criteria presented in the *Criteria for the Assessment of Sediment Quality in Quebec and Application Frameworks: Prevention, Dredging and Remediation* (EC and MDDEP 2007) have been applied since 2008. They include one set of quality criteria for freshwater sediments and another for marine sediments. These criteria replace the interim criteria that had been in effect since 1992. These new criteria were developed based on the Biological Effects Database for Sediments and the approach adopted by the Canadian Council of Ministers of the Environment. Only the criteria applicable to the management of sediments in freshwater are considered here.

A summary of the sediment management frameworks currently in effect in Quebec and the management options (EC and MDDEP 2007) is provided in Text Box 1. Depending on the degree of sediment contamination and the particular management context, the current protocol provides for the use of additional assessment tools to determine the ecological risk associated with contaminated sediments. For the management of sediments resulting from dredging work, two critical thresholds are used for the initial assessment of the degree of sediment contamination: the occasional effect level (OEL) and the frequent effect level (FEL). The OEL constitutes the threshold above which dredged sediments contaminated by one or more chemicals cannot be disposed of in the aquatic environment unless it has first been demonstrated that these sediments will not adversely affect the receiving environment. When the concentration of a substance is higher than the FEL, disposal of the sediments in open water is prohibited.

To determine whether open-water disposal is a possible management option when dredged sediments contain a contaminant whose concentration is higher than the OEL but lower than or equal to the FEL, chemical analyses must be supplemented by toxicity tests aimed at more clearly identifying the ecological risk associated with these sediments.

[2] The *Disposal at Sea Regulations* prescribe quality criteria for assessing the chemical quality of sediments in the area covered by the Disposal at Sea Program. In this area, the marine sediment quality guidelines of the Canadian Council of Ministers of the Environment are also used.

The use of ERA as a decision support tool for managing contaminated sediments is an internationally accepted practice. The ERA approach described in this document makes it possible to fine-tune the analytical process and improve decision making with respect to the management of dredged material. This document examines only the management framework relating to maintenance dredging or dredging for marine engineering projects for which open-water disposal of the dredged sediments is being considered (Text Box 1; Figure 1-1 – lines indicated in bold). The question that the assessment must answer can be stated as follows: "In the context of a specific dredging project, is open-water disposal of dredged sediments acceptable?"

Text Box 1: Sediment management frameworks in Quebec (EC and MDDEP 2007)

Prevention of sediment contamination (Part A of Figure A-1)

This management framework aims to prevent any sediment contamination caused by a new input of contaminants into a water body. This therefore requires that the status of vulnerable sites be monitored in order to provide advance warning of incipient contamination. In this context, the actions to be taken are aimed at controlling the sources in order to avoid increasing the contamination or to avoid a new input of contaminants.

Maintenance dredging and marine engineering projects (Part B of Figure A-1)

Dredging is often necessary in order to keep navigation channels open (maintenance dredging), improve port infrastructures or construct additional infrastructures (marine engineering projects). In these cases, various options are available for managing the sediments resulting from this type of work. The management method chosen must ensure that the sediments do not pose a threat to aquatic or terrestrial biota. Open-water disposal of dredged sediments can therefore be considered a valid option only if it has been demonstrated, for example through appropriate toxicity testing, that the sediments have no adverse effects on the receiving environment. In cases where open-water disposal is prohibited, the sediments must be treated or safely contained. The sediment management option that is chosen must be the one that entails the least impact on the environment, while also being technically and economically feasible, regardless of the level of sediment contamination. In analyzing the options, the beneficial use of sediments in a terrestrial or aquatic environment must also be considered.

It should also be pointed out that the management of dredged sediments in the marine environment is subject to the provisions of Part 7, section 3, of the *Canadian Environmental Protection Act, 1999* (CEPA 1999), while in the terrestrial environment, this activity is governed by Quebec's Soil Protection and Contaminated Sites Rehabilitation Policy as well as the management grid applicable to that policy (MEF 1998).

Remediation of contaminated sites (Part C of Figure A-1)

Remediation of a contaminated aquatic site is justified by the fact that the sediments can pose a serious threat to the integrity of the ecosystem and to the organisms that live in it. Environmental studies (toxicity tests, biological studies and assessment, etc.) may be required to supplement the assessment of the contamination, evaluate the risk and determine the remediation requirements. It is necessary to determine the feasibility of the remediation process, set the priorities for action and identify the environmental gains. The decision to remediate a contaminated site is generally made after an in-depth analysis concludes that the advantages of restoring the site outweigh the disadvantages.

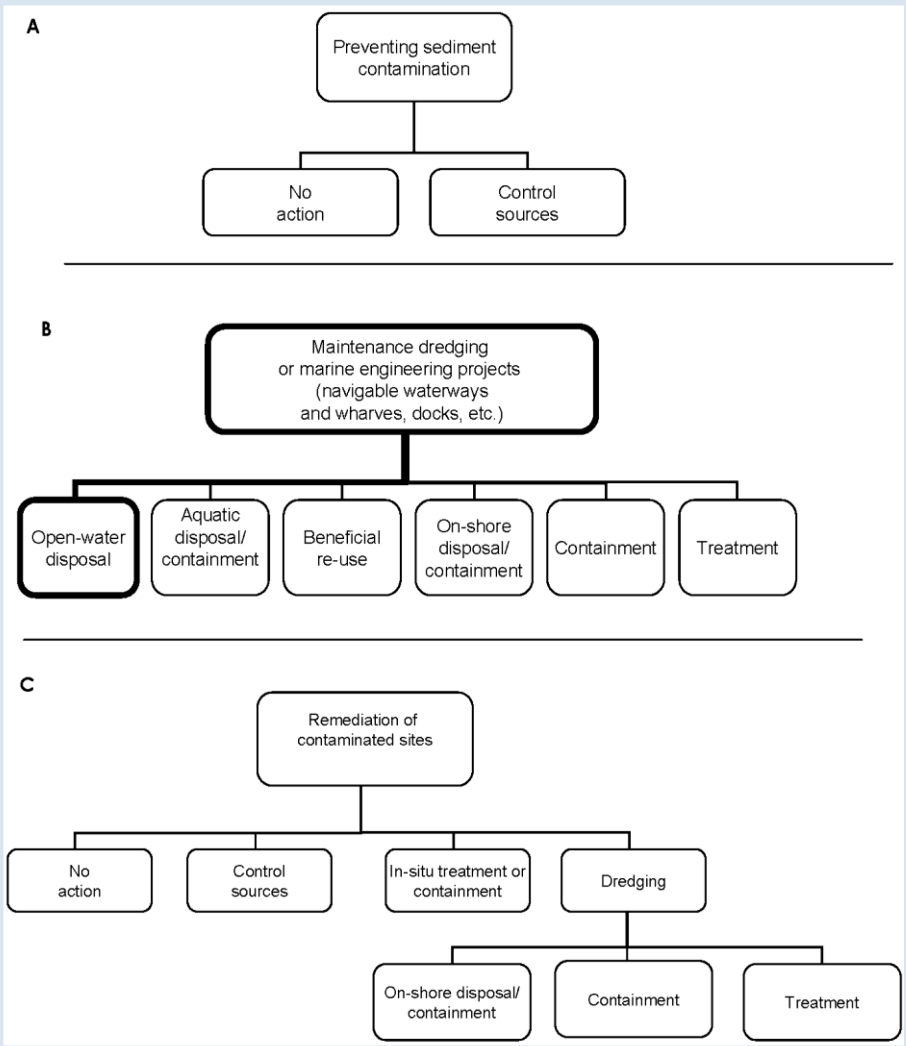

Figure 1-1: Synopsis of management options available for
(A) preventing contamination;
(B) maintenance dredging and marine engineering projects; and
(C) remediation of contaminated sites.

2.0 Development of the ERA Approach

Generally, the objective of an ERA is to assess the ecological risk associated with a given problem or project. The assessment may cover one or more sites contaminated by one or more contaminants and suggest priorities for action. An ERA makes it possible, first of all, to identify problem situations in a specific context defined based on conservative scenarios. This process can lead to three possible conclusions: (1) that there is no risk and that the assessment is complete, (2) that more detailed assessments are required, or (3) that the management options being considered must be modified (CEAEQ 1998).

Numerous ERA frameworks applicable to contaminated sediments have been published, either for dredging activities or remediation, including Babut et al. (2006), Chapman (2005), and EC and OME (2007). In addition, in Quebec, there is an ecological risk assessment procedure for contaminated lands (CEAEQ 1998). From a decision-making standpoint, the published assessment frameworks are generally based on tiered approaches intended to optimize the resources invested, taking into account any uncertainty surrounding the available knowledge and data. If the uncertainty is considered too great at a given tier, it is necessary to move on to the next tier, where more comprehensive investigations will be conducted. These are the principles that guided the development of the ERA approach presented here.

For dredging projects carried out in Quebec outside the disposal at sea area, the *Criteria for the Assessment of Sediment Quality in Quebec* stipulate that, in order to determine the possible management options for dredged sediments, chemical analyses must be supplemented by toxicity tests when the concentration of one of the contaminants listed in the quality criteria and present in the sediments falls between the OEL and the FEL (EC and MDDEP 2007).

In the context of this study, a detailed analysis was carried out on the ability of the threshold values (OEL and FEL) to predict sediment toxicity, compared to the results of several toxicity tests; this analysis provided the foundation for the development of the tiers in our ERA approach.

2.1 Data Collection

The ERA approach was developed using an ecotoxicological study on the St. Lawrence River that collected and characterized samples in the natural environment and incorporated data from the literature on managing St. Lawrence River sediments.

In order to obtain a chemical, toxicological and biological characterization of the sediments, two sampling campaigns were carried out: the first in the fall of 2004 and the second in the fall of 2005. During selection of the sampling stations, areas of sedimentation in the fluvial section of the St. Lawrence (fluvial lakes, port areas, mouths of tributaries, etc.) were preferred, since previous sampling campaigns have shown that these areas have the highest contaminant levels due to the accumulation of fine particles.

Sediments and macroinvertebrates were sampled in 59 stations located in the fluvial lakes (Saint-François, Saint-Louis and Saint-Pierre) and in the Montréal island area (Figure 1) in order to determine the following variables:

- inorganic contaminants (aluminum, arsenic, calcium, cadmium, chromium, copper, iron, mercury, manganese, nickel, lead and zinc)
- organic contaminants (PCBs, PAHs, petroleum hydrocarbons C_{10}–C_{50}, pesticides and organotins)
- particle size distribution, nutrients and organic matter content of the sediments
- two pore water toxicity tests (*Pseudokirchneriella subcapitata* and *Brachionus calyciflorus*)
- two whole-sediment toxicity tests (*Chironomus riparius* and *Hyalella azteca*)
- the taxonomic and functional structure of the macroinvertebrate community

All the chemical analyses and toxicity tests were performed in the Centre d'expertise en analyse environnementale du Québec (CEAEQ) laboratories according to standardized methods and with the requisite quality controls. The analytical methods are briefly presented in Appendix A.

