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CONCEPTUAL SITE MODEL AND RECOMMENDATIONS

ST. MARYS RIVER SEDIMENTS
REVISION 7

RAMBOLL

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ACRONYMS AND ABBREVIATIONS

Algoma	Algoma Steel Inc.
AOC	Area of Concern
Aqua Terre	Aqua Terre Solutions Inc.
AquaTox	AquaTox Testing and Consulting Inc.
BEAST	BEnthic Assessment of SedimenT
BOD	biological oxygen demand
BPAC	Binational Public Advisory Council
BUI	beneficial use impairment
COA Framework	Canada-Ontario Agreement
COPC	chemicals of potential concern
CSM	conceptual site model
CSO	combined sewer overflow
EBMP	East of Bellevue Marine Park
ECCC	Environment and Climate Change Canada
Golder	Golder Associates Ltd.
IJC	International Joint Commission
JWEL	Jacques Whitford Environmental Limited
Kg	kilogram(s)
LEL	lowest effect level
m	metre(s)
m^2	square metre(s)
m^3	cubic metre(s)
mg/kg	milligram(s) per kilogram
MECP	Ontario Ministry of Environment, Conservation and Parks
MRW	M.R. Wright and Associates Co. Ltd.
PAH	polycyclic aromatic hydrocarbon
PBDE	polybrominated diphenyl ethers
PCB	polychlorinated biphenyl
PEL	probable effect level
PWGSC	Public Works and Government Services Canada
PWQO	Provincial Water Quality Objective
Ramboll	Ramboll US Corporation
ROPC	receptor of potential concern
SEL	severe effect level
SMO	sediment management option
SMR	St. Marys River
SNC-Lavalin	SNC-Lavalin Environment
SSMIC	Sault Ste. Marie Innovation Centre
TBT	tributyltin
TIE	toxicity identification evaluation
TOC	total organic carbon
TPH	total petroleum hydrocarbon
TSH	Totten Sims Hubicki Associates
USEPA	U.S. Environmental Protection Agency

1. INTRODUCTION

This report describes a conceptual site model (CSM) for sediments in St. Marys River (SMR) Area of Concern (AOC) and provides recommendations to support the contaminated sediment management strategy for the AOC. SMR flows eastward, linking Lake Superior and Lake Huron, and forms the border between Ontario and Michigan in the vicinity of Sault Ste. Marie (Figures 1 and 2). This CSM is intended to represent the current state of understanding of the sources, extent, fate and transport of contaminants, as well as their contact by human and ecological receptors. This CSM provides an organized framework for understanding and communicating current conditions at the site relative to the potential for contaminants to interact with humans and the environment, in order to aid in effective remediation decision-making. This CSM is intended to help Environment and Climate Change Canada (ECCC), Ontario Ministry of the Environment, Conservation and Parks (MECP), Algoma University, Binational Public Advisory Council (BPAC), and other stakeholders to address contaminated sediments in SMR.[1]

The overall purpose of this report is to reach one of the following three conclusions with respect to risks posed and the need for further investigation and/or risk management actions:

1. Sufficient evidence exists to conclude that current conditions in the AOC do not pose a significant risk to human health and/or the environment and therefore risk management actions are not warranted.
2. Insufficient evidence exists to draw conclusions regarding risks to human health and/or the environment under current conditions and further investigation and/or monitoring is warranted to make those conclusions.
3. Sufficient evidence exists to conclude that current conditions pose significant risks to human health and/or the environment and therefore risk management actions are warranted in specific locations.

In light of that underlying purpose, specific objectives of this CSM are to:

- Characterize contaminant sources, migration pathways, human and ecological receptors, exposure pathways, and the linkages among these components;
- Provide an information framework to support development of a contaminated sediment management strategy for the river;
- Identify key issues and knowledge gaps that influence such a strategy, such as the cause of observed toxicity in East of Bellevue Marine Park (EBMP); and
- Offer recommendations that will assist stakeholders in formulating the contaminated sediment management strategy for the AOC.

Ramboll US Corporation (Ramboll) prepared this CSM largely based on documents provided by ECCC and MECP. A list of the documents reviewed is provided in Section 8 (References) of this report. Like

[1] The current names for ECCC and MECP are used throughout this report for consistency, even when referencing reports that were issued by either agency before the most recent name change.

most CSMs, this one is intended to be iterative; as additional information becomes available, it can be updated. This version supersedes all prior versions.

The remainder of this report is organized as follows:

- Section 2. Nature and Extent of Contamination: Historical and ongoing contaminant sources are identified, including likely chemicals of concern and the status of source mitigation efforts. In addition, Section 2 presents the current concentrations and spatial distribution of contaminants in sediment.
- Section 3. Contaminant Migration Pathways, Fate and Transport: This section identifies important and minor pathways by which contaminants may reach biological receptors and discusses natural recovery mechanisms.
- Section 4. Sediment Stability: This section discusses factors that affect whether contaminants in buried sediments are likely to remain physically isolated from biological receptors into the future.
- Section 5. Sediment Related Risks: Current knowledge of sediment-related risks to benthic invertebrates, fish, wildlife, and humans is summarized.
- Section 6. Summary of Key Findings and Knowledge Gaps: This section integrates information from the foregoing sections in order to assess impairment within the AOC, identify data gaps, and offer specific recommendations for developing a contaminated sediment management strategy.
- Section 7. Conclusions: This section assesses if current conditions pose a significant risk to the environment based on available information and based on comparisons of sediment and fish tissue chemistry results to screening-level benchmarks.
- Section 8. References.

2. NATURE AND EXTENT OF CONTAMINATION

This section of the CSM details the nature and extent of contamination within the SMR AOC, considering historical and ongoing sources, current conditions, and temporal trends in sediment chemistry.

2.1 Historical and Ongoing Sources

Understanding the range of known and suspected sources—both ongoing and historical—is a critical precursor to the characterization of contamination, investigation of contaminated sediments, and evaluation of sediment management options (SMOs). Source control is critical to the success of any SMO, including both natural recovery and engineered measures. Understanding and managing sources is critical to monitoring and quantifying recovery processes. In the absence of a solid understanding of sources, recovery may fall short of goals and/or remediated areas may be re-contaminated. This is not to say that elimination of all sources is a prerequisite for successful sediment management, but rather that any ongoing sources should be understood well enough to judge the adequacy of expected risk reductions to be achieved through candidate SMOs.

An illustration of major historical sources and exposure pathways for contaminants in the sediment of SMR is provided in Figure 3. Primary historical and potential ongoing sources of sediment contaminants include:

- Algoma Steel Inc. (Algoma; formerly Essar Steel Algoma and Algoma Inc.)
- St. Marys Paper (formerly Abitibi Paper Company) [decommissioned]
- Municipal wastewater treatment facilities
- Consumers Energy former manufactured gas plant (USA) [decommissioned]
- Tannery Bay/ Cannelton Industries, Inc. Superfund Site (USA) [decommissioned]
- Transport Canada Water Lot (including the Purvis Marine Site)
- Non-Point/background sources (e.g., storm sewer discharges, urban runoff, atmospheric deposition)

For major sources located in Ontario, MECP (2011) compiled effluent discharge data that document source control from the early 1980s through 2009 (Figures 4 through 7). In addition to these primary sources, contaminated sediments within the river may act as secondary sources, to the extent that sediment and contaminant transport processes continue to move contaminants downstream. Each of the primary sources is briefly discussed below.

2.1.1 Algoma (Ontario)

An active steel manufacturing facility, Algoma[2] was historically a source of oil, polycyclic aromatic hydrocarbons (PAHs), and metals to SMR (Kauss 2000, Pope and Kauss 1995). Other operations linked to the steel plant, such as a Domtar facility that processed coal tar from the plant's coke ovens, also contributed to these waste streams (Hamdy et al. 1978). The steel manufacturing facility also was a source of cyanide and phenols (Kresin 2008), but these contaminants are not typically persistent in sediments and have not been identified as significant sediment contaminants after multiple investigations (Kresin 2004).

Notable features of the Algoma site include the former main trunk sewer outfall, Algoma Boat Slip (formerly Essar Steel Algoma Slip), and a waste disposal and slag management area sometimes referred to as the Algoma Slag Dump (Figure 2). Algoma discharges effluent to SMR and tributaries (e.g., Bennett Creek) through multiple outfalls located both upstream and downstream of the St. Marys Rapids. The former main trunk sewer (Figure 2) combined discharges from Algoma and St. Marys Paper (Ripley et al. 2011); those facilities' discharges are now separate. The Algoma Boat Slip is located adjacent to the steel manufacturing facility, at the confluence of Bennett Creek and East Davignon Creek with SMR. Sediments in the slip were historically highly contaminated. The Algoma Slag Dump is an active slag handling and waste disposal area located adjacent to the river, upstream of the Algoma Boat Slip.

The following source control measures have been implemented on the overall Algoma property (ECCC et al. 2002; AMEC 2004):

- The wastewater treatment plant was upgraded between 1997 and 1999, which reduced phenol, ammonia, cyanide, oil and grease, and suspended solids concentrations in wastewater and optimized water re-use by up to 90%.
- Upgrades and refurbishment activities have been occurring on all three coke oven batteries since 2016 to control air emissions of particulate matter and PAHs (namely benzo(a)pyrene), which resulted in significant reductions in emissions from those processes. Installation of a blast furnace contact water recirculation facility in 1998 reduced ammonia and cyanide discharges
- A coal tar collection system was installed in 1990 to address contaminated groundwater migration to the river.
- Partial dredging of the Algoma Boat Slip in 1995 (11,500 cubic metres or m^3) for maintenance purposes
- Additional maintenance dredging of the Algoma Boat Slip in 2006 (2,630 m^3) and in 2017-2019 (10,900 m^3 + 6,198 m^3). Algoma will conduct a post-dredge sediment assessment to assess what contaminated sediment remains in the slip.

As a result of these measures, discharges of contaminants in effluent from Algoma, notably oil and grease (solvent extractables), decreased dramatically in the mid-1990s (Figure 4).

The role of the Algoma Slag Dump as a possible ongoing source of contaminants to SMR remains a subject of investigation and monitoring. Past studies found "little or no impact on SMR water quality"

[2] Algoma is used throughout this report for consistency, even when referencing activities that were conducted at the site before the name change.

from groundwater discharges in this area (ECCC et al. 2002). In 2019, drilling began on a program to establish a site wide monitoring program, comprising 31 existing monitoring wells largely centered around the landfill and materials recycling area, plus 63 new monitoring wells. Algoma has implemented a greening program, using St. Marys Paper biosolids to initiate the growth of grass and other plants along the water's edge (MECP 2011).

In 2010, Algoma completed a year-long sampling and monitoring program along the Algoma Slag Dump shoreline to determine if there were any off-site impacts from contaminants leaching from the industrial landfill portions of the Slag Dump (CRA 2011). Shallow groundwater along the perimeter of the Slag Dump area exhibited concentrations of several constituents above Provincial Water Quality Objectives (PWQOs), including pH, un-ionized ammonia, high molecular weight PAHs, certain metals (notably vanadium), and phenols. Acute toxicity tests conducted on groundwater samples from the shallow perimeter wells showed mortality in rainbow trout (*Oncorhynchus mykiss*) and the water flea (*Daphnia magna*), which was attributed to the pH greater than 12. In general, the chemical concentrations in the shallow perimeter groundwater were greater than in groundwater from presumed landfill source areas, indicating that the shallow perimeter groundwater quality was likely affected by the slag fill material that constitutes the land in this area (CRA 2011). Of the chemicals that exceeded PWQOs in perimeter groundwater samples, PAHs and metals are most relevant to sediment contamination in SMR (Figure 3). Ammonia and phenols are not persistent in sediment, and elevated pH readily attenuates through dilution with river water. CRA (2011) did not collect sediment data in the vicinity of the perimeter groundwater wells. However, as described in Section 4, sediment deposition is limited in the vicinity of the Slag Dump, such that contaminants released to the river through groundwater discharge are likely dispersed downstream.

Algoma continues to monitor shallow groundwater along the perimeter of the Slag Dump. On May 2, 2019, Algoma was issued an Environmental Compliance Approval for treatment of stormwater and seep water reporting to the baseline road ditch area north of the Materials Storage and Reprocessing Site. Results of the monitoring program will be incorporated into the CSM when they are made available.

In addition to the monitoring program along the Slag Dump shoreline, CRA (2011) assessed sediment chemistry and benthic invertebrate community characteristics in the Bennett Creek Diversion, which is northeast of the Slag Dump and flows into the Boat Slip. Sediments upstream of the Boat Slip contained concentrations of PAHs up to 526 milligrams per kilogram (mg/kg) and supported few or no benthic invertebrates (CRA 2011). Thus, to the extent that contaminated sediments may have migrated from the Bennett Creek Diversion into the Algoma Boat Slip, that area may be a source of contaminants to SMR sediments. While former dredging events have been focused on maintaining shipping access, the 2017 to 2019 dredging program was focused on removing impacted sediment from the boat slip. Some historical sediments likely remained in the boat slip after the 2019 dredging event. Post- dredge monitoring, which will allow evaluation of ongoing impacts to the sediments.

The Algoma site-wide groundwater monitoring program that began in 2019 targets legacy areas on either bank of Bennett Creek. That monitoring will help to determine if there are source areas that could be addressed to reduce impacts to creek sediment.

Several investigations have focused on sediment quality within the boat slip. On behalf of Algoma, Totten Sims Hubicki Associates (TSH) collected sediment samples from the boat slip in 2005 (TSH 2006) prior to the dredging operations in 2006. Ten surface sediment samples and ten core samples were collected and analyzed for bacteria (fecal and total coliforms), leachate quality, loss on ignition, PAHs, total petroleum hydrocarbons (TPH), oil and grease, and total organic carbon (TOC). Many of the samples collected from boat slip had a slight to moderate petroleum hydrocarbon odor. TSH reported that black particles (likely coal) were visible in many samples. PAHs were detected in all surface and subsurface sediment samples at concentrations greater than MECP's lowest effect level (LEL). In one surface sediment sample and three subsurface samples, concentrations of PAHs also exceeded the severe effects level (SEL). There were no exceedances of the parameters analyzed using the Toxicity Characteristic Leaching Procedure. When compared with concentration in sediment samples collected from the boat slip in 2000, average total PAH concentrations decreased from 499 mg/kg in 2000 to 289 mg/kg in 2005. Maximum total PAH concentrations decreased from 2,347 mg/kg in 2000 to 1,571 mg/kg in 2005.

To document changes in sediment quality over time, Pinchin Ltd. (on behalf of Algoma) surveyed the boat slip in 2014 and analyzed sediment samples for the same parameters that were analyzed in 2000 and 2005 (Pinchin 2015). During the 2014 survey, Pinchin collected 16 surface sediment samples and 24 core samples. Pinchin retained AquaTox Testing and Consulting Inc. (AquaTox) evaluate the results relative to the pre-dredge 2005 concentrations. AquaTox found that concentrations of PAHs in sediment increased after 2005. For sample locations that were duplicated between 2005 and 2014, the 2014 samples had average total PAH concentrations approximately 30% higher than those sampled in 2005. Across all samples, average total PAH concentrations were 90% higher in 2014 than in 2005. While PAH impacts were most notable, some elevated concentrations of TPH, total metals, and oil and grease also were observed. Pinchin concluded that the elevated concentrations observed in 2014 may have resulted from residual contamination that had not been remediated during the 2006 dredging of the boat slip. They hypothesized that vessel traffic may have distributed residual contamination throughout the boat slip.

On May 6, 2019, MECP issued a program approval under the Ontario Environmental Protection Act for Algoma's Legacy Environmental Action Plan, through which the company will assess and address historical contamination at the steel mill, landfill site, and related properties in Sault Ste. Marie. Algoma proposed a program to address legacy impacts at the site by undertaking a site-wide assessment to identify issues, followed by a risk-based plan to assess, and mitigate off-site risks associated with legacy environmental contamination. In 2019, Algoma commenced the site wide assessment, which will prioritize future activities to reduce off-site impacts.

Additional dredging of the boat slip commenced in 2017 and removed 10,900 m^3. In 2018, Pinchin conducted an additional sediment survey of the boat slip on behalf of Algoma, to assess the change in sediment quality from the previous assessments (Pinchin 2018). The 2018 survey included collection of 18 surface samples using an Eckman dredge and 29 sediment core samples; all samples were collected from the same locations as samples collected during the 2014 survey and analyzed for the same parameters. Pinchin again contracted AquaTox to interpret the sediment chemistry results. AquaTox (2018) found that total metals concentrations in sediment collected from the boat slip in 2018 were consistent with concentrations in 2014. Manganese was the only metal that exceeded the sediment SEL in some samples; none of the samples had metal concentrations that exceeded MECP's probable effects level (PEL) (AquaTox 2018). In contrast, average concentrations of PAHs exceeded their SEL in 2018, including fluorene, phenanthrene, anthracene, fluoranthene, pyrene, and chrysene. These same PAHs exceeded the SEL in 2014. Despite dredging in 2017, concentrations of total PAHs did not decline between 2014 and 2018, suggesting that uncontaminated sediment has not mixed sufficiently with the dredging residuals to reduce PAH concentrations (AquaTox 2018). Elevated concentrations of metals and TPH also persisted in the boat slip sediment. AquaTox concluded that removal of the remaining contaminated sediments was critical to restoring SMR, as these sediments likely were an ongoing source of contamination to the river. An alternative interpretation is that residuals have not been sufficiently managed to ensure the effectiveness of dredging.

In 2019, Algoma removed additional 6,198 m^3 of contaminated sediment. Algoma will be conducting post-dredge monitoring study to assess the quality of sediments remaining in the boat slip.

In addition to its direct discharges to SMR, Algoma's air emissions could affect the river through air deposition to the watershed and subsequent surface runoff of particulates, as well as air deposition onto the river itself. Air emissions of PAHs are of primary interest with respect to contributions to sediment contamination. The main sources of PAHs released from the facility are fugitive emissions from the facility's three coke oven batteries. MECP required that Algoma install individual oven pressure controls for the #9 coke oven battery. This installation is the best available technology for reducing coke oven emissions. Other improvements to air emissions controls included a new baghouse to reduce iron oxide emissions, a new cogeneration plant that reduces the need to flare blast furnace gas, and a road dust suppression and paving plan (MECP 2011). As of 2019, Algoma had completed improvements to their coke battery # 7, 8 and 9 and added the Dekishing baghouse as outlined in the site-specific standard for suspended particulate matter. Algoma implemented a monitoring program for particulate and metals emissions and continue to reduce fugitive emissions from roadways through their road paving program. In addition, Algoma is in compliance with the 2020 visible emissions limits, as required by the site-specific standard.

2.1.2 St. Marys Paper (Ontario) [decommissioned]

Until 2011, when operations ceased, St. Marys Paper manufactured paper immediately to the east of Algoma (Figure 2). The major contaminants historically released to SMR from St. Marys Paper were resin acids and bulk waste materials (paper fines, wood fibres and wood chips), which were associated with elevated suspended solids and biological oxygen demand (BOD). Hamdy et al. (1978) listed the paper mill as a historical source of oil releases. Although historical accounts identified chlorine bleaching of pulp as part of the early industrial development of Sault Ste. Marie (Sault Ste. Marie Public Library 2008), St. Marys Paper relied on sodium hydrosulphite as a brightening agent (AMEC 2004, Beak International 2000). Dioxins and furans, which often are associated with chlorine bleaching, are not of concern in SMR sediments (see Section 3.3).

In the mid-1990s, significant upgrades to effluent and air emissions treatment systems at St. Marys Paper largely mitigated releases (ECCC et al. 2002). The biggest improvement came in late 1995 with the installation of a secondary wastewater treatment system with activated sludge, which reduced loading of total suspended solids, BOD, total resin acids, toluene, and phenol (AMEC 2004, Beak International 2000). Figure 5 illustrates a large reduction in particulate and BOD discharges from St.

Marys Paper in 1996. Stormwater runoff from most of the property was routed through the effluent discharge; thus, the data presented in Figure 5 reflects stormwater runoff (MECP 2011). Even when the mill was operational, effluent was not expected to cause significant environmental effects. Based on effluent toxicity monitoring results and information on dilution of the effluent in the river, sublethal effects on invertebrates were expected to be limited to the immediate vicinity of the discharge (within 5 m), and no effects on fish were expected (Stantec 2007, St. Marys Paper Corp. 2010).