The identification of benthic organisms was carried out by Laboratoires SAB Inc. in Longueuil, Quebec. Several descriptors of the benthic communities were then defined; examples include the abundance (or presence) of genera or families, community structure indices (e.g. abundance index, richness index) and biotic tolerance indices (e.g. the Hilsenhoff Biotic Index). Functional traits were also determined.

A detailed description of the approach followed as well as all the information compiled during this study are presented in additional articles published in peer-reviewed journals (Desrosiers et al., 2008, 2010, 2012; Masson et al., 2010).

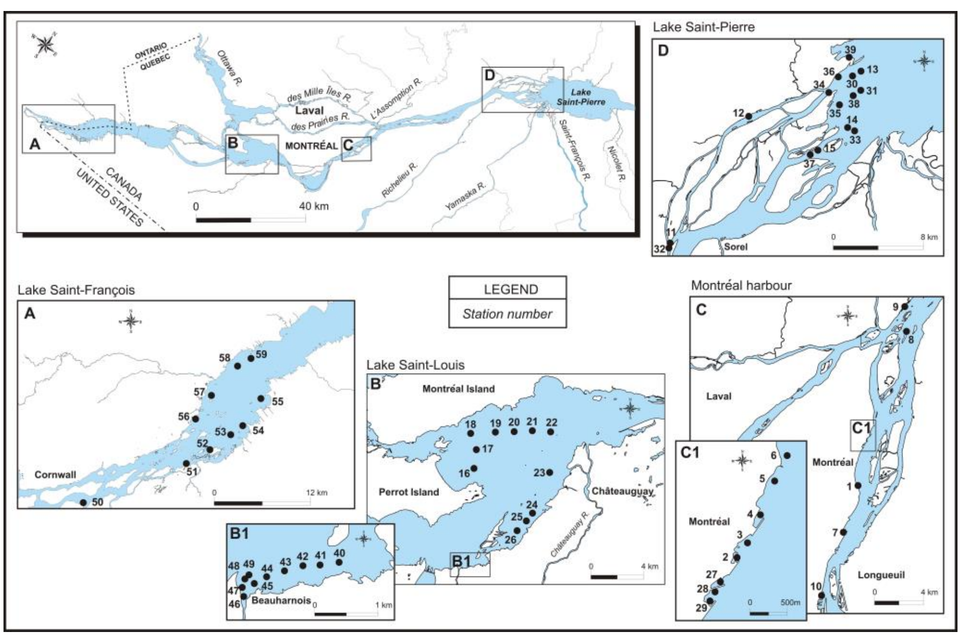

Figure 1: Location of the stations sampled in Lake Saint-François, Lake Saint-Louis, Lake Saint-Pierre and Montréal harbour

2.2 The Conceptual Model

The first phase of the ERA involved the development of the conceptual ecotoxicological model of the study site and surrounding area. This model consists of a representation of the environmental system under study, including the chemical, physical and biological processes that determine the transformation of contaminants as well as contaminant transport from the source of contamination to the receiving environments. This step is essential to determining the routes of exposure and the potential ecotoxicological responses of the receptors (CEAEQ 1998).

In this section, we present the generic conceptual model developed to cover the open-water disposal of dredged sediments in general. All aspects of the problem are considered, i.e., sources of exposure, biological targets and anticipated effects. Section 2.2.1 presents the sources of stress associated with the impacts of open-water disposal, while the description of the receiving ecosystem, the identification of the biological targets and the anticipated responses are provided in section 2.2.2. Finally, the hypotheses concerning the ecotoxicological impacts of open-water disposal of dredged sediments are presented in section 2.2.3. In these sections, the elements that must be assessed during the ERA approach are presented, as well as additional elements required during the environmental assessment process for the management of dredged sediments.

2.2.1 Analysis of the source of stress: description of the impacts of open-water disposal of dredged sediments

The behaviour of sediments during open-water disposal can be described in five distinct phases: convection (also called convective descent or mass descent), passive diffusion, dynamic collapse, deposit formation and dispersion after resuspension (Truitt 1988; Jacinto et al. 1999). The rapid descent of the sediment mass under the influence of gravity constitutes the first phase of the process, i.e., convection. This convective descent usually continues until the sediments reach the bottom of the water body. However, as a result of the process of passive diffusion, fine particles detach from the mass during descent, disperse and are carried by currents. Dynamic collapse is the interruption of the convective descent when the sediment mass reaches the bottom and spreads out horizontally. A density current then forms radially around the point of impact, carrying the sediments over varying distances depending on the size of the particles. Deposit formation is characterized by the development of a sediment mound. In the absence of hydrodynamic disturbances, this deposit consolidates and becomes increasingly resistant to erosion. However, if the hydrodynamic forces in the receiving environment are sufficiently strong to remobilize the deposited sediments, this will cause erosion of these sediments over the medium to long term and resuspension of the sediment particles. These particles, or suspended solids (SS), will then be dispersed and transported over varying distances until they settle again. Disposal sites may be described as dispersive or non-dispersive depending on the stability of the deposit.

The ecotoxicological impact of open-water disposal must be assessed over the short, medium and long term, given the potential for resuspension and transport of sediments on the bottom over time. First, in order to determine whether open-water disposal poses an ecological risk, analyses of various chemical parameters must be performed and the results compared to the sediment quality criteria (EC and MDDEP 2007). Depending on the results, the use of other assessment tools in addition to the quality criteria, such as toxicity tests, may be necessary.

Even when sediments are not contaminated, it is important to limit the dispersion of SS during dredging or disposal in order to prevent high concentrations of SS from physically affecting organisms, for example by obstructing fish gills, or affecting habitats through accumulation of sediments in spawning grounds downstream. Consequently, the extent of sediment resuspension must be carefully monitored during the work and, if necessary, measures must be taken to limit the increase in SS concentrations in the water column. Guidelines are therefore being developed for managing SS associated with dredging activities and open-water disposal. These guidelines will include criteria for managing SS specific to dredging, taking into account the surface water quality criteria,[3] ambient SS concentrations in the St. Lawrence and the concentrations measured during dredging. While the surface water quality guidelines consist primarily of objectives to follow in the natural environment immediately downstream of effluent discharges,[4] the management criteria for dredging that have been adapted for work in aquatic environments help ensure that best practices are put in place to minimize the impacts of SS on aquatic biota. If monitoring of suspended solids during dredging work is required, the concentrations of suspended sediments will have to be measured before the work begins in order to determine the range of natural concentrations in the affected area. Subsequent environmental monitoring during the work will make it possible to verify that the predefined objectives are being met and to adjust the actions for minimizing the impacts, as necessary.

It is important to assess the potential physical impacts of SS dispersion, erosion of the disposal sites and the disposal of dredged sediments in sensitive areas in order to determine whether the receiving environment is at risk of degradation. These aspects are not included in the ERA approach; however, they must be the subject of further studies on the potential physical disturbances of SS in order to guide the process of selecting an acceptable disposal site. The choice of disposal site can thus be influenced by the presence of sensitive components in nearby natural or human environments.

2.2.2 Analysis of the receiving ecosystem: identification of biological targets and anticipated responses

The receiving ecosystem includes various biological targets, such as the species living in the water column, from phytoplankton to fish, and the species living on and in the sediment, from periphyton to macroinvertebrates. The fish and macroinvertebrates present in juvenile feeding grounds and spawning grounds also represent significant targets in the event of erosion of the initial deposit. Certain birds and mammals that depend on the aquatic environment for food or habitat may also be part of the receiving ecosystem.

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[3] Based on the *Canadian Environmental Quality Guidelines*, and available from the Quebec Ministère du Développement durable, de l'Environnement, de la Faune et des Parcs (MDDEFP) website (www.mddefp.gouv.qc.ca/eau/criteres_eau/index.asp).

[4] Calculation and Interpretation of Effluent Discharge Objectives for Contaminants in the Aquatic Environment, available from the MDDEFP website (www.mddep.gouv.qc.ca/eau/oer/index.htm).

When sediments are released into the water during open-water disposal (mass descent of the sediments and passive diffusion), the conceptual model considers the short-term effects of dredged sediment disposal on the survival of the organisms in the water column (Figure 2 – process A: phytoplankton, zooplankton, fish). During deposit formation, one would obviously expect to see the partial or complete destruction of the existing benthic communities as a result of their being buried under the mass of sediments deposited (Figure 2 – process B). This effect can extend over an area well beyond the boundaries of the release point since, firstly, the sediment mass spreads out horizontally upon impact with the bottom and, secondly, hydrodynamic forces cause erosion and spreading of the deposit over the medium to long term (Figure 2 – process C). This short- and medium-term effect of sediment disposal may be offset by recolonization of the site by adjacent benthic communities. However, the particle size distribution of the dredged material dumped may be different from the sediments naturally found on the bottom, which would affect the potential for recolonization of the site by benthic communities. To promote recolonization, a disposal site with a particle size distribution similar to that of the sediments being disposed of should be chosen.