2.1.3 Municipal Wastewater Treatment Facilities

Three municipal wastewater treatment facilities are located within in the Sault Ste. Marie area (Figure 2):

- West End Water Pollution Control Plant (Ontario): average flow 8,500 m^3/day
- East End Wastewater Treatment Plant (Ontario): average flow 41,400 m^3/day
- Sault Ste. Marie Wastewater Treatment Plant (Michigan): average flow 9,500 m^3/day

Contaminants typically associated with wastewater treatment facilities include solids, organic matter/oxygen demanding substances, nutrients, pathogens, and trace contaminants.

Since 1995, wastewater effluent quality at the Sault Ste. Marie plants has substantially improved. In Ontario, the West End plant was constructed as a secondary treatment facility in 1985 and has an excellent record of performance with respect to compliance with environmental requirements (Elliott 2008). As shown in Figure 6, effluent discharges from the West End plant have been relatively consistent over time and are much lower than those from other major point sources in Ontario (Figures 4 through 7). In 2006, the older and larger East End plant was upgraded from primary to secondary treatment, employing biological nutrient removal and ultraviolet light for disinfection. The resulting improvement in effluent quality is evident in Figure 7.

In addition, the outfall pipe with discharge diffuser was positioned into deeper waters of the river. In Sault Ste. Marie, Ontario, there are no combined storm and sanitary sewers (i.e., CSOs); the systems are separate (Elliott 2008). Until 1968, homeowners were permitted to cross-connect their foundation drains into the sanitary sewers. During road reconstruction projects, the city provided residents connections to the sanitary and storm sewer systems, but older connections continue to overload the city's wastewater treatment plants during snow melts and heavy rainfalls (Hamilton Beach 2011).

Sault Ste. Marie, Michigan has one secondary wastewater treatment plant that serves a population of 15,000. There are numerous storm sewer and drainage outlets located along the Michigan shoreline, and the city has been undertaking a 30-year program to separate CSOs from sanitary sewers to better control discharge of sewage into the river. The city has made significant progress and few locations need work over the next few years.

2.1.4 Consumers Energy Former Manufactured Gas Plant (Michigan) [decommissioned]

From the 1900s through the 1940s, a manufactured gas plant operated adjacent to SMR (Figure 2). Plant operations contributed coal tar and PAHs to sediment contamination. Dredging of shoreline deposits at this site was initiated in 2010 (USEPA 2010). The project aimed to remove all sediment with concentrations of PAHs greater than 115 mg/kg or a toxicity unit greater than 1. During Phase I of the sediment remediation project, approximately 6,000 m^3 of contaminated nearshore sediment was dredged adjacent to the former plant. In Phase II of the project, which was completed in 2011, approximately 15,000 m^3 of PAH-contaminated sediment was dredged from the natural channel portion of SMR (CH2M Hill 2012). Following dredging, an approximately 7,000 square metres (m^2) sand cap was placed in the river. The remediation project was completed in November of 2011 (Tuchman 2012).

The 2010 dredging project used booms and silt curtains to minimize contaminant releases, and water quality was monitored (An 2011). Post-dredge monitoring results indicated that total PAH concentrations exceeded 115 mg/kg in 26 samples (41% of samples) (CH2M Hill 2012). The sand cap was placed over all locations with post-dredge concentrations greater than 115 mg/kg. The river flow patterns and velocities documented by Krishnappan (2011) indicate that any releases associated with the dredging project would not have likely migrated to the Canadian side of the river.

2.1.5 Tannery Bay/Cannelton Site (Michigan) [decommissioned]

The Northwestern Leather Company tannery operated along SMR during the early 20th century (Figure 2). Although chromium was the primary contaminant at this site, it was present in sediment in a relatively non-bioavailable form, and sediments at the site were found to be non-toxic (Golder 2004). Mercury, cyanide, and sulphide were secondary contaminants of concern (Ripley et al. 2011). Dredging completed in 2007 removed 31,000 m^3 of sediment from Tannery Bay (USEPA 2007).

In its October 2, 2019 agency update to BPAC, Michigan Department of Environment, Great Lakes, and Energy reported that a sampling plan was in development for this site to verify that remediation is complete. Associated sampling was planned for 2020 and was expected to inform delisting decision on the US side for the beneficial use impairment (BUI) related to impaired benthos.

2.1.6 Transport Canada Water Lot

Transport Canada owns a 245-ha water lot on the Ontario side of SMR, extending from Old Vessel Point to Church Street 9 kilometres downstream (Figure 2). The major industrial activities that have impacted the water lot include Algoma and St. Marys Paper plant, as well as vessel traffic and hydroelectric power generation (Golder 2008). Transport Canada classified the water lot as a high priority aquatic site in 2007 and implemented the Canada-Ontario Contaminated Sediments Decision-Making Framework (COA Framework) (ECCC and MECP 2008) at the site to understand alternatives for managing contaminated sediments.

In 2008, Public Works and Government Services Canada (PWGSC) retained Golder Associates Ltd (Golder) on behalf of Transport Canada to complete Screening Steps 1-3 of the decision-making framework. The goal of Step 1 was to develop an initial CSM for the water lot, showing the relationships between chemicals of potential concern (COPCs) and ecological and human receptors. Based on existing site data, Golder identified the following COPCs in water lot sediment: metals (including methylmercury), PAHs, cyanide, polychlorinated biphenyls (PCBs), tributyltin (TBT), dioxins and furans, organochlorine pesticides, TPH, and oil and grease (Golder 2008). Of these, only trace metals and PAHs were detected in sediment at concentrations exceeding the Canadian Council of Ministers of the Environment's Interim Sediment Quality Guidelines and LELs. In addition to metals and PAHs, the other COPCs listed above (e.g., PCBs, dioxins and furans, organochlorine pesticides, TPH, oil and grease) were evaluated given their elevated detection limits, the association of the COPCs to known activities in the water lot (e.g., TBT associated with ship repair and maintenance activities), and low detections in historical sampling. Golder identified benthic invertebrates as the primary receptor of potential concern (ROPC). However, the report also considered demersal fish, piscivorous fish, and piscivorous wildlife given the potential for biomagnifying substances to be present in sediment. Based on an assessment of the ROPCs, exposure pathways, reference locations, assessment endpoints and measured effects, Golder (2008) concluded that the COPCs (i.e., metals, PAHs, PCBs, organochlorine pesticides and TBT) posed a potential risk to ecological receptors and recommended further assessment of water lot sediment.

In 2009 PWGSC retained Aqua Terre Solutions Inc (Aqua Terre). on behalf of Transport Canada to conduct a sediment assessment of the Transport Canada water lot sediment. The scope of work included: 1) developing an assessment strategy to characterize sediment on the basis of a data gap analysis of the findings of previous investigations, 2) completing an investigation of the sediment quality and evaluation of the data with respect to the COA Framework, Step 5 Decision Point 4, and 3) providing conclusion regarding the risk to human health and the environment under current conditions and providing recommendations for management of risk. The study concluded that management action would likely be required at the site.

Prior to identifying management actions, the following additional work was proposed: 1) measure tissue concentrations in fish obtained from the site or perform laboratory bioaccumulation studies, 2) assume that fish species area use factor is 1, with fish feeding and mobility confined to site limits, 3) if necessary to complete a detailed risk assessment, a consumption rate study involving anglers and local First Nations individuals could be undertaken to define local consumption rates, 4) field studies or laboratory studies to identify an appropriate site-specific fish biota-sediment accumulation factor for dioxins and furans could be undertaken, 5) identify the cause of sediment toxicity and benthos alterations by evaluating COPCs.

In 2010, PWGSC retained SNC-Lavalin Environment (SNC-Lavalin) on behalf of Transport Canada to conduct a detailed quantitative risk assessment of sediment within the water lot to support the development of a contaminated sediment management strategy (SNC-Lavalin 2010). The scope of work included 1) conducting a data gap analysis, 2) further characterizing areas predicted to be at elevated risk based on sediment concentrations, 3) conducting a detailed quantitative assessment of biomagnification potential, sediment toxicity, and benthic community structure, and 4) evaluating appropriate sediment management actions, including whether risks can be managed satisfactorily in-place or if additional targeted study was required. Considering the work conducted by Golder (2008) and AquaTerre (2009) and undertaking its own independent assessment of COPCs, SNC-Lavalin assessed trace metals, PAHs, dioxins and furans, and oil and grease as COPCs. SNC-Lavalin noted that, when normalized for TOC in sediment, dioxin and furan concentrations in water lot sediment were not greater than concentrations in reference area sediment. In addition, evaluation of biomagnification potential of methylmercury and dioxins and furans through a fish tissue sampling program found that measured concentrations of methylmercury and dioxins and furans in fish tissue were not expected to result in adverse effects in human or ecological receptors.

MECP, in collaboration with the University of Toronto, is in the process of finalizing an assessment of the fish consumption BUI within the AOC. The assessment notes that mercury in the majority of species (pink salmon was an exception) and decreases in dioxin and furan concentrations during the 2000s. This finding was based on limited sample sizes for pink salmon and rainbow trout. PCBs were detectable in the majority of fish species (northern pike was an exception).

Based on the 2008 sediment sampling, SNC-Lavalin (2010) found that three areas within the water lot had potential or significant toxicological effects, as well as possible benthic community alteration. Following the COA Framework Step 6, Decision 5, SNC-Lavalin (2010) concluded that no further action was required at four sampling sites (S-4, S-7, S-12 and S-18). Benthos alteration at six sites (S-1, S-6, S-8, S-9, S-13 and S-17) was attributed to physical factors. Sediment toxicity testing suggests no toxic effects at these six sites. SNC-Lavalin (2010) concluded that three sites (S-10, S-15 and S-16) required further assessment to determine if management action is warranted for sediment toxicity and possible alteration of benthic community.