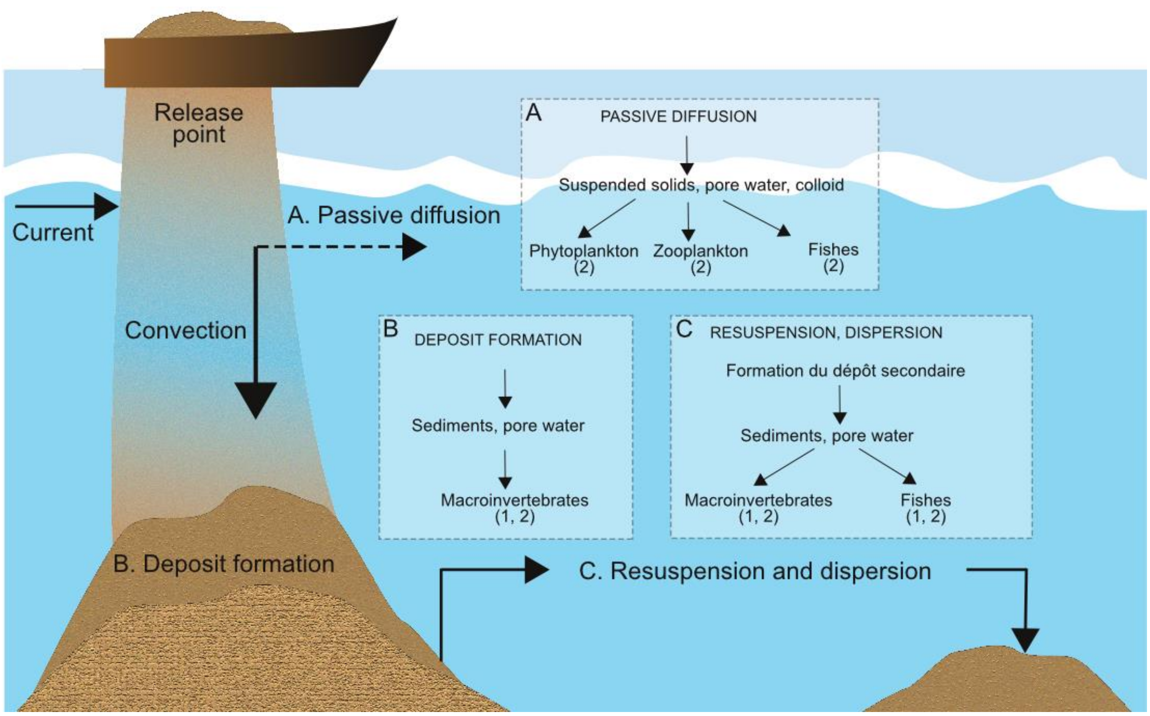

Figure 2: Generic conceptual model for the ecological risk assessment of open-water disposal of dredged sediments. The anticipated ecotoxicological effects on organisms for each compartment are the (1) growth, (2) survival, and/or (3) reproduction of aquatic organisms.

2.2.3 Ecotoxicological impact hypotheses for open-water disposal of dredged sediments

The target disposal site for dredged material may host a benthic community and thus serve as a feeding ground for fish and birds in the area. Many St. Lawrence fish species, especially juveniles, feed mainly on benthic organisms. Dredging operations can therefore indirectly affect fish by causing a relative loss of their feeding grounds. In addition, if a high percentage of fine particles are carried downstream, silting effects can also occur, particularly in spawning grounds. Fish in areas sensitive to silting can thus be affected by erosion of the deposit, especially during flood periods. Hence, even non-contaminated sediments could have a significant physical effect on the integrity of fish spawning and feeding grounds. In addition, if the eroded sediments are contaminated, it is necessary to ensure that this contamination will not have significant effects on the survival, growth and reproduction of fish (at these sites) when the eroded sediments are redeposited downstream.

Although they are an inextricable part of the process of assessing the impacts of a dredging and open-water disposal project, several biological targets or anticipated effects are not included in this ERA approach since they are already taken into account in other stages of the environmental assessment process:

- Assessment of the physical impact of dredged sediments on sensitive wildlife habitats (e.g. spawning grounds) is not included in the ERA approach, but must be considered when selecting the disposal site. To this end, it is suggested that reference be made to the *Atlas des habitats critiques connus ou d'intérêt particulier pour les poissons du fleuve Saint-Laurent entre le port de Montréal et l'Île aux Coudres*. This atlas was published in 2003 by the Société de la faune et des parcs du Québec with the specific goal of applying wildlife criteria when selecting sites for open-water disposal of dredged sediments. However, the fact that these habitats may not have been surveyed to date in a particular area should not be interpreted to mean that there are no impacts on species in that area.
- The physical impacts of the increase in SS concentrations during dredging work are not considered in the ERA approach. The increase in SS concentrations associated with dredging work and open-water disposal must be considered during the environmental impact study, and mitigation measures must be proposed in order to minimize the physical impacts on aquatic life. Monitoring programs implemented during the work must include SS. The criteria (currently under development) for managing SS associated with dredging and open-water disposal activities will make it possible to set objectives based on the dredging area under study.
- During the mass descent of sediments, the risks to fish, zooplankton and algae communities are considered minor compared to the risks to benthic communities, given the short exposure time, the small proportion of sediments that generally detach from the mass during disposal, and the high mobility and recolonization potential of pelagic species. In addition, the relatively high flow rates in the St. Lawrence River generally permit rapid dispersion of SS in the water column.

Finally, the acceptability of sediment disposal will depend on whether a new benthic community, similar to the original community, can be established at the site. This recolonization will be possible if the sediment disposal does not threaten the survival, growth and reproduction of benthic species.

The ERA approach presented in this document focuses on assessing the risks to benthic communities exposed over the short, medium and long term to contaminants in dredged sediments. This contamination could adversely affect the recolonization of a disposal site and the preservation of feeding grounds temporarily degraded by open-water disposal activities.

2.3 ERA Tier 1 Components: Detecting Sediment Contamination Levels

The objective of Tier 1 of the ERA approach for the management of dredged sediments is to qualify the level of sediment contamination. This step is based on comparing sediment contaminant concentrations with the *Criteria for the Assessment of Sediment Quality in Quebec* (EC and MDDEP 2007).

The database employed in developing these quality criteria contains various types of data that can be used to establish links between the concentration of a given chemical and the presence or absence of a biological effect. These data were derived from field and laboratory studies (species abundance and richness of benthic communities, and toxic effects on living organisms, especially on growth, reproduction and survival), spiked-sediment toxicity tests and equilibrium partitioning models of contaminant in water and sediment phases. The database also includes sediment quality assessment criteria adopted by other competent authorities.

2.3.1 Determining the chemical quality of sediments

First, using the characterization data obtained from the ecotoxicological study of St. Lawrence sediments, the sediments were classified based on the quality criteria and according to the most critical factor. This meant that the sediment under study was assigned the quality rating associated with the chemical with the worst ranking. Class 1 is assigned when the concentration of all the toxic substances measured is lower than or equal to the OEL, class 2 is assigned when the concentration of one or more substances is higher than the OEL but lower than or equal to the FEL, and class 3 is assigned when the concentration of one or more substances is higher than the FEL.

Other methods were then used to assess overall sediment quality and the risk to the aquatic environment. For example, the risk associated with a given contaminant can be represented by a quotient obtained by calculating the ratio between the concentration of the contaminant measured in the environment under study and a threshold effect. To consider the risk associated with the various sediment contaminants, aggregated quotients are proposed in various equations that can be used to calculate means or sums (Appendix B). Most clustering methods assume that risk is additive, which is generally considered acceptable in the literature (Long et al. 1998; Ingersoll et al. 2000; MacDonald et al. 2000; Fairey et al. 2001). Contaminants can also be grouped by chemical family (e.g. metals, pesticides) (Ingersoll et al. 2001). With data from this study, various methods for assigning quality classes were evaluated (Desrosiers et al. 2010). A few examples are presented in Appendix B.

Following the assessment of different methods, the approach whereby the sediment is assigned the quality class of the chemical with the worst ranking was maintained. In addition to protecting the environment, this approach is the simplest of the methods that we tested, and its results are comparable to the quotient-based approaches.

2.3.2 Cross-validating chemical screening and toxicity test results

The ecotoxicological study showed that 8 of the 10 samples (80%) with contamination levels below the OEL (class 1) had significant toxicity results on at least one toxicity test. In class 2, the probability of detecting adverse effects on benthic organisms was relatively high, with 20 of 29 samples (69%) presenting significant toxicity results. This study showed that the majority of the sediments in which at least one contaminant exceeded the FEL (class 3) were toxic, with 15 of 20 samples (75%) presenting significant toxicity results. Although the other 5 class 3 sediments were not toxic to the organisms tested, they posed a high risk of bioaccumulation and trophic transfer, since they contained high concentrations of mercury (Desrosiers et al. 2010).

From the perspective of sustainable management of dredged sediments in the St. Lawrence, the presence of class 1 sediments that were significantly toxic to the tested species or potentially toxic to benthic communities posed a problem that the study sought to understand. The open-water disposal of class 2 sediments can only be considered a valid option if toxicity tests have demonstrated that the sediments will not have adverse effects on the receiving environment and if the disposal does not contribute to degradation of the receiving environment. For class 3 sediments, these observations were consistent with a high probability that adverse biological effects will be observed.

2.3.3 Looking for explanations for the unexpected toxicity (false negatives)

In an effort to explain the unexpected toxicity in the class 1 sediments, we first looked for potential sources of toxic substances not measured during sediment characterization (polybrominated diphenyl ethers [PBDEs], dioxins/furans, etc.), but that might be present at the class 1 stations where the toxicity was detected.

Agricultural activities and pesticides can exert significant pressure on ecosystems (Hela et al. 2005). Pesticides can be toxic to invertebrates (Anderson et al. 2005) and can modify the structure of benthic communities or the functional traits of species (Anderson et al. 2005; Liess and Von Der Ohe 2005). Some of our class 1 stations may be affected by agricultural activities. For example, station 51 is located near the Rivière au Saumon and the Saint-Régis River, where a few pesticides were measured in 1989 (Fortin et al. 1994) and in 1992–1993 (Rondeau 1996). Three other class 1 stations that showed significant toxicity are located in Lake Saint-Pierre (stations 13, 15 and 34). These stations were affected by the Maskinongé and Bayonne rivers; there is extensive or intensive agriculture in the watersheds of these two rivers (Robitaille 1997, 2005; Giroux 2007; Pelletier et al. 2008).

Municipal wastewater represents another source of contaminants, including pharmaceuticals, that were not analyzed in this study. Currently, little is known about the toxicity of pharmaceuticals (Fent et al. 2006; Gros et al. 2007), although some of them are known to be fairly persistent in the environment (Bendz et al. 2005). One class 1 station, located in the Montréal island area (station 9), presented toxicity as measured by the significant mortality of *Chironomus riparius*. This station is influenced by highly turbid water coming from the Rivière des Mille Îles, the Rivière des Prairies and the Rivière L'Assomption. The first two rivers collect municipal wastewater, while the watershed of the Rivière L'Assomption is subject to heavy agricultural pressure (Rondeau 1996).