In 2011, Public Works and Government Services Canada retained SNC-Lavalin on behalf of Transport Canada to refine the assessment of Transport Canada's water lot, following Step 6 and Decision 5 of the COA Framework. Step 6 involves conducting further assessments, if necessary. Decision 5 determines whether environmental risks exist. The scope of work included assessment of S-10, S-15 and S-16 by: 1) achieving lateral delineation of contamination within the Transport Canada water lot to identify any surficial sediments, 2) determining the relationship, if any, between concentrations of COPCs (i.e. nutrients, TOC and total Kjeldahl nitrogen, PAHs, and oil and grease) and the magnitude of toxicological response, 3) determining the level of response attributable to COPC concentrations, and 4) identifying possible linkages of findings to spatial characterization of risks at S-10, S15 and S-16.

SNC-Lavalin (2011) concluded that S-10 and S-15/S-16 can be addressed through in-place management, involving a sediment monitoring program with samples collected from at least three sampling locations within each station. At S-10, SNC-Lavalin recommended using toxicity results to refine the lateral extent of sediments. At S-15 and S-16, SNC-Lavalin recommended using toxicity results to confirm that conditions are not deteriorating. The authors further recommend that the monitoring program be conducted every two years for the first four years; the frequency of monitoring can be reassessed based on the results of the first two rounds of monitoring. They recommend establishing approval procedures to maintain current conditions. They advised that, before any proposed future dredging is initiated, additional assessment should be conducted to evaluate potential effects resulting from the exposure of deeper sediments.

In 2018, Transport Canada retained SNC-Lavalin to conduct sediment sampling for chemistry and toxicity testing (Figure 8). The goal of this work was to update Step 6 and Decision 5 within the COA Framework. In October 2018, SNC-Lavalin sampled 9 locations, collecting 31 samples and 6 duplicates, all of which were analyzed for bulk sediment chemistry and 16 of which also were tested for toxicity. In addition, six reference samples were analyzed for bulk sediment chemistry and two were tested for toxicity. This study's design reflected the conclusions of the previous assessments and targeted the following objectives:

- Delineate the lateral extent of sediment exhibiting significant overall toxicity at Station S-10;
- Confirm that conditions are not deteriorating at Stations S-15 and S-16; and
- Confirm that conditions at Stations S-1, S-6, S-8, S-9, S-13, and S-17 have not worsened relative to historical findings.

Findings are illustrated in Figure 8. Near Station S-10, three of four stations tested showed negligible overall toxicity, and one suggested significant overall toxicity. Sediments over an estimated area of 3,400 m^2 have the potential to pose significant toxic effects to benthic invertebrates, though the ranges of toxic responses for all endpoints were within the ranges observed in reference samples. The authors concluded that Station S-10 can be satisfactorily addressed through in-place management (i.e., no action or monitored natural recovery, as defined by SNC-Lavalin, 2019).

Near Stations S-15/S-16, all sediment samples collected in 2018 exhibited negligible overall toxicity. Among these were five locations that had previously indicated potential toxicity and six that previously had concentrations greater than the effect range high concentrations. The authors concluded that sediments in the vicinity of Stations S-15/S-16 also can be satisfactorily addressed through in-place management.

Confirmatory sediment chemistry monitoring at select other stations yielded variable results. Concentrations of PAHs in sediment have increased since 2008 at Stations S-8 and S-9. At the remaining four stations (S-1, S-6, S-13, and S-17), conditions have either remained the same or improved slightly since 2008. With respect to concentrations of metals, sediment quality has deteriorated at stations S-6, S-9, S-13, and S-17) since 2008. The authors concluded that these stations can be satisfactorily addressed through in-place management.

In summary, next steps to be undertaken at the Transport Canada water lot involve continued monitoring at the nine stations, every five years. The next round of monitoring will be conducted in 2023 to confirm that conditions are not worsening based on bulk chemistry and toxicity testing. At that point, the frequency of future monitoring will be reassessed; if conditions are improving, monitoring may be less frequent. Finally, if dredging is contemplated in the future, characterization of deeper sediment will be considered.

2.1.7 Non-Point and Background Sources

Local diffuse sources of contaminants to watersheds typically include urban runoff from streets and parking lots (oil, PAHs, metals, bacteria, pesticides and nutrients), agricultural runoff (pesticides, nutrients), septic system discharges (nutrients, pathogens), and atmospheric deposition (mercury, PCBs). In a study specific to Sault Ste. Marie, Ontario, Stone and Marsalek (1996) demonstrated that "street sediment" contained somewhat elevated concentrations of bioavailable metals, but that common runoff control measures such as street cleaning and stormwater detention could eliminate much of the potential loading to SMR. Generally, nonpoint sources are considered minor contributors of contamination to SMR, relative to the point sources identified above. However, they may be the primary ongoing sources of low levels of biomagnifying compounds (mercury, PCBs) in the system.

Episodic fuel releases from commercial shipping and recreational vessels also have contributed contaminants to SMR (ECCC et al. 2002). A review of recent spill report provides an indication of their low frequency; no incidents were reported on SMR from 2007 through mid-2009 (U.S. Coast Guard et al. 2010; Transport Canada 2009). In 2012 and 2013, ECCC commissioned an assessment on vessel-based discharges to the SMR based on the Canadian Coast Guard's "Marine Pollution Incident Reporting System" database (French and Sutton 2013). Covering a decade of reported cases from 2001 to 2011, the report concludes, "the number of vessel discharge incidents within [SMR] AOC vary from year to year but remains fairly low" (French and Sutton 2013).

The placement of cleared snow from Sault Ste. Marie, Ontario at a "snow pile" location near the SMR also have been identified as a potential source of contamination to the river. MECP and ECCC funded a grant to investigate the significance of this potential source. The results of the study indicated that the runoff exceeded the guidelines, but the flow in SMR is sufficient to provide the necessary dilution of the contaminants. The report concluded that, while SMR provides sufficient dilution, the city may elect to implement treatment (pers. comm. 2019. City of Sault Ste. Marie and Mark Chambers, ECCC).

2.2 Characterization of Current Conditions

In order to understand the nature and extent of potential contaminants in SMR sediment, numerous investigations have been conducted in the AOC over the last several decades. Over time, the focus of investigations has narrowed to the Algoma Boat Slip (described in Section 2.1.1) and the area east of Topsail Island, known as East of Bellevue Marine Park (EBMP). The Algoma Boat Slip has completed its maintenance dredging and will be conducting post-dredge monitoring to assess the quality of the remaining sediments. Discussed below is the current state of understanding for current conditions and temporal trends in contaminant concentrations at Algoma Slag Dump, Lake George Channel, Little Lake George, Lake George, and EBMP.

Within the Algoma Slag Dump, Lake George Channel, Little Lake George and Lake George, few stations have been sampled repeatedly over time. Consequently, limited data are available that would allow temporal trends to be tracked. Based on data reported by Jacques Whitford Environmental Limited (JWEL) (2002) and George (2006) for the period from 1989 to 2005, declines in concentrations of PAHs were observed at all of the above locations except Little Lake George. Declines in concentrations of oil and grease in surface sediment were observed at all the above locations except Algoma Slag Dump. Variability in the depth intervals sampled over time, however, makes it difficult to differentiate true changes in chemical concentrations over time from differences in sample design. For example, JWEL's (2002) report described a study undertaken in 1999, in which surface sediment was defined as 0-5 cm. George (2006) described a study undertaken in 2005, in which surface sediment was defined as 0-3 cm. George (2006) also summarized the 1992 sampling data collected by Arthur and Kauss (2000), in which sediment was collected from the upper 10 cm.

Surface sediment (0-10 cm) sampled by ECCC between 2002 and 2010 showed no clear pattern of declining PAH concentrations over time (Figures 9a-d). Because a temporal trend analysis was not the focus of the ECCC sampling conducted between 2002 and 2010, there was little overlap in sampling locations from year to year. Lake George Channel was the only area sampled by ECCC in 2009/2010 that also was sampled and analyzed for temporal trends by JWEL (2002). A comparison of the average total PAH concentration of samples collected in 2009/2010 to the average total PAH concentrations of samples collected between 1989-1999 (reported by JWEL (2002) suggested that average total PAH concentrations increased. Again, because the depth intervals sampled differed in the two studies (0-5 cm for JWEL and 0-10 for ECCC), however, that apparent trend may partly or fully reflect differences in study design, rather than truly declining conditions. The average total PAH concentration in surface sediment in 2009/2010 was 14.7 mg/kg, approximately 94% higher than the average total PAH concentration in 1999 (7.59 mg/kg), when only the upper 5 cm were sampled. As such, this comparison may be interpreted as evidence that the more highly contaminated sediments lie below 5 cm. Sampling and analyses conducted in 2018, as discussed below, helps elucidate trends with depth and trends over time.

As shown in Figures 10a-d, concentrations of several metals in the AOC exceed the SEL. For samples collected in 2009 and 2010, concentrations of iron exceeded the SEL at most locations in EBMP and at approximately one-third of sampled locations in the Lake George Channel (MMM Group 2015).

Concentrations of metals in 2009/2010 were generally higher in samples collected from EBMP compared to the Lake George Channel.

PAH data collected in 2018 from EBMP are summarized in Table 1, excerpted from Milani and Grapentine (2019a). Based on this most recent round of sampling, EBMP sediment has concentrations of PAHs that are higher than those measured in sediment from local (upstream) and regional reference sites. Milani and Grapentine (2019a) reported:

> "The sum of 34 PAHs (18 parent + 16 alkylated PAHs) was ≤ 37 mg/kg in EBMP and was mostly ≤ 1 mg/kg at reference sites. Parent PAHs (total) exceeded the provincial LEL of 4 µg/g[3] at all EBMP sites. Petroleum hydrocarbons (sum of F2-F4 fractions)[4] were also elevated in EBMP compared to reference locations, with an average concentration of 3,759 mg/kg, compared to ≤ 440 mg/kg at reference sites. Metal concentrations in EBMP sediment were higher than those from reference sites and generally fell between the LEL and SEL. Exceedance of the PEL or SEL occurred mainly for iron, with few other marginal exceedances (e.g., Pb, Zn). Comparatively, the 0-5 cm sediment layer had lower concentrations than the 0-10 cm layer, except for a single EBMP site (EC35). Over time, contaminant concentrations in the 0-10 cm have mostly remained stable or have decreased."

Within EBMP, surface sediment samples have been collected from the same locations for multiple sampling events. Figures 11a-e show changes in concentrations of metals in surface sediment (0-10 cm) from cores collected within EBMP from 2006 through 2018. Like the rest of the AOC, no clear declining trend in average concentrations of metals and PAHs in surface sediment is apparent in EBMP. However, as noted previously, there is a trend of declining concentrations of metals and PAHs in the top surface interval (i.e., 0-2.5 cm) compared with concentrations from 5-10 cm, indicating that the most recently deposited sediments in EBMP tend to have lower concentrations than those deposited previously.

Sediment cores collected from BMP, EBMP, and LGC in 2007 and from EBMP in 2018 show variability in concentrations of metals, PAHs, and petroleum hydrocarbons at depth. In 2007, the maximum concentration of total petroleum hydrocarbons (C6-C50) was found in subsurface sediment (>10 cm) in six out of 10 cores collected from BMP and EBMP (ECCC, unpublished data). Similarly, the maximum concentration of numerous metals, including, but not limited to, iron, cadmium, chromium, lead, and zinc, was found in subsurface sediment in cores collected in 2010 and 2018 from EBMP. These results indicate that at depths below the biologically active zone (i.e., below 10 cm), contaminant concentrations are elevated compared to concentrations in surface sediment.

As discussed above, substantial progress has been made in controlling sources of contaminants to SMR. This finding is consistent with observed recovery of the benthic invertebrate community, as discussed in Section 5.1. The results of ongoing and recently completed investigations and actions

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[3] µg/g is equivalent to mg/kg

[4] No guideline has been established for petroleum hydrocarbons (sum of F2-F4 fractions)

offer additional clarity as to whether contaminant sources are sufficiently controlled to permit effective sediment management.

3. CONTAMINANT MIGRATION PATHWAYS/FATE AND TRANSPORT

The current distribution of sediment contaminants in the AOC is a function of historical releases and environmental fate and transport processes. Fate and transport processes are integral to natural recovery processes that reduce environmental exposures over time. This section describes current patterns of sediment contamination, available information related to natural recovery trends and processes, and bioaccumulation as a fate process in this system.

3.1 Natural Recovery Processes

Trends in sediment quality over time, as discussed further in Section 5.1, suggest that substantial natural recovery of AOC sediments has occurred since the 1980s. Notably, the extent and severity of sediment toxicity in laboratory tests has decreased, and benthic invertebrate community structure has improved. The observed natural recovery likely reflects significant source reductions (Section 2), biodegradation, and burial of contaminants.

Building a site-specific understanding of natural recovery processes is worthwhile, whether to evaluate the feasibility of monitored natural recovery or when contemplating constructed remedies. Natural processes are always ongoing, and the overall success of remedial action may be enhanced by combining natural recovery processes with other engineering approaches (Magar et al. 2009). Figure 12 illustrates contaminant migration, fate, and transport processes potentially affecting contaminated sediments in the AOC. In general, fate and transport processes that contribute to natural recovery include chemical transformation, reduction in bioavailability/mobility, physical isolation, and dispersion (Magar et al. 2009, Fuchsman et al. 2014). Each of these processes is discussed below with regard to SMR.

- Chemical transformation: Wood waste material, hydrocarbons, and PAHs are all susceptible to biodegradation. Biodegradation is slowed by anoxic conditions in buried sediments but may be a significant recovery process in biologically active surface sediments.
- Reduction in contaminant mobility or bioavailability: Sequestration of contaminants via sorption or precipitation achieves risk reduction to the extent that it minimizes the potential for human or biological exposure. Sorption to black carbon (e.g., soot) can increase sequestration of hydrophobic organic compounds such as PAHs (Cornelissen et al. 2005). Analyses of PAHs in sediment porewater, as commissioned by ECCC and as conducted by Hawthorne (2010), provide greater insight into chemical bioavailability in SMR sediments. Hawthorne (2010) reported PAHs to be generally below the detection limit in pore water.
- Physical isolation: Natural sediment deposition segregates contaminated sediments from organisms through: 1) burial beneath cleaner surface sediment, 2) dilution of contaminated surface sediment by mixing with cleaner sediment, 3) consolidation and cohesion of the sediment bed, and/or 4) bed armouring processes (i.e., coarsening of the surficial sediment layer relative to the grain size distribution of underlying sediment). The resulting physical isolation achieves risk reduction by reducing chemical exposures in surface sediment (where biological receptors contact sediment) and by reducing the potential for resuspension and transport of contaminated sediments. The available sediment core data, though limited, suggests that chemical concentrations tend to be higher in subsurface sediment than in surface sediment (Kilgour et al.

2001, Burniston and Kraft 2007). Sediment core data collected from EBMP in 2018 indicate that concentrations of metals and PAHs are lower in the top 0-5 cm, compared with concentrations from 5-10 cm (ECCC, unpublished data), and the highest concentrations of contaminants in each core are generally found below 10 cm. These results indicate that cleaner sediment may be depositing over more contaminated sediments and physically isolating the higher concentrations of contaminants.

- Dispersion: Chemical dispersion, through resuspension and transport of contaminated sediments downstream or dissolution and transport of dissolved contaminants, achieves net risk reduction to the extent that dispersion processes reduce biological exposures in the original contaminated area without resulting in unacceptable risks elsewhere. Because dispersion does not in itself eliminate contaminant exposures, it is not generally considered a recovery process. However, understanding dispersion processes is important for predicting future sediment stability. In depositional areas of SMR, subsurface gas generation (due to degradation of buried wood-derived waste material) and subsequent eruption of gas bubbles may provide a dispersion mechanism by which contaminants move from buried sediments to the surface. This mechanism is of concern if free oil is present. The extent and importance of gas generation in depositional areas is not known. At a broader scale, the spatial extent and magnitude of sediment quality impairment has decreased over time, implying that downstream transport is not the most important recovery process in this system. Specifically, sediment monitoring results for oil and grease in Little Lake George and Lake George – at the receiving end of downstream transport processes – indicate substantial recovery.

In addition to identifying whether natural recovery processes are occurring, understanding the rate of natural recovery can be useful for predicting the future progress of natural recovery. Krishnappan (2013) conducted a modelling study to calculate the rate of fine sediment deposition in the embayment area east of Topsail Island near Bellevue Marine Park. Krishnappan incorporated the following parameters into Krone's equation to model fine sediment deposition: 1) knowledge of the flow field within the river, 2) concentrations of suspended sediment, 3) settling velocity of fine sediment, 4) sediment bulk density, and 5) critical sheer stress for erosion and deposition of fine sediment. Other than information regarding the flow field, which was computed using a hydrodynamic flow model (RMA2), all other parameters were measured in the field. Results of the RMA2 flow model indicated a large circulation eddy located east of Topsail Island, which likely traps fine sediment and enhances fine sediment deposition (Ingribelli and Naccarato 2013). Krishnappan calculated total deposition for the year from May 2012 until April 2013 as 2,572 metric tonnes, with monthly deposition amounts ranging from 165,000 kilograms (kg) to 322,000 kg (Krishnappan 2013). The total thickness of sediment deposited within the eddy East of Topsail Island during the year was estimated to be 20.7 millimeters.

More quantitative retrospective methods, such as radioisotope dating of sediment cores, are not practical in some of the areas of interest (e.g., Bellevue Marine Park), due to the presence of wood chips in the sediment and the high variability of historical deposition rates associated with bulk waste material discharged from St. Marys Paper. Sediment trap data were collected in the Bellevue Marine Park area in the 1980s and in 2005 (Boyd 2011; unpublished data). These data have some limitations in terms of characterizing sediment deposition rates in this environment. Specifically, sediment traps can potentially create a micro-environment that enhances particle deposition, thereby overestimating the deposition rate (Fuchsman et al. 2014). When bias in deposition rate is suspected, sediment trap data are often better suited to characterizing the chemical quality of newly deposited sediment.

Conducting high resolution bathymetric surveys within the areas of interest over a span of five to ten years, possibly in combination with sediment profile imagery (SPI) and use of a tracer, may help quantify the rate of sediment deposition over time. Bathymetric surveys would need to be sufficiently high resolution to support an estimation of sediment thickness changes, and therefore deposition and erosion rates, within targeted areas. If a tracer material were deposited in a thin layer during an initial SPI monitoring event, subsequent SPI monitoring could provide precise measurement of the thickness of sediment deposited after application of the tracer. The need for alternative data to estimate rates of sediment deposition and/or other recovery processes collection – whether through retrospective analyses or prospective monitoring – may warrant further consideration after decisions have been reached regarding the need for sediment management.

3.2 Contaminant Migration to Depositional Areas

Current patterns of sediment contamination in SMR reflect sediment transport from high-energy areas near point sources to depositional areas farther downstream. Due to the river's morphology, deposition largely occurs on the Ontario side of the river. Because many contaminants (e.g., PAHs, petroleum hydrocarbons, metals) tend to bind to solids in the water column, depositional areas where sediment accumulates also tend to be areas where contaminants accumulate. From upstream to downstream, important depositional areas for contaminated sediments are identified as follows (Figure 2):

- Bellevue Marine Park
- EBMP
- Lake George Channel
- Little Lake George
- Lake George

In contrast with the high-energy conditions near the major point sources discussed in Section 2, sediment contamination is primarily found in depositional areas, which have consequently been the focus of sediment quality investigations in recent years.

3.3 Bioaccumulation

The tendency of different contaminants to bioaccumulate depends on the chemical and physical properties of the various contaminants. Biomagnification and food web transfer are not major fate processes for the key sediment contaminants in the AOC, namely PAHs, petroleum hydrocarbons, and wood-derived material. PAHs are rapidly metabolized by fish and thus do not occur at significant concentrations in fish tissue[5].

Metals such as iron, zinc, and manganese are not expected to pose any adverse effects for organisms higher up in the food chain, particularly since many trace metals are essential for normal growth and function. This is not to say that such metals present no concerns related to bioaccumulation. If localized pockets of higher concentrations of bioavailable metals in the sediment are present, they

[5] It is worth noting, however, that PAHs are associated with photo-induced toxicity in fish, as well as fin erosion (Sved et al. 1997, Woodward et al. 1983) and lesions and tumors (Hawkins et al. 1988, Hawkins et al.1990, Baumann 1998).

could lead to bioaccumulation of the metal in some sediment-dwelling organisms, with the possibility of adverse affects on those organisms and/or their primary consumers. The significance of bioaccumulation-related exposure pathways is discussed in Sections 5.3 and 5.4.