In Lake Saint-Louis, we measured significant toxicity in four class 1 stations. Station 22 may be influenced by groundwater coming from the city of Montréal and may potentially be contaminated by industrial activities or by surface water coming from small contaminated streams (Deschamps et al. 2005). On the south shore, station 23 could be influenced by the Saint-Louis River, even though the contaminant concentrations are clearly lower than those measured in the stations located directly in the river's plume. Since stations 16 and 17 are located in a zone where there is mixing of water masses that are generally considered to be only weakly contaminated, the toxicity observed for these two stations remains unexplained. (Desrosiers et al. 2010)

Second, the study examined the effect of confounding factors. The bioavailability and toxicity of contaminants can be influenced by sediment particle size (Watzin et al. 1997) and by the presence of inorganic substances that form complexes with metals (Ankley 1996; Bervoets et al. 1997; Huerta-Diaz et al. 1998; Fan and Wang 2001), or organic matter (Bervoets et al. 1997; Di Toro et al. 2005). In this context, the use of certain statistical tools, specifically regression trees, is helpful in determining the chemical or environmental variables that can influence or at least be correlated with the toxic response. Statistical tools are also helpful in determining the relative contribution and probable threshold effect level of these variables. The main conclusion of these statistical tests was that the toxic response observed for the 10 stations with class 1 sediments was explained mainly by total sulphur, a substance not targeted by the quality criteria but which might explain up to 80% of the toxicity observed in the class 1 sediments (Desrosiers et al. 2010).

It is generally recognized that sulphates and sulphites can bind strongly with metals and, consequently, reduce the toxicity of the metals (Word et al. 2005). However, sulphites are also recognized as being potentially toxic to invertebrates (Knezovich et al. 1996; Wang and Chapman 1999), while elemental sulphur has been shown to be potentially toxic to bacteria and fish (Svenson et al. 1998). In our study, based on the results obtained at the 59 stations, we observed significant positive relationships between the concentrations of total sulphur and certain inorganic (arsenic, cadmium, copper, zinc) and organic (PCBs, PAHs, petroleum hydrocarbons) contaminants. In addition, a significant relationship between total sulphur and organic carbon was observed, thus supporting the hypothesis of a relationship between sulphur and the presence of organic contaminants included in the measurement of total organic carbon (TOC) (Desrosiers et al. in preparation).

Sulphur pollution can originate from many sources, the main ones being agriculture (fertilizers, pesticides), atmospheric depositions as a result of combustion of fossil fuels, acid mine drainage and industry (Schlesinger 1997). In industry, sulphur and especially sulphides are known to be associated with many chemicals (lubricants, varnishes) and various types of industrial processes (tanning, paper making). Sulphur can also be found associated with construction materials such as plaster (Kloppmann et al. 2011) and in refinery wastewater (van Leerdam et al. 2006).

Third, our research also assessed the structure of benthic macroinvertebrate communities; total sulphur was found to be a factor that influences the structure of benthic communities in class 1 sediment as in others (Masson et al. 2010). Generally, high concentrations of sulphur are associated with low levels of oxygen in sediments. The taxa that are generally hypoxia-tolerant are also sulphur-tolerant, such as diptera (Wiederholm 1976, 1984; Pinder 1986) and gastropods (Goodnight 1973). We observed that certain macroinvertebrates can colonize sulphur-rich areas, probably because of their ability to use the available oxygen and to resist the toxic effects of sulphur.

In addition, crustaceans and gastropods, which, unlike diptera, are relatively mobile, can avoid severe conditions of anoxia and high sulphur concentrations. Macroinvertebrates can develop physiological adaptations that enable them to survive in anoxic areas, either through hemoglobin production or ventilation (Stief et al. 2005), or by changing their mobility behaviour (Salánki et al. 2003), thus creating selection pressure in favour of the more tolerant taxons and to the detriment of more sensitive species.

2.3.4 Discussion on the need to review the methodology used in Tier 1

The toxicity observed in the class 1 sediments could have led to changes in the original Tier 1 risk assessment model, by suggesting for example that toxicity tests be incorporated from the outset. However, as mentioned in section 2.3.2, this first screening step proved effective in distinguishing class 2 and class 3 sediments. In addition, the statistical analysis results presented in the ecotoxicological study demonstrated that for the subset of the 10 class 1 stations, this unexpected toxicity was in most cases associated with total sulphur concentration. Hence, exceeding the 1400 mg/kg total sulphur concentration in class 1 sediments appears to be indicative of the presence of sulphur of anthropogenic origin potentially associated with other substances that could pose a risk to benthic organisms. It is therefore proposed that the total sulphur concentration be considered during the ecological risk assessment and analyzed at the same time as the contaminants for which quality criteria have been established. Although this is only exploratory, the use of sulphur as a potential indicator to assess the presence of a toxic substance other than those included in the sediment quality criteria must be evaluated before authorizing open-water disposal. The data collected during monitoring will make it possible to confirm this hypothesis. According to the ecotoxicological study, adding sulphur as an analytical parameter significantly reduces the percentage of samples determined to be toxic in class 1 sediments—from a figure of nearly 80% based on *Hyalella azteca* and/or *Chironomus riparius* mortality to 30% of false negative when the sulphur concentration is taken into consideration (Desrosiers et al. 2012).

In conclusion, the toxicity observed in class 1 sediments could have led to changes in ERA Tier 1, by suggesting for example that toxicity tests be incorporated from the outset. However, this screening step proved to be effective for distinguishing class 2 and class 3 sediments. In addition, the statistical analysis results demonstrated that for the subset of class 1 stations, total sulphur concentrations were significantly associated with this unexpected toxicity. Although this is only exploratory, it is proposed that the total sulphur concentration be considered during the ecological risk assessment and analyzed at the same time as the contaminants. In fact, the use of sulphur as a potential indicator of the presence of a toxic substance other than those considered in the sediment quality assessment criteria, as well as its effect on the structure of benthic communities, must be evaluated before authorizing open-water disposal by measuring the concentration of sulphur in the sediments. The 1400 mg/kg threshold proposed in this document is based on the results of the ecotoxicological study as well as on the ambient levels observed in Lake Saint-François and Lake Saint-Pierre. This threshold may need to be changed depending on site-specific ambient levels. The data generated by project monitoring will make it possible to validate the approach adopted for ERA Tier 1.

Quality criteria therefore remain the foundation of this first screening tier of the ecological risk assessment process. The list of recommended chemical analyses must include analyses of total sulphur, and the results must be compared to the concentration of 1400 mg/kg, which is generally higher than ambient levels (see Text Box B) and likely to indicate the presence of a potential risk for benthic organisms.

Text Box 2: Ambient total sulphur levels in Lake Saint-François and Lake Saint-Pierre

In St. Lawrence River sediments, sulphur can be of natural or anthropogenic origin. Sulphur is part of the mineral structure of sulphides and sulphates. Sulphides form a group of approximately 350 minerals, the most common of which is iron sulphide or pyrite (FeS_2), whose primary characteristic is that it oxidizes easily, thus releasing the sulphur (Aubert et al. 1978). Sulphides are present in the basic materials used in the majority of Cu, Zn and Pb mining operations in Quebec. They are found in abundance in many sedimentary and volcanic rocks such as those of the Appalachians and the Canadian Shield (Hocq et al. 1994; AFNOR 2003; La Violette 2004; MDDEP and EC in preparation). The mineral group of sulphates (SO_4) includes approximately 220 minerals, the most well-known of which is gypsum (calcium sulphate) (Aubert et al. 1978). Sulphur can also come from the reduction of sulphate by sulphate-reducing bacteria in environments rich in organic matter and deficient in oxygen. Because of their more soluble and more friable characteristics, sulphates are used extensively in chemical fertilizers, in the manufacture of gypsum board and also as pesticides and in paints. Finally, sulphur is also found, again naturally, in various organic forms, primarily in oil, coal and natural gas. Anthropogenic input of sulphur into the environment is primarily through mineral and petroleum products treatment and processing processes.

Data collected during water quality monitoring in Quebec have made it possible to determine the ambient levels of total sulphur for sediments in Lake Saint-François and Lake Saint-Pierre (Magella Pelletier, Environment Canada, personal communication). In view of the fact that sulphur is present in potentially high concentrations in nature, the ambient levels were determined using the 75th percentile of distribution. The average ambient value for the data taken as a whole is 752 mg/kg, with 1730 mg/kg in Lake Saint-François (LSF) and 538 mg/kg in Lake Saint-Pierre (LSP), respectively (Figure 2-1). Certain areas of Lake Saint-François have ambient levels higher than 1400 mg/kg, with an observed median of 872 mg/kg (Figure 2-2). Ambient sulphur levels are lower in Lake Saint-Pierre, with a maximum concentration of 1202 mg/kg and a median value of 370 mg/kg.

The total sulphur concentration of 1400 mg/kg proposed in the ERA approach for dredged sediments corresponds to the 90th percentile of the overall distribution of the ambient levels. Consequently, the proposed concentration of 1400 mg/kg is generally higher than the ambient levels in Lake Saint-François and Lake Saint-Pierre. However, it is important to bear in mind that the total sulphur concentration of 1400 mg/kg as an indicator of a potential risk was determined based on data from a relatively small number of stations (10). This parameter will therefore have to be monitored based on studies of dredging carried out in the coming years.

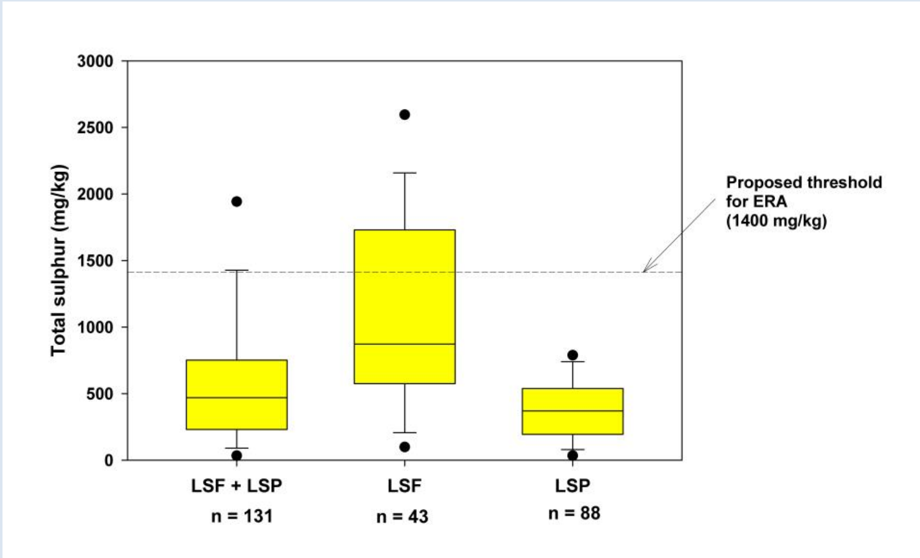

Figure 2-1: Total sulphur concentrations in the sediments of Lake Saint-François (LSF) and Lake Saint-Pierre (LSP) expressed in mg/kg (minimum, 10th, 25th, 50th, 75th and 90th quantiles and maximum) (Magella Pelletier, Environment Canada, personal communication).