4. SEDIMENT STABILITY

Sediment stability is very important to the viability of managing contaminated sediments through either natural or engineered physical isolation processes. The available sediment core data, though limited, suggest that chemical concentrations tend to be higher in subsurface sediment than in surface sediment (Kilgour et al. 2001, Burniston and Kraft 2007). Sediment cores collected from EBMP in 2018 show higher concentrations of contaminants at depths of 5-10 cm than at depths of 0-5 cm. This apparent vertical stratification supports the hypothesis that sediments have been stable in the past and that sediments are likely to continue to be stable in the future unless there is a significant change to the system (e.g., an extreme high-energy event or a major hydrologic alteration).

Sediment transport modelling provides a useful tool to predict future sediment stability, for instance in the case of reasonably plausible high-energy events. Krishnappan (2011) assessed the stability of sediments in the AOC based on several lines of evidence, including development of a sediment transport model for SMR, as well as sediment grain size profiles and sediment videography. The model used information on sediment erodibility (critical shear strength) with information on water flow conditions and associated bed shear stress to predict whether and under what conditions sediments in SMR will erode. Sediment erodibility was estimated based on video imaging work, sediment grab sample analysis, and in-situ erosion flume experiments. Krishnappan (2011) used a two-dimensional hydrodynamic flow model (RMA2) and a fine sediment transport model (RMA4) to predict the movement of water flow and the subsequent flow-induced fate and transport of sediment particles. The modelling results indicate that, under a range of flow conditions including ice cover, contaminated sediment deposits along the edges of the river are stable at sediment depths greater than approximately 5 cm (Krishnappan 2011). The current inventory of buried wood chips in the Bellevue Marine Park area confirms that there is limited transport of even light-weight material from this area.

Krishnappan (2012) conducted an additional sensitivity analysis to further explore the predicted amount of erosion versus the magnitude and frequency of high flow events. Krishnappan (2011) evaluated a range of flow conditions spanning the highest and lowest monthly average flows recorded since 1900. The greatest erosion potential typically occurs over time intervals of hours to days, rather than over a full month. In SMR, the very large size of the watershed (i.e., the Lake Superior watershed) is expected to mitigate the effects of severe storms on flow conditions, although flow at storm water discharge points could be significant on a localized basis. Water releases from the Compensating Works[6] may constitute the highest flow events in the river under current operating practices. To investigate the importance of short-term flow events, Krishnappan (2012) examined available daily flow data and considered how operations at the Edison Sault Hydro Electric Power plan would affect monthly versus daily flows, given its proximity to EBMP and the back eddy. This supplemental analysis did not change the conclusion of Krishnappan (2011), namely that sediment layers below a depth of 5 cm are stable in SMR. Krishnappan (2012) also estimated the critical flow rate that would cause sediment erosion at sediment depths greater than 5 cm. This information may

[6] The Compensating Works consist of 16 gates located at the mouth of the rapids in St. Marys River, and are used to control the outflow of water from Lake Superior.

aid future refinements of the CSM with regard to sediment stability, for instance if hourly flow data were available.

The sediment stability investigations described above have focused on Bellevue Marine Park. In recent years, risks to benthic invertebrates appear more pronounced in EBMP and the Lake George Channel, as compared to those in Bellevue Marine Park. Consequently, sediment stability in EBMP and Lake George Channel also warrants consideration. M.R. Wright and Associates Co., Ltd. (MRW) conducted a geotechnical investigation of the sediment in EBMP in December of 2011 (MRW 2012), intended to complement the flow model developed by Krishnappan (2011) particularly with respect to variability in sediment composition with depth and potential impacts of sustained near-field flow stresses. MRW retrieved depth-stratified sediment samples using Cone Penetration Test soundings in conjunction with an ultraviolet induced fluorescence module and sediment cores. MRW reported a layer of soft organic silt material, ranging in thickness from 2.3 to 5.3 metres (m), underlain by a denser nonorganic sediment layer. The nonorganic stratum was found to range from 0 m to approximately 8 m in thickness and was underlain by glacial till deposits.

The RMA model developed by Krishnappan provides a good basis for identifying erosional and depositional areas of the river under different flow conditions. We also suggest that the hydrodynamic component of the model (i.e., RMA2) be validated by measuring flow velocities during a different time of year than the measurements collected for model calibration, if feasible during the next phase of study. This model component was developed from an existing coarse-grid model that has been validated for SMR, but the fine-grid model (i.e., modelling of eddies) has not been further validated. We also suggest collecting additional sediment grab samples in areas in the river that have not been sampled,[7] if that would help improve model accuracy. Some of the sediment model (RMA4) parameters may need to be fine-tuned following validation sampling.

As discussed in Section 3.2, Krishnappan (2013) used the RMA model, in conjunction with measured field parameters, to estimate fine sediment deposition rates in the embayment area east of Topsail Island. While the study calculated that fine sediment deposition was occurring within this area, data will need to be collected over a longer period to assess the stability of the depositing sediment in this area (Ingribelli and Naccarato 2013).

Krishnappan's (2011) sediment transport model does not directly account for potential sediment disturbance due to ice scour or vessel traffic. However, MECP, ECCC, and Sault Ste. Marie Innovation Center (SSMIC) staff familiar with the area have observed no evidence of ice scour in Bellevue Marine Park or in EBMP. In these areas, ice cover persists until the spring melt, at which time the low-energy flow conditions allow the melting ice to remain in place or move off-shore when winds are favourable. Photographs of ice melt conditions in the EBMP area are shown in Figure 13. Additionally, SSMIC staff researched historical documentation of ice conditions in SMR and found no documentation of ice jams or ice-related sediment disruption outside the shipping channel and the ferry route from Sugar Island to mainland Michigan (Antunes 2011). Indeed, as reviewed by Bolsenga (1992), "ice seemed to

[7] For example, Transects 7 and 8 (see Krishnappan 2011).

protect rather than to damage the river banks" of SMR. Further characterization of ice scour does not appear warranted at this time.

Several factors limit the significance of vessel traffic with regard to sediment stability, at least in the Bellevue Marine Park area (Biberhofer 2011). Large vessel traffic is limited to the navigation channel and has speed limits managed by the seaway control. An island and ridge complex protect the central portion of Bellevue Marine Park from ship wake. Additionally, the shallow, soft sediments in this area are further protected by substantial macrophyte coverage. Review of the spring 2018 sediment core sampling results could aid determination of specific portions of EBMP and the Lake George Channel, if any, where the stability of buried contaminants merits further evaluation. The need for any additional evaluation of small vessel traffic-related effects on sediment stability should be determined after this information is considered.

Krishnappan's (2011) model accounts for changes in Great Lakes water levels, within the range of conditions included in the historical record. Changes in lake levels beyond historical conditions, such as might possibly occur due to climate change, were not evaluated. However, due to the relatively small change in elevation (low gradient) within SMR, it is expected that changes in lake levels would have to be extreme to affect sediment stability. The issue of climate-related water level changes is evaluated as part of the International Joint Commission's (IJC's) Upper Great Lakes Water Level Regulation Study (IJC 2018), which considers adaptive management strategies. Further evaluation of changing water levels, beyond that already conducted by IJC, does not appear warranted to support sediment management strategy development for the AOC.

In addition to the above considerations, the evaluation of sediment stability in SMR should consider whether any major physical changes to the system are expected, such as changes in navigational dredging patterns or large-scale habitat restoration measures. A critical question relates to whether areas that are not currently dredged will need to be dredged in the future. Contaminants could be exposed if areas with buried contaminated sediments were dredged to support navigation. However, the National Oceanographic and Atmospheric Administration's Nautical Chart 14884 indicates that the navigational channel is located well south of the contaminated portions of Bellevue Marine Park and EBMP, suggesting that neither area is likely to be targeted for navigational dredging in the future. Furthermore, the physical characteristics of Bellevue Marine Park are not conducive to navigational dredging. Small-scale dredging, for instance to maintain the marina in Bellevue Marine Park, might be undertaken in the future. This matter can be considered further as part of future consideration of sediment management options. Additionally, if modifications of the navigation channel to accommodate a deep-water port are planned in the future, the ramifications for sediment stability in nearshore depositional areas should be evaluated through supplemental sediment transport modelling.

5. SEDIMENT-RELATED RISKS

The objective of this section is to describe current understanding of human health and ecological risks posed by exposure to contaminants in SMR sediments. Risks to benthic invertebrates, fish, wildlife, and human receptors are summarized in this discussion. Understanding adverse biological effects of contaminated sediments is central to developing a contaminated sediment management strategy, because the goal of sediment management is to reduce human health and ecological risks to acceptable levels. Overall, benthic invertebrates appear to be risk drivers for contaminated sediments in the AOC, though benthic impacts are both localized and improving. As discussed in this section, the incidence in tumours in fish (6%) marginally exceeds the target threshold of 5%. As such, risks to fish are considered marginal. Because fish are exposed to contaminants in the water column and are mobile, compared to risks posed to benthic invertebrates, risks posed to fish are less directly related to sediment contaminants. Wildlife and humans that consume fish or contact sediment also may be exposed to contaminants. As discussed in this section, researchers have found little evidence of impairment to colonial waterbird populations within the AOC. Although a formal human health risk assessment has not been conducted, human exposures to contaminants in sediment are expected to be minimal, due to the low bioaccumulation potential of the primary contaminants. As detailed in Section 5.4, human health concerns are not a driving factor in developing a sediment management strategy.