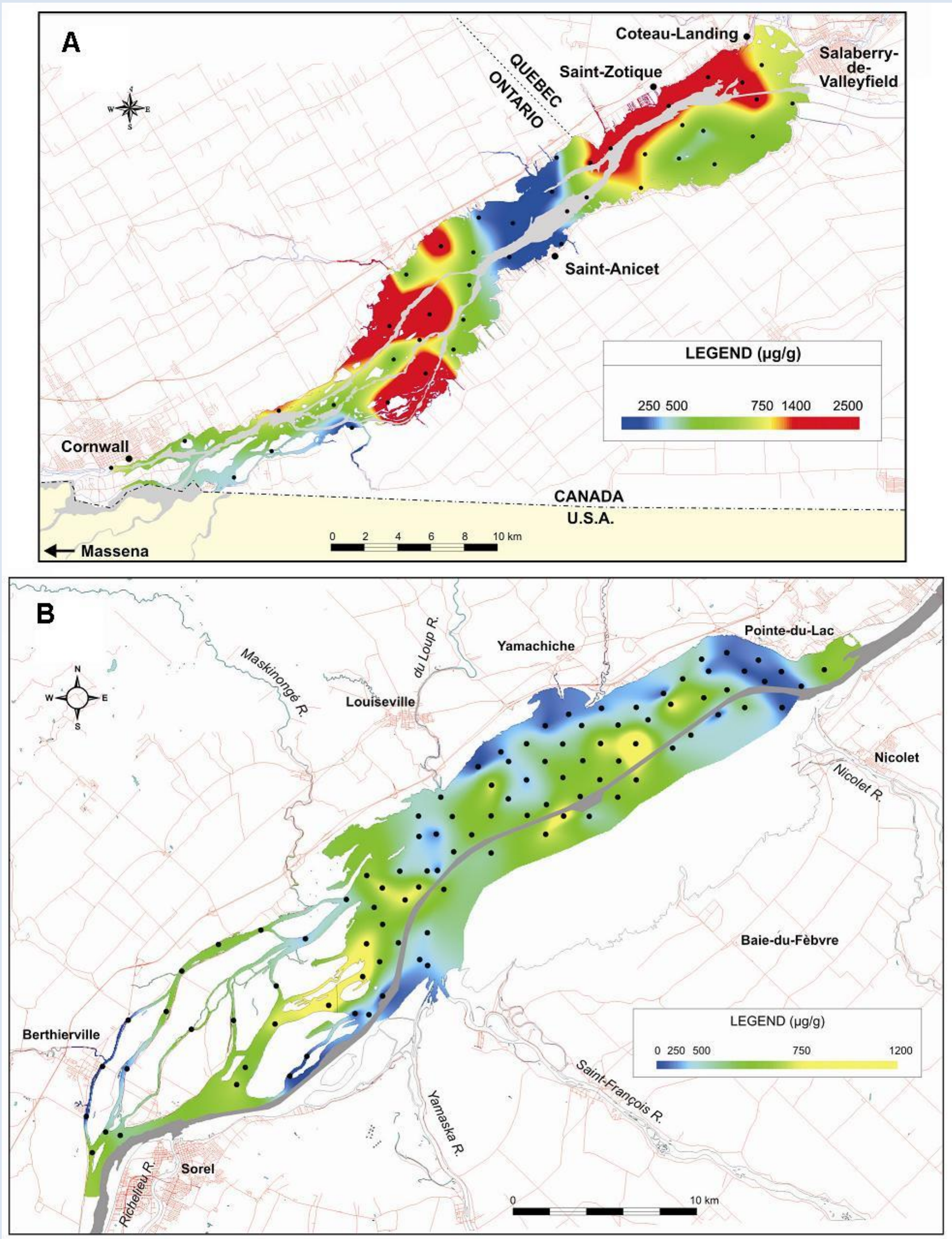

Figure 2-2: Total sulphur concentrations in the sediments of Lake Saint-François (A) and Lake Saint-Pierre (B) (Magella Pelletier, Environent Canada, personal communication)

2.4 ERA Tier 2 Components: Laboratory Toxicity Tests

Introducing toxicity tests in Tier 2 enabled us to examine, alongside the quality criteria, the exposure routes and the impact on aquatic organisms of all the toxic substances in dredged sediments. This summary report presents only the main findings that served as the basis for deciding which tests would be included in the assessment approach. The detailed results are presented in a scientific article (Desrosiers et al. in preparation).

2.4.1 Toxicity tests

Four standardized toxicity tests were evaluated during this study: two whole-sediment tests and two pore water tests. The whole-sediment toxicity tests were conducted with juvenile amphipods (*Hyalella azteca*), epibenthic organisms living at the water-sediment interface, and with chironomid larvae (*Chironomus riparius)*, benthic organisms. The whole-sediment toxicity tests were conducted with methods adapted from standardized protocols:

- 7-day survival and growth of *Chironomus riparius* (AFNOR Experimental Standard T 90 339-1 [EC 1997b; AFNOR 2004])
- 14-day survival and growth of *Hyalella azteca* (modified protocol of Method EPS1/RM/33 [EC 1997a; AFNOR 2003])

The pore water toxicity tests were carried out on organisms representing two trophic levels: the alga *Pseudokirchneriella subcapitata*, representing primary producers, and the rotifer *Brachionus calyciflorus*, representing primary consumers. The rotifer *Brachionus* was preferred over the cladocean *Ceriodaphnia* because a toxicity test on reproduction can be performed faster (48 hours versus 7 days) and a much smaller volume of pore water is required (20 to 30 ml versus a few litres). The pore water toxicity tests were carried out with methods adapted from standardized protocols:

- 48-hour reproduction of *Brachionus calyciflorus* (AFNOR Standard T-90-377 [AFNOR 2000])
- growth of the unicellular algae *Pseudokirchneriella subcapitata* (formally *Selenastrum capricornutum*) using two protocols: one with the flask technique (96 hours without EDTA [CEAEQ 2005]) and one with the microplate technique (72 hours with EDTA [EC 1992])

For the four toxicity tests, the effect thresholds were validated by considering whether or not there was a significant difference (t-test; $p < 0.05$) between the test sample and the control sample, as well as considering the variability observed in the controls. It was thus determined that the sediment was detrimental to the survival or growth of *Chironomus riparius* and *Hyalella azteca* (mortality and growth) when the response (observed effect) was greater than 20% and significantly different from the control results. For the pore water tests, the threshold was 15% for the two tests with *Pseudokirchneriella subcapitata* and 40% for the reproduction test with *Brachionus calyciflorus*.

2.4.2 Particle size distribution

Apart from selecting which tests to use, one of the main points studied concerning whole-sediment toxicity tests was whether these tests should be carried out on the coarse or fine fractions of the sediments. In fact, as was mentioned in the presentation of the conceptual model (Section 2.2), during open-water dumping of sediments, current velocity may result in differentiation by particle size. Since information on the particle size distribution of the sediments to be dumped can influence the risk assessment, several questions were studied in detail based on the data collected during the 2004 and 2005 sampling campaigns, as well as on historical data.

The general picture of the relationships between chemical contamination, sediment characteristics (particle size distribution, organic matter) and toxicity is complex. In fact, generally, the relationships observed were highly variable depending on the database and the toxicity tests used, and no general trend could be identified during our study. For example, in the historical database, the effects of toxicity in *Hyalella* and algae declines as the proportion of clay increases, although this is not the case with silt. However, in our tests, the toxicity observed increased as the proportion of sand increased. This influence of particle size distribution is only partially observed in the 2004–2005 data: *Hyalella azteca* mortality declined in the presence of clay, but sand had no apparent effect. Hence, at this stage, there is no evidence to support a requirement to systematically conduct toxicity tests on different sediment fractions in Tier 2.

2.4.3 Results obtained and selection of toxicity tests

The second point studied concerns the choice of the battery of tests to be performed and the complementarity of the tests. The relationships observed between sediment contamination and the toxicity test response varied with the species. Indeed, *Hyalella azteca* mortality increased significantly in the presence of arsenic, cadmium, copper, lead and zinc, whereas *Chironomus riparius* mortality increased in the presence of cadmium, copper, zinc and butyltin. *Hyalella azteca* and *Chironomus riparius* growth was not very sensitive to the presence of contaminants. This parameter was more sensitive to the presence of nutrients in the sediment.

The *Brachionus calyciflorus* test is a reproduction test. Reproduction was a more sensitive parameter than mortality, and the test appeared to demonstrate the influence of certain metals (cadmium, lead and zinc) as well as organic contaminants (PCBs, PAHs and butyltin) on sediment toxicity. This was a real advantage. However, the results of this test can be difficult to interpret. For example, it is possible that stimulation of reproduction may be linked to the presence of endocrine disruptors, as was observed during the study of a site exposed to significant pollutant loads from agriculture. In addition, the variability of the controls seemed high, which resulted in an increase in the toxicity threshold (≈40% effect for the “reproduction” parameter) compared to the other tests. In view of these problems, it is recommended that this test not be used alone (Desrosiers et al. in preparation).

In the present study, the tests on algae, including *Pseudokirchneriella subcapitata*, frequently showed growth stimulation and did not seem appropriate for detecting toxicity in sediments. However, these tests could be useful if the assessment parameter in a particular case study included a risk of eutrophication (Desrosiers et al. in preparation).

Because of the similarities observed between the results of the pore water toxicity tests (*Pseudokirchneriella subcapitata* and *Brachionus calyciflorus*) and the whole-sediment toxicity tests (*Chironomus riparius* and *Hyalella azteca*) as well as the difficulty of interpreting the results when these tests were performed on pore water, not to mention the need to extract this water, there did not appear to be any real benefit in including these types of pore water tests in the ERA approach. In addition, as mentioned in section 2.2, during open-water disposal, significant dispersion of SS occurs in the St. Lawrence River, which tends to dilute the contaminants.

In conclusion, the whole-sediment tests conducted with *Hyalella azteca* and *Chironomus riparius* were retained in the ERA approach to assess the impact of dredged sediments on benthic communities.

3.0 Application of the ERA Approach

The ERA approach presented in this document incorporates the quality criteria at the outset of the assessment process (Tier 1) and outlines a multi-tier process for assessing the ecological risk associated with open-water disposal of dredged sediments that incorporates toxicity tests in the second tier. The objective of this ERA approach is to determine whether open-water disposal of sediments is acceptable from an ecotoxicological perspective for benthic communities.

3.1 Tier 1: Detecting Sediment Contamination Level

In Tier 1 of the approach, the chemical analysis results are compared with the sediment quality assessment criteria adopted in Quebec (see Figure 3). In managing dredged sediments, sediment quality classes are determined based on two contamination levels: the OEL and the FEL (EC and MDDEP 2007).

The basic parameters that must generally be analyzed in order to assess sediment quality during a dredging project are particle size distribution, metals (arsenic, cadmium, chromium, copper, mercury, nickel, lead and zinc), PCBs, PAHs, TOC and petroleum hydrocarbons (C_{10}–C_{50}). Total sulphur must also be added to the list in view of the observations concerning its role as a potential indicator of toxicity. Sediment quality criteria are available for metals, PCBs and PAHs (EC and MDDEP 2007). For total sulphur, the threshold chosen is 1400 mg/kg[5] (see Text Box 4). Other parameters may be added to this list depending on the specific context of each dredging operation.

[5] According to the results of research carried out to explain the toxic response of class 1 sediments (Desrosiers et al. 2010).

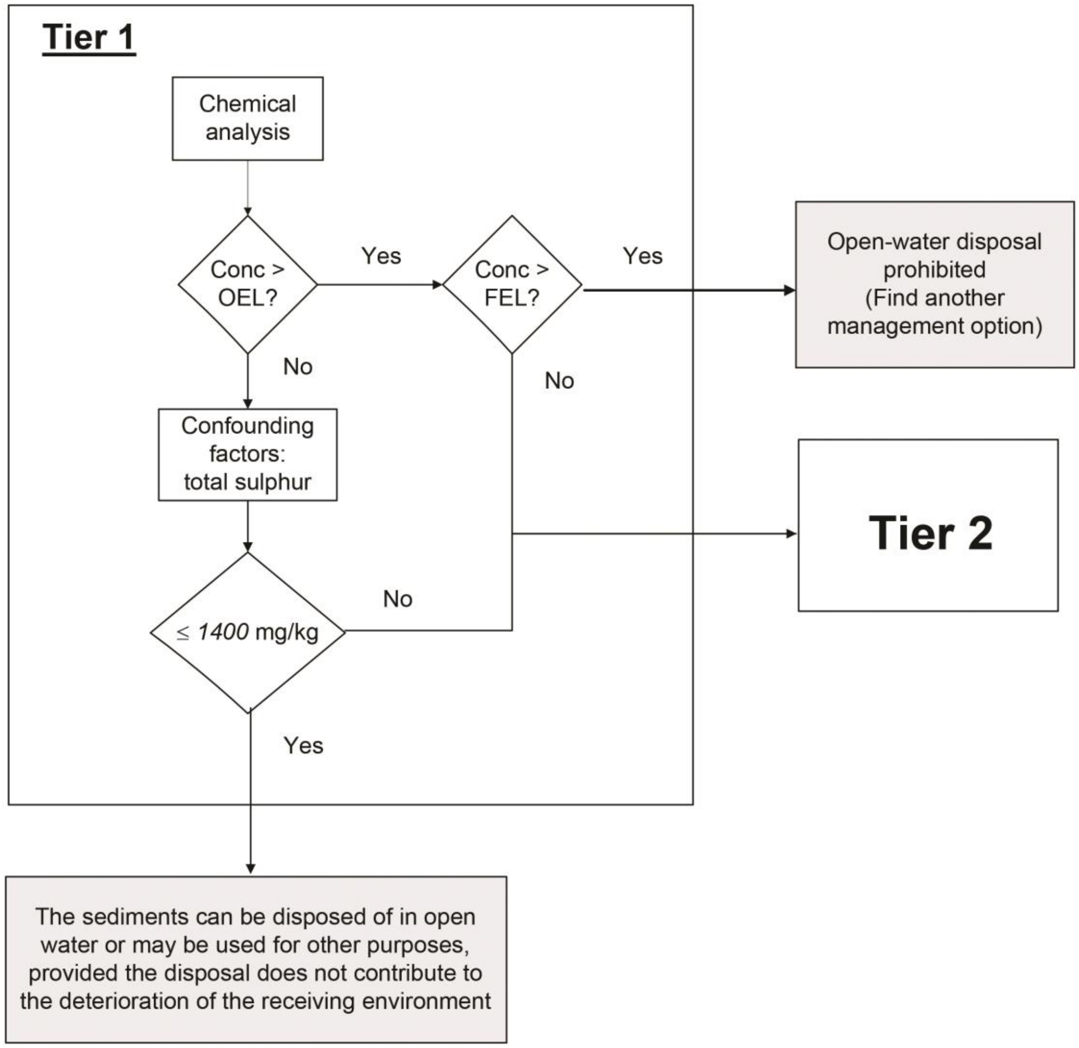

Figure 3: Detailed logic diagram of Tier 1 of the ecological risk assessment approach

Situation 1:

A sediment sample is defined as class 1 when the concentration of all the contaminants is less than or equal to the OEL,[6] the concentration at which the probability of observing adverse biological effects is considered relatively low. However, if a sediment is classified as class 1, but the total sulphur concentration is higher than 1400 mg/kg (or higher than the ambient level specific to the study area if that level is higher than 1400 mg/kg), it may indicate the presence of toxic substances not taken into account by the sediment quality assessment criteria. The sample could therefore be toxic and pose a risk to benthic organisms. For that reason it is necessary to proceed to Tier 2 of the assessment (see Figure 3).

[6] The *Criteria for the Assessment of Sediment Quality in Quebec* document includes the possibility of comparing the concentrations to natural background levels if they are higher than the standard criteria (EC and MDDEP 2007).

If all the contaminants are below the OEL and total sulphur is less than 1400 mg/kg, the ERA can be considered complete (see Figure 3). The sediments can be disposed of in open water or be used for other purposes, for example for beneficial uses such as creating wildlife habitats or for beach replenishment, provided that the receiving environment is not thereby adversely affected (see Text Box 3).

Text Box 3: The principle of non-degradation of the receiving environment

Although meeting chemical quality criteria is generally indicative of good sediment quality, ecosystems may still be degraded or disturbed by open-water disposal of sediments. First of all, the chemical quality of the sediments to be disposed of must be equivalent to or higher than the sediment quality present in the disposal area, based on the quality classes assigned. In addition, as mentioned in the conceptual model, even if the sediments are not toxic, the process of dumping dredged sediments and the accompanying significant increases in the concentration of suspended solids can physically alter aquatic ecosystems or cause habitat loss. Considerations relating to the health of the receiving ecosystem, in terms of protecting both aquatic life and human health or in terms of a specific use of the site, the need to protect a threatened or vulnerable species, or the presence of spawning grounds, may require specific mitigation measures or additional actions. At no time should the sediment quality criteria be considered as implicit approval to allow a site to degrade until it reaches the adopted threshold values (EC and MDDEP 2007).

Situation 2:

A sediment sample is defined as class 2 when the concentration of at least one contaminant is between the OEL and the FEL. Between these two thresholds, the probability of observing adverse biological effects is relatively high and increases with the concentration measured. As a result, and in accordance with what is suggested in the sediment quality assessment document (EC and MDDEP 2007), toxicity tests will be performed on these sediments, and these Tier 2 results will have to demonstrate that the sediments are safe in order to be considered suitable for disposal in an aquatic environment (see Figure 3).

Situation 3:

A sediment sample is defined as class 3 when the concentration of at least one contaminant is higher than the FEL, the concentration above which the probability of observing adverse biological effects is very high. As a result, in the case of class 3 sediments, open-water disposal is prohibited, as stipulated in the document describing the sediment quality criteria (EC and MDDEP 2007) (see Figure 3). In this case, the sediments must be treated or properly contained.

Text Box 4: Tier 1 – Detecting sediment contamination levels

Measurement Parameters

- Metals (arsenic, cadmium, chromium, copper, mercury, nickel, lead and zinc), PCBs, PAHs and petroleum hydrocarbons (C_{10}–C_{50}).
- Total sulphur.
- Particle size distribution, TOC.
- Other parameters may be added depending on the specific context of each dredging operation (e.g. organotins, flame retardants)
- Sediment sampling must be carried out in accordance with the *Sediment Sampling Guide for Dredging and Marine Engineering Projects on the St. Lawrence River* (EC 2002a, 2002b; MDDEP and EC in preparation).
- Physicochemical analyses must be carried out in accordance with the *Guide de charactérisation physico-chimique et toxicologique des sédiments* (MDDEP and EC in preparation).

Interpreting the Results (Figure 3)

- In this approach to assigning quality classes, a sediment is assigned the quality class of the chemical with the worst ranking.

Situation 1:

- **The concentration of all the contaminants is less than or equal to the OEL and the total sulphur concentration is less than or equal to 1400 mg/kg.**
  - If the concentrations of all the contaminants are less than or equal to the OEL (or the natural background level if that level is higher than the OEL) and if the total sulphur is less than 1400 mg/kg, the ERA can be considered complete. The sediments can be disposed of in open water or can be used for other purposes, provided that the receiving environment is not thereby adversely affected.

Situation 2:

- **The concentration of at least one contaminant is between the OEL and the FEL or the concentration of all the contaminants is less than or equal to the OEL, but total sulphur is higher than 1400 mg/kg.**
  - The probability of detecting adverse effects on benthic organisms is relatively high in this class.
  - Open-water disposal can be considered a valid option only if toxicity tests demonstrate that the sediments will not have adverse effects on the receiving environment and that, as for class 1 sediments, disposal does not contribute to the degradation of the receiving environment.
  - In the case of class 2 sediments, it will be necessary to proceed to assessment Tier 2, where toxicity tests will be performed.

> **Situation 3:**
> - **The concentration of at least one contaminant exceeds the FEL.**
>   - The probability of observing adverse biological effects is very high.
>   - Open-water disposal is prohibited.
>   - The sediments must be treated or properly contained.

3.2 Tier 2: Assessing Sediment Toxicity

If the dredged sediments were rated as class 2 during Tier 1, or if rated as class 1 but total sulphur concentration is higher than 1400 mg/kg, open-water disposal can only be considered a valid option if toxicity tests clearly demonstrate that the sediments are safe for the receiving environment (see Figure 4).