5.1 Invertebrates

Over the past four decades, numerous studies have been conducted in the AOC to investigate effects of contaminated sediments on invertebrates. These studies involved sediment chemistry, laboratory toxicity tests, and/or benthic invertebrate community sampling. Early studies documented the absence of the mayfly *Hexagenia* from SMR sediments in association with oil contamination (Hiltunen and Schloesser 1983, Schloesser et al. 1991). Studies conducted in the early to mid-1980s were not reviewed for this report but are summarized by others (Kilgour et al. 2001, JWEL 2002). Studies conducted in the 1990s included investigations near the Algoma facility (Pope 1990, Kauss 2000, Pope and Kauss 1995), toxicity testing on Bellevue Marine Park sediments (Murphy et al. 1995); sampling between the Algoma Boat Slip and Lake George in 1992 (Bedard and Petro 1997, Arthur and Kauss 2000), further investigation of Bellevue Marine Park in 1995 (Bedard and Petro 1997, Kilgour et al. 2001), and investigation in the vicinity of the Algoma facility and Lake George Channel in 1999 (MECP 2002, JWEL 2002). This section focuses primarily on the last two decades of investigation. Following a chronological summary of the work conducted to date, this discussion evaluates the most likely causes of observed toxicity, reasons why sediment toxicity continues to be observed even though benthic community alterations are no longer apparent, and the overall significance of recent sediment toxicity and invertebrate community findings.

5.1.1 Temporal Trends

As described in this section, temporal trends through 2018 generally indicate reduced sediment toxicity and improved benthic invertebrate community quality over time. For example, Bellevue Marine Park had supported essentially no invertebrates in the early 1980s, whereas by 2002 the resident invertebrate community was either equivalent to or only possibly different from reference conditions. Similarly, the Algoma Boat Slip was largely devoid of organisms in 1992, whereas the 2002 investigation found the invertebrate community there to be equivalent to reference conditions. Throughout the areas studied during the past decade, only a small number of stations (in EBMP and Lake George Channel) have been classified as different or very different than reference conditions, although a substantial number of stations were identified as possibly different. Similarity to reference sites has been increasing for most EBMP sites (Milani and Grapentine 2019b). Similarly, major toxicity (as indicated by results that differ from the reference mean by more than three standard deviations) was rarely observed in the 2018 ECCC laboratory tests, although less severe effects were observed more frequently. In 2005, MECP examined sediment chemistry, toxicity, and benthic community composition within Lake George and Little Lake George as part of a recovery study (George 2006, MECP 2010). The main goal of this study was to compare current conditions in Little Lake George and Lake George with historical data to determine the extent of recovery of the system. It concluded that the Lake George system has improved since 1992 and that, "while some parameters exceeded [PWQOs] and Sediment Quality Guidelines, those exceedances were slight and generally not cause for concern" (George 2006).

Beginning in 2002 and continuing through 2018, ECCC applied the BEnthic Assessment of SedimenT (BEAST) methodology, initially at 31 locations from Izaak Walton Bay to Little Lake George (Milani and Grapentine 2006). Subsequent investigations provided increasingly targeted evaluation of potentially impacted areas, ultimately focusing on EBMP and the Lake George Channel (Milani and Grapentine 2009, 2010a, 2010b, Milani 2011). Milani and Grapentine (2018) evaluated BEAST results from 2008-2010 using a reference site data collected from 2008 to 2010 to allow for comparisons that were representative of conditions of the time of sampling of the EBMP sites. In the original assessment of the 2008 to 2010 data, old reference site data had been used. Milani and Grapentine (2018) concluded that, as of 2010, benthic communities at 64% of the EBMP sites were similar to those from reference sites, while 36% were possibly different. As of 2010, the COA Framework concluded that management action was necessary at four sampling locations.

In 2016, ECCC collected sediment from ten sites within the AOC and conducted toxicity studies in fish,[8] mussels, and aquatic invertebrates (Parrott et al. 2018). The ten sites, plus one reference site, were identified as exhibiting toxicity to exposed invertebrates during previous toxicity testing conducted between 2002-2010 (Milani 2012, Milani and Grapentine 2006, 2009, 2010a). Parrott et al. (2018) evaluated exposures to fathead minnows (Section 5.2), amphipods (*Hyalella azteca*), and juvenile mussels (*Lampsilis siliguoidea*). To evaluate amphipod risks, three endpoints (survival, growth, and reproduction) were assessed using sediment collected from six sites. Survival was greater than 91% at all sites except at the Algoma Boat Slip site, where survival was significantly lower (18%). Amphipod growth at the Algoma Boat Slip was significantly lower than at other sites tested. Results of toxicity tests on juvenile fatmucket mussels showed that mussel survival and burial after 28 days was not significantly lower in sediment collected from any of the St. Marys River sites compared to the upstream reference site. A non-significant decrease in survival was observed between mussels exposed to sediment collected from two sites in the AOC (Algoma Boat Slip 182 and EBMP-EC31) compared to the local reference site.

[8] See Section 5.2 for discussion of methods and results from components of this study related to fish.

In 2018, ECCC collected sediment from eight sites in EBMP and subjected them to a series of chemical analyses, benthic community structure analysis, and further toxicity testing that included invertebrates and fish (Section 5.2) (Milani and Grapentine 2019a,b, Bartlett et al. 2019). Results were then interpreted according to the COA Framework to determine the environmental risk posed by contaminated sediment (Milani and Grapentine 2019c). The eight sampling locations had been chosen based on results of previous benthic assessments that had indicated sediment management action was required (ECCC 2018). Samples also were collected from four upstream locations to serve as local reference conditions, and data from comparable Great Lakes reference stations were also considered. To determine whether natural recovery was occurring, additional sediment was collected at a subset of sites from the 0-5 cm layer, tested, and compared to results for the standard 0-10 cm assessment layer. Based on application of the COA Framework to the 2018 EBMP sediment data, management action was not warranted at any of the eight sampling locations, because benthic community composition was similar to reference conditions. At two of the sampling locations (EC26 and EC52), application of the COA Framework concluded that no further action was warranted (Table 2). At the remaining six locations, the COA Framework indicated that the reasons for sediment toxicity should be determined. Appendix A provides a detailed evaluation of the potential causes of toxicity in the 2018 EBMP sediments, considered in the context of historical site-specific investigations and other relevant case studies. The following analysis discusses the most likely reasons for sediment toxicity in St. Marys River sediments over time.

5.1.2 Causes of Toxicity

Four decades of investigations provide substantial evidence to assess the causes of toxicity to benthic invertebrates in St. Marys River sediments. In general, TPH and/or PAHs have been identified as the most probable causes of toxicity. Some evidence has also suggested potential toxicity related to metals, but as discussed further in Appendix A, metal toxicity in laboratory tests may not have been representative of conditions in the field. Historically, hydrogen sulphide was also suggested as a potential toxicant in this system, but little evidence has been gathered to support this hypothesis.

Lines of evidence related to site-specific effects of TPH and PAHs on benthic invertebrates include the following:

- Midge and mayfly growth reductions were correlated with sediment PAH concentrations in the MECP 1992 data set (Algoma Boat Slip to Lake George), and with sediment TPH concentrations in the MECP 1995 data set (Bellevue Marine Park) (Bedard and Petro 1997). Kilgour et al. (2001) also found a relatively strong association between TPH and benthic invertebrate community composition for the 1995 data set.
- JWEL (2002) pointed to TPH as the principal contaminant associated with effects on benthic invertebrates in the MECP 1999 data set, largely due to high TPH concentrations and poor benthic community quality in the Algoma Boat Slip. The observed effects also were associated with several other covarying constituents (PAHs, metals, cyanide, oil and grease, organic carbon). This finding is consistent with early reports (1960s to 1980s) linking the presence of visible oil with absence of the mayfly *Hexagenia* from SMR sediments (Hiltunen and Schloesser 1983, Schloesser et al. 1991).
- Golder (2004) agreed that TPH and/or PAHs were the likely cause of benthic impairment in the Algoma Boat Slip and Bellevue Marine Park. Physical conditions such as sediment grain size were

considered the dominant factors affecting benthic invertebrates in the Algoma Slag Dump area, Little Lake George, and Lake George. Organic enrichment was identified as a likely cause of benthic impairment in the Lake George Channel (Golder 2004).

- The results of MECP's 2005 recovery study (MECP 2010) showed impaired growth of both *Hyalella azteca* and *Chironomus tentans* in Upper Lake George sediments. The impaired survival of *C. tentans* was significantly correlated to the concentration of oil and grease in the sediment.
- In ECCC's 2002 data set, a combination of chemical and physical characteristics of the sediment was required to explain toxicity test results, with TPH identified as the chemical measure most clearly associated with toxicity. In subsequent ECCC investigations, however, TPH concentrations were lower, and efforts to identify relationships between toxicity and chemistry were less conclusive than the earlier results. Zinc may have contributed to observed toxicity in the 2006 tests (Milani and Grapentine 2009).
- In 2006 laboratory tests, concentrations of PAHs in invertebrate tissue were not correlated with toxicity test results (Milani and Grapentine 2009). Tissue concentrations were expected to provide a better relationship than sediment concentrations to biological effects, because they reflect bioavailable exposures. These results thus suggested that other factors contributed significantly to the observed biological effects in the sediments studied.
- In collaboration with ECCC, U.S. Environmental Protection Agency (USEPA) performed a toxicity identification evaluation (TIE) for two SMR sediment samples, to investigate potential causes of observed toxicity (Milani et al. 2008). The investigation used whole-sediment TIE techniques, which involve applying treatments to the sediment to remove toxicity due to selected classes of chemicals. None of the treatments substantially reduced toxicity to amphipods or midges, indicating that candidate toxicants such as PAHs, cationic metals, and ammonia were not primarily responsible for the observed effects. Rather, the test results were consistent with emerging research indicating that physical effects of oil may dominate over PAH-related effects in some oil-contaminated sediments (Mount et al. 2009). Relationships between oil and grease (solvent extractable) exposure and midge biomass for the two SMR sediment samples are consistent with those observed at other sites outside the SMR AOC where physical toxicity of oil is suspected. This relationship is clearer when the oil and grease concentrations are expressed on a volume-normalized basis, rather than the usual dry-weight basis (Figure 14).
- Parrott et al.'s (2018) toxicity studies, discussed above and in Section 5.2, found elevated toxicity at the Algoma Boat Slip site, where the concentration of total PAHs in sediment was 30-fold higher than the other sediment sites sampled within the AOC. Parrott et al. (2018) concluded that elevated concentrations of PAHs was most likely the cause of reduced survival and growth of amphipods and fathead minnow within the Boat Slip.
- As discussed further in Appendix A, 2018 sediment toxicity test results for EBMP and upstream reference stations (Milani et al. 2019a) show that toxicity to mayflies and worms was associated with elevated TPH exposures, although the observed toxicity was not manifested in significant benthic community alterations. Toxicity test results for amphipods and midges were not related to TPH but may have been confounded by testing artifacts and/or unmeasured chemicals unrelated to EBMP sediment contamination.