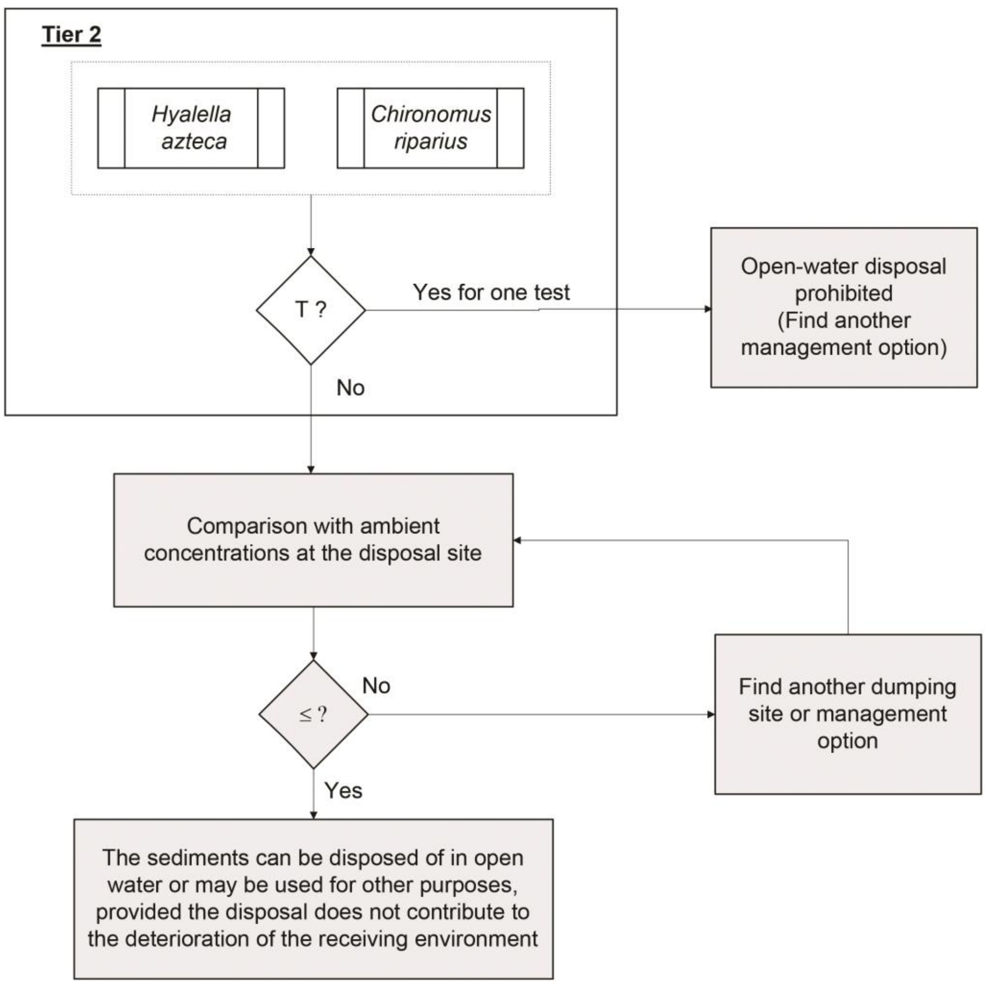

Figure 4: Detailed logic diagram of Tier 2 of the ecological risk assessment approach for dredged sediments

The two toxicity tests adopted for this ERA approach have been the subject of numerous standardized test protocols. For amphipods (*Hyalella azteca*), the toxicity test protocol of Environment Canada (1997a) was chosen and modified for the water/sediment ratio (4/1; MDDEFP and EC in preparation). For chironomids, since the standardized test protocol of Environment Canada (1997b) is more suited to the life cycle of *Chironomus tentans*, which is longer than that of *Chironomus riparius*, the test protocol of AFNOR (2004), developed for *Chironomus riparius*, was therefore chosen and modified for temperature (23°C; MDDEFP and EC in preparation). This makes it possible to complete the test in 7 days instead of 10. The whole-sediment toxicity tests must be performed and interpreted in accordance with the methodologies outlined in the *Guide de caractérisation physico-chimique et toxicologique des sédiments* (MDDEFP and EC in preparation). The toxicity test results are considered significant when the difference in response (observed effect) between the contaminated sediments and the laboratory reference sediments is ≥ 20% in terms of mortality or growth inhibition.

Open-water disposal can be considered a valid option if none of the toxicity tests yields a significant toxic response (see Figure 4). If a significant toxic response is observed on one or more of the tests performed, the sediments are considered detrimental to the health of benthic organisms and open-water disposal is therefore prohibited.

In addition, even in the absence of toxicity, a proper characterization of the disposal site is required before authorizing open-water disposal. The concentrations measured in the dredged sediments must be less than or equal to the levels measured in the sediments at the disposal site. This comparison must be carried out based on the sediment quality class for each of the substances targeted by the quality criteria. According to this procedure, non-toxic class 2 sediments cannot be deposited on top of class 1 sediments. It is also advisable to ensure that the site selected for disposal of the dredged sediments minimizes adverse effects on the environment and related activities. The Tier 2 application procedure is summarized in Text Box 5.

Text Box 5: Tier 2 – Toxicity tests

Measurement parameters

The whole-sediment toxicity tests to be performed are as follows:

- 7-day survival and growth of *Chironomus riparius* (AFNOR Experimental Standard T 90 339-1 [AFNOR 2004]).
- 14-day survival and growth of *Hyalella azteca* (Method EPS1/RM/33 [EC 1997a]).
- Physicochemical characterization of the sediments used for the toxicity tests (see Tier 1 measurement parameters).

Interpreting the results (Figure 4)

- Sediments are considered detrimental to the health of benthic organisms and ***open-water disposal is prohibited*** if
  - One or more of the toxicity tests produces a significant toxic response:
    - The toxicity test results are considered significant when the response observed is ≥ 20% in terms of mortality or growth inhibition in comparison with controls.
  - The sediments must then be treated or properly contained.
- ***Open-water disposal can be considered a valid option*** if
  - None of the toxicity tests produces a significant toxic response;
  - The site selected for disposal of the dredged material minimizes the adverse physical effects on the environment and related activities; and
  - The concentrations in the dredged sediments are less than or equal to the levels measured in the sediments at the disposal site. This comparison must be carried out based on the quality class for each contaminant.

Conclusion and Recommendations

The complete ecological risk assessment approach to support the management of dredged sediments in the St. Lawrence River is presented in Figure 5.

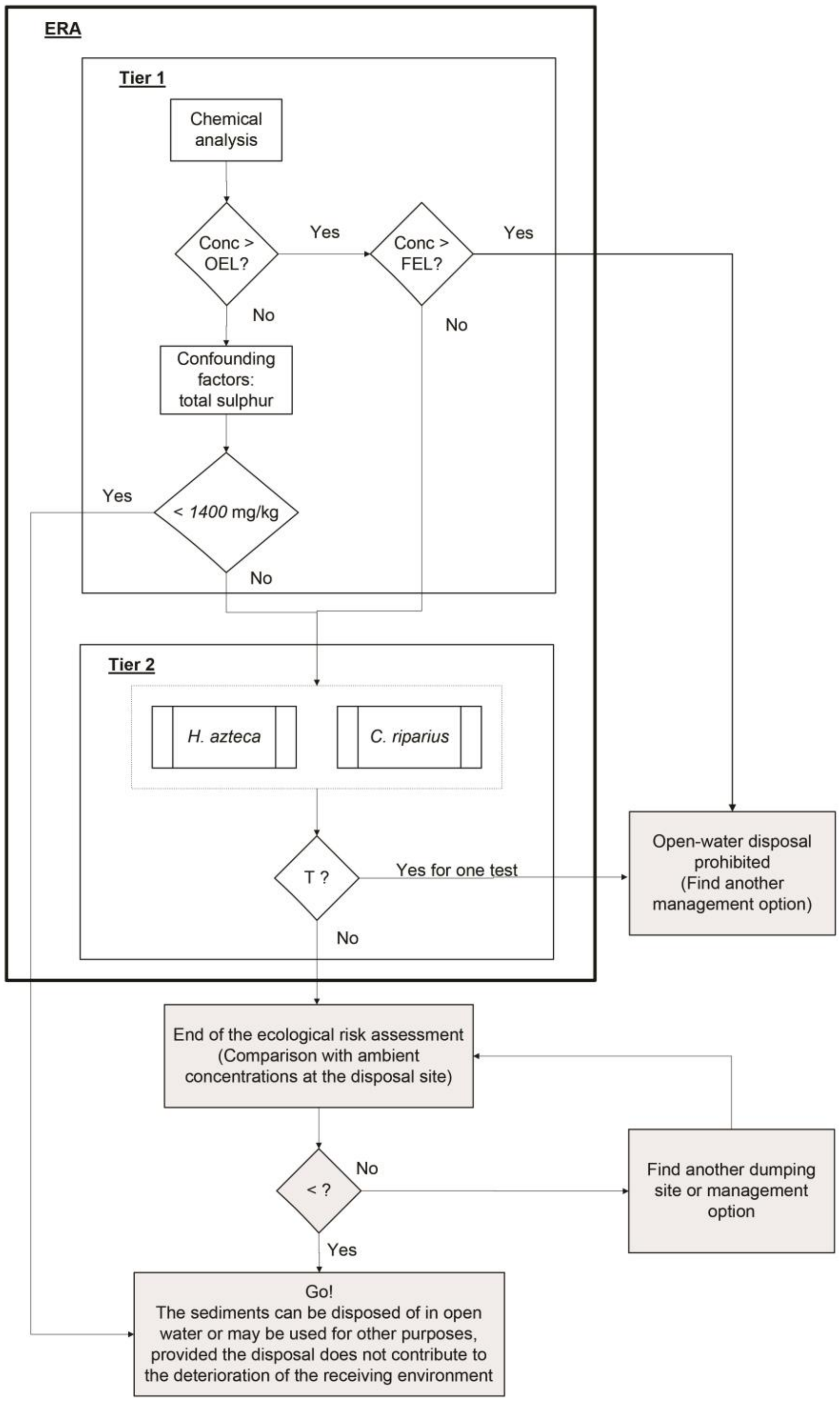

Figure 5: Ecological risk assessment approach to support the management of dredged sediments in the St. Lawrence River

This ecological risk assessment approach for open-water disposal of sediments to support the management of dredging projects was developed using data collected from the freshwater portion of the St. Lawrence River. Consequently, this ERA approach applies only to open-water disposal of dredged sediments located in the freshwater portion of the St. Lawrence, i.e., from Cornwall, Ontario, to the eastern tip of Île d’Orléans, including Lake Saint-François, Lake Saint-Louis and Lake Saint-Pierre. For dredging work planned in saltwater, it is recommended that reference be made to the procedures and toxicity tests outlined in Environment Canada’s Disposal at Sea Program (see Environment Canada website). For the area of brackish water, a project under development as part of the St. Lawrence Action Plan (SLAP) will determine the tolerance limits of toxicity tests for use in freshwater and saltwater and develop or adapt toxicity tests to the brackish water conditions present in certain parts of the St. Lawrence River. In addition, as part of another SLAP project, there are plans to develop an ecological risk assessment approach whose primary objective will be to determine, using available prediction and assessment tools, the criteria and analytical process for assessing the ecological risk applicable to contaminated sediment sites that may be candidates for remediation.

Finally, it is important to bear in mind that the total sulphur concentration of 1400 mg/kg as an indicator of potential toxic effects was established based on data from a relatively small number of stations (10). This parameter will therefore have to be monitored based on studies of dredging carried out in the coming years. We also recommend continued research in order to (1) obtain more accurate and complete data on background levels (ambient and natural), particularly for Lake Saint-Louis and the fluvial section; and (2) determine the nature of the contaminants associated with sulphur in the various sections of the St. Lawrence River.

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Appendix A: Summary of the Analytical Methods Used

Table A-1: Summary of the analytical methods used for metals and metalloids, nutrients, organic matter and particle size distribution of the sediments and in pore water

Variables measured	Matrix	Method	Equipment used	Detection limit	Reference
Al As Ca Cd Cr Cu Fe Mn Ni Pb Zn	Sediments	Argon plasma emission spectrometry with two mineralization methods: a) Total extractible fraction (HCL 2.4N/HNO_3 8N; 3/1) b) HCl-extractible fraction (HCl 1N)	Optima 3000DV; Perkin Elmer Elan DRCII; Perkin Elmer	12.0 mg/kg 0.27 mg/kg 17.0 mg/kg 0.22 mg/kg 3.0 mg/kg 2.1 mg/kg 18.0 mg/kg 1.1 mg/kg 0.6 mg/kg 1.2 mg/kg 2.5 mg/kg	MA. 200 – Mét. 1.1 (CEAEQ 2003e)
As Cd Cr Cu Ni Pb Zn	Chironomids	Argon plasma emission spectrometry using the following mineralization method: a) Total extractible fraction (HCL 2.4N/HNO_3 8N; 3/1)	Optima 3000DV; Perkin Elmer Elan DRCII; Perkin Elmer	0.01 mg/kg 0.01 mg/kg 0.01 mg/kg 0.01 mg/kg 0.01 mg/kg 0.01 mg/kg 0.1 mg/kg	MA. 200 – Mét. 1.1 (CEAEQ 2003e)
Hg	Sediments	Thermal decomposition and atomic absorption spectrometry	DMA-80; Milestone	0.035 mg/kg	MA. 207 – HG 2.0 (CEAEQ 2007)
Total sulphur	Sediments	Infrared spectrophotometry	LECO SC-444	50 mg/kg	(CEAEQ 2006a)
Total organic carbon (TOC)	Sediments	Titration		0.05%	(CEAEQ 2006g)
Total Kjeldahl nitrogen (TKN)	Sediments	Colorimetric method	Technicon Model II	100 mg/kg	(CEAEQ 2006f)
Total phosphorus (TP)	Sediments	Colorimetric method	Technicon Model II	200 mg/kg	(CEAEQ 2006f)
Particle size distribution	Sediments	Sedimentation analysis	Hydrometer 152H	0.1%	(Pelletier 2008)
pH	Sediments	Electrometric method	Accumet AP72 portable pH meter	Not applicable	Not applicable
Dissolved organic carbon (DOC)	Pore water	Infrared spectrophotometry	Shimadzu Model TOC-5000A	0.20 mg/L	(CEAEQ 2003g)
Total dissolved phosphorus (TDP)	Pore water	Colorimetric method	Skalar San^{plus} system	0.01 mg/L	(CEAEQ 2006d)

Table A-1 cont.

Variables measured	Matrix	Method	Equipment used	Detection limit	Reference
Orthophosphate (H_2PO_4)	Pore water	Colorimetric method	Technicon Model II	0.01 mg/L	(CEAEQ 2005)
Nitrite and nitrate (NO_2-NO_3)	Pore water	Colorimetric method	Skalar San^{plus} system	0.02 mg/L	(CEAEQ 2006e)
Ammonium (NH_4)	Pore water	Colorimetric method	Skalar San^{plus} system	0.02 mg/L	(CEAEQ 2003d)

Table A-2: Summary of analytical methods used to measure the organic contaminants in sediments

Variable measured	Method	Equipment used	Detection limit	Reference
PCBs	Conger (41) method with low-resolution mass spectrometer Acetone/hexane and dichloromethane extractions Purification on silica gel column and addition of activated copper	GC/MS; Agilent, 6890N GC, 5973N MS	2–6 µg/kg	MA. 400 – BPC 1.0 (CEAEQ 2003h)
PAHs	Determination by gas chromatography/mass spectrometry Acetone/hexane and dichloromethane extractions Purification on silica gel	GC/MS; Agilent, 6890N GC, 5973N MS	0.02–0.10 mg/kg	MA. 400 – HAP 1.1 (CEAEQ 2003c)
Organochlorine pesticides	Gas chromatography/mass spectrometry Acetone/hexane extraction Purification on Florisil	GC/MS; Thermo Quest, Trace GC and Trace MS	1–18 µg/kg	MA. 416 – P. OCl 1.0 (CEAEQ 2003a)
Organophosphate pesticides	Gas chromatography/mass spectrometry Ethyl acetate extraction	GC/MS; Agilent, 6890N GC, 5973N MS	5–260 µg/kg	MA. 416 – Pest 1.0 (CEAEQ 2003f)
Aryloxy acid pesticides	Gas chromatography/mass spectrometry $NaHCO_3$ and C-18 column extractions Purification on silica gel	GC/MS; Agilent, 6890N GC, 5973N MS	1–7 µg/kg	MA. 416 – P. ChlP 1.1 (CEAEQ 2006c)
Toxaphene pesticides	Gas chromatography/electron capture detector (ECD) Acetone/hexane extraction and separation on deactivated silica gel	GC/ECD; Hewlett Packard, 5890 Series II CG/ECD	3.5 mg/kg	MA. 405 – Toxaphène 1.0 (CEAEQ 2003b)
Petroleum hydrocarbons	Gas chromatography/flame ionization detector (GC-FID) Hexane extraction	GC/FID; Hewlett Packard, 5890 Series II GC/FID	12 mg/kg	MA. 416 – C10C50 1.0 (CEAEQ 2006b)

Table A-2 cont.

Variable measured	Method	Equipment used	Detection limit	Reference
Organotins	Liquid chromatography/mass spectrometry Acetate/THF extraction	LC/MS/MS; LC: Waters, 1525 μ MS: Micromass, Quattro Ultima PT	0.6–1.5 μg/kg	–

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Appendix B: Example of Quotients Used to Estimate Ecological Risk

The first quotient tested is a mean quotient including all the contaminants covered by the sediment quality guidelines (SQGs):

$$Qmean_1 = \frac{\sum \left(\frac{C_i}{SQGs_i} \right)}{20}$$

Where C_i corresponds to the concentrations measured for each contaminant: As, Cd, Cr, Hg, Ni, Pb, Zn, total PCBs, PAH1, PAH2, … PAH11.

The second quotient, Qadd, is based on contaminant additivity. In this quotient, all the ratios between the concentration in the sediments and the quality criteria are added together:

$$Qadd = \sum \frac{C_i}{SQG_i}$$

The last aggregation example evaluated was developed for the Laurentian Lakes (Grapentine et al. 2002b; Marvin et al. 2004): sediment quality index (SQI). This index can be considered a mean quotient that takes into consideration three other elements: scope, i.e., the number of variables that do not meet the recommendations; frequency, which represents the number of times these recommendations are not met; and amplitude, which corresponds to the degree of divergence of the non-compliant measurement relative to the recommendations. This index can be calculated for each site and, in this case, only scope and amplitude are considered (Grapentine et al. 2002a; Marvin et al. 2004). The calculations were performed using SQI 0.1: Sediment Quality Index Calculator (an MS EXCEL workbook that contains macros; CCME 2006).

The predictive ability of the quotients is evaluated using a method developed by Shine et al. (2003) and by Vidal and Bay (2005), in which we consider the number of samples above or below the toxicity threshold or > and < a mean quotient value (Figure A-1):

Sensitivity = B/(A + B)
Specificity = C/(C + D)
Specificity of the toxic response = B/(B + D)
Specificity of the non-toxic response = C/(A + C)
Overall performance = (B + C)/(A + B + C + D)
Type I error = (D/(D + B))*100
Type II error = (A/(A + C))*100

In these equations, A represents the number of samples where the toxicity is significant but the Qmean < 1 (or an SQI of 80–100), B represents the number of samples where toxicity is significant and the Qmean > 1 (or an SQI < 80), and C and D represent the number of samples with non-significant toxicity, which, respectively, have quotients < or > the threshold (Figure A-1). The type II error represents the percentage of stations for which the toxicity is significant among the stations whose quotient is less than the chosen threshold, while the type I error represents the stations for which there was no toxicity observed among the stations whose quotient is higher than the chosen threshold. In this portion of the study, only the toxicity tests considered the most promising were retained, i.e., mortality of *Hyalella azteca* and *Chironomus riparius* and inhibition of reproduction of *Brachionus calyciflorus* (see Tier 2).

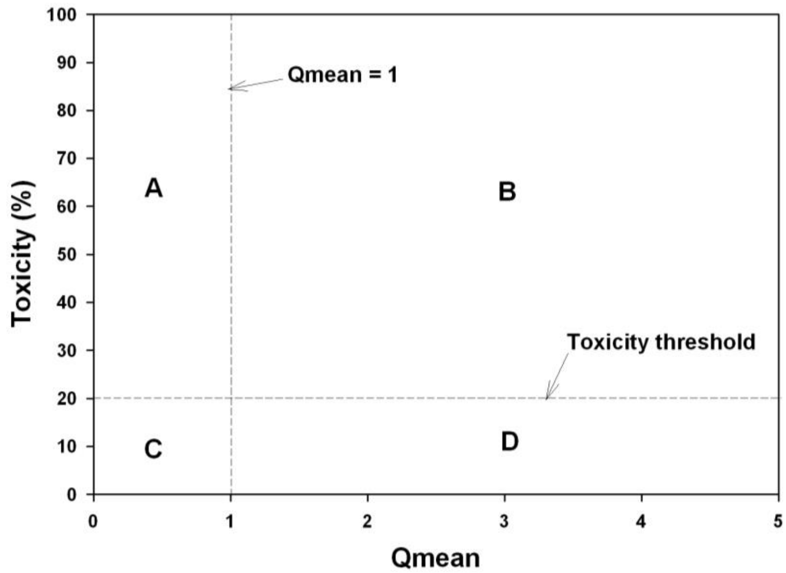

Figure A-1: Example of thresholds to be determined in order to evaluate the predictive ability of the quality criteria (inspired by Shine et al. 2003; Vidal and Bay 2005)

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