Taken together, these collective results support identification of TPH and PAHs as the predominant stressors in the system, although the signals of particular toxicants have become less pronounced as sediments have recovered over time.

As discussed in Section 2.2, several metals are present within the AOC at concentrations greater than the SEL. Golder (2004), however, hypothesized that the bioavailability of most metals is probably low in the organic-rich, fine-grained sediments of SMR's depositional areas. Also, metal concentrations in AOC sediment are not greatly elevated (Milani and Grapentine 2010a, Keller et al. 2011). However, different metals have different affinities for organic carbon, and the low-oxygen conditions typical of organically enriched sediments also affect metal speciation, which can promote bioavailability of some metals (e.g., iron and manganese). Milani and Grapentine (2009) evaluated metal bioavailability by analyzing of metals in toxicity test organisms (*C. riparius*) and the sediments and overlying water to which they were exposed. This analysis indicated a correlation between bioavailable concentrations of zinc and toxicity test outcomes, among the samples tested. However, cause-effect relationships between metal exposure and toxicity are not well understood for *Chironomus* species; the amphipod *Hyalella azteca* has been more thoroughly studied in this regard (Borgmann et al. 2004). Similarly, Parrott et al. (2018) measured metals in overlying water from sediment toxicity test chambers and found some evidence of bioavailable metal exposures. However, as discussed in Appendix A, recent research indicates that artifacts related to sediment handling can cause metal bioavailability to be artificially high in toxicity tests as compared to sediments in the field.

Murphy et al. (1995) hypothesized that hydrogen sulfide could be an important cause of toxicity in Bellevue Marine Park sediments. They injected several tonnes of iron (ferric chloride) into the sediments *in situ*, with the objective of rendering sulfide unavailable to benthic organisms. Although a bacterial bioassay showed some positive response to this treatment, more ecologically relevant toxicity tests with the mayfly (*Hexagenia* sp.) and the water flea (*Daphnia magna*), demonstrated no toxicity in untreated or treated sediments (despite observations of poor infaunal invertebrate abundance). Overall, this study was inconclusive with regard to contribution of hydrogen sulfide to benthic impairment. However, it is reasonable to expect that stressors associated with organic enrichment (such as hydrogen sulfide or oxygen depletion) could have been significant in some SMR sediments.

5.1.3 Relationship Between Toxicity and Benthic Community Condition

As discussed in Appendix A, several factors likely contribute to the observation that EBMP benthic invertebrate community composition is consistently similar to reference conditions, even though paired sediment toxicity tests indicate toxicity (Milani et al. 2019a,b,c). First, observed effects on amphipods and midges in the toxicity tests did not appear to be due to EBMP-related sediment contaminants and occurred in sediment from references stations as well as the EBMP area. Sediment handling and aeration procedures during toxicity testing may have artificially increased sediment toxicity to metal-sensitive species. Also, interactions between toxicity test organisms and native organisms in the sediment samples may have created the appearance of toxicity to both test species, when effects were actually caused by competition or predation. Further, *Hyalella azteca* is especially sensitive to current-use pesticides, which were not measured in St. Marys River sediments but could potentially have been present. In the case of such pesticides, *Hyalella* may be anomalously sensitive (REF), in which case the *Hyalella* toxicity tests would be more sensitive than the overall benthic community.

Laboratory toxicity to mayflies and worms was associated with hydrocarbon exposure and thus seems more likely to be representative of effects potentially occurring within the EBMP area. Mayflies were a major component of the benthic invertebrate community at all upstream St. Marys River reference stations in 2018, but they were less abundant or absent at most of the Great Lakes reference stations considered by Milani et al. (2019b). In the EBMP area, mayfly abundance was comparable to upstream St. Marys River conditions at some stations and Great Lakes reference conditions at other stations. It is uncertain whether these results represent a toxic effect or merely natural variability in mayfly abundance. Possible reasons why adverse reproductive effects in *Tubifex* worms might not be manifested in the benthic invertebrate community include: prevalence of other less sensitive worm species, local adaptation to tolerate hydrocarbon exposure, and/or density-dependent interactions that limit the extent to which reduced offspring production affects population abundance.

5.1.4 Significance of Effects

Application of the COA Framework by location for the 2018 ECCC study provides a means of delineating benthic impairment. Milani and Grapentine (2019c) conclude that:

> "sediment chemistry and benthic community structure studies appear to indicate that sediment from the SMR are being replaced with cleaner sediment over time, and that the area appears to be recovering naturally (except for chemistry at location EC35 of EBMP). Invertebrate toxicity showed variable responses depending on the endpoint and improvements were difficult to discern due to high variability, but some endpoints did show better survival, growth and reproduction in the 0-5 cm vs. the 0-10 cm, whereas the opposite was not observed for the most part. The COA Framework outcome for six of the eight EBMP sites was to investigate the reason(s) for toxicity and no further action for remaining two sites. The assessment outcomes were the same or improved from previous studies (2008-2010) conducted in EBMP. The overall toxicity line of evidence contradicted the results of the benthic community structure line of evidence for EBMP sites, which was indicated as similar to reference. However, toxicity to *Hyalella* in the 28-day exposures may have been confounded by low overlying water pH observed at the end of the tests for some EBMP sites. Additional toxicity tests (i.e., longer exposures to *Hyalella*, fish exposures) showed similar responses between EBMP sites and local (upstream) SMR reference sites but *Hyalella* survival, growth, and reproduction were lower in EBMP when compared to regional (Great Lakes) reference sites."

Based on the COA Framework results, management action is not needed to address adverse effects on the benthic invertebrate community. Although the possibility of moderate to subtle effects related to petroleum hydrocarbons at some EBMP locations cannot be fully ruled out, benthic invertebrate community condition in the EBMP area continues to recover and is now similar to reference conditions. The causes of sediment toxicity (discussed in Appendix A) are sufficiently well understood to confirm that the benthic community findings can be relied upon for sediment management decision-making. However, continued monitoring is warranted to confirm ongoing natural recovery, and to the extent that monitoring continues to include toxicity testing, certain modifications to the test methods should be considered (see Appendix A). Additionally, measures to safeguard against accidental disturbance and exposure of buried contaminants may be warranted (i.e., institutional controls).

5.2 Fish

Three lines of evidence provide insight regarding the likelihood and severity of adverse effects (if any) in SMR fish: liver tumour survey data, larval fathead minnow toxicity tests, and long-term fish population surveys. Together, these three lines of evidence indicate that fish populations have been stable over several decades, tumour incidence is low, and sediment toxicity is low at all but one sampling location. Consequently, no further action is warranted to mitigate risks posed by SMR sediment to fish.

5.2.1 Liver Tumour Surveys

As reported by ECCC et al. (2002), white suckers sampled from SMR (1985-1990) exhibited liver tumour prevalence in excess of 9% (n=185), and liver cancers also were identified in brown bullheads from Munuscong Bay. Elsewhere, liver tumours in bottom-dwelling fish have been shown to be related to PAHs in sediment (Baumann and Harshbarger 1998, Johnson et al. 2002). At least in brown bullhead, populations affected by liver tumours also tend to exhibit external anomalies, such as skin tumours and lesions (e.g., Black River RAP 2004). Fisheries management planning documents developed for SMR do not mention external anomalies in fish as a concern (Fielder et al. 2002, 2004, Gebhardt et al. 2002), though the absence of discussion does not necessarily demonstrate the absence of anomalies in local fish populations. Recent surveys indicate that the overall health of SMR fish community compares favourably with relatively unimpacted sites from Lake Huron (Pratt and O'Connor 2011).

In late 2009, ECCC collected 141 white suckers from EBMP and near Partridge Point within the Lake George Channel for examination of liver tumours and other health parameters (Chambers and McMaster 2018). Of the 141 white suckers collected from both areas, 15 were found to have liver tumours (10.6%). Three of the 40 fish sampled from the Lake George Channel were found with tumours (7.5%), while 12 of the 101 fish from EBMP were found with tumours (11.9%). The increase in tumour prevalence since the 1985-1990 sampling was surprising, given the remedial cleanup efforts and source control measures implemented within the AOC since the first assessment. Baumann (2013) provided two possible explanations for the higher tumour incidence in 2009: 1) elevated PAH concentrations in sediment are preventing recovery in the fish population or 2) the fish population includes older individuals that may have cancers caused by exposure prior to sediment remediation. Baumann recommended a statistical comparison of age and tumour incidence between the late 1980s studies and the 2009 study. Baumann also concluded that an additional fish tumour survey or sediment survey should be considered.

ECCC conducted an additional fish tumour study in 2015 to assess changes in liver tumour rates. In the 2015 study, 100 white suckers were collected from the same areas within the AOC and 6 were found to have tumours (6%) (Chambers and McMaster 2018). This was a significant decline since the 2009 assessment, and the tumour rate is approaching the RAP's delisting criteria of 5%.

5.2.2 Sediment Toxicity Testing

Laboratory toxicity tests were recently conducted to determine the toxicity of AOC sediment to larval fish. In 2016, Parrott et al. (2018) evaluated fathead minnow embryo-larval exposure assessments using sediments collected from 11 sites in SMR (Algoma Boat Slip, EBMP, and Bellevue Marine Park). The chronic 21-day embryo-larval test showed no impact on larval minnows from sediment collected at 10 of the 11 sites (Parrott et al. 2018). The only exception was sediment collected from the Algoma Boat Slip, which was found to be toxic to larval fathead minnows. The decreased toxicity in surface sediment from EBMP may indicate that cleaner sediments are being deposited within this area, thereby supporting natural recovery.

In 2018, Parrott and Milani (2019) collected sediment from four locations in EBMP to further evaluate toxicity to larval fathead minnows. At each location, samples were collected from both the 0-5 cm depth interval and the 0-10 cm depth interval to determine whether there are differences in toxicity from deeper sediments. As concluded by Parrott and Milani (2019):

> "Fathead minnows showed normal survival and growth after exposure to sediment from SMR sediments, compared to sediment collected from two reference sites on Georgian Bay. There was no difference between sediment collected to a depth of 5 cm compared to sediment collected to a depth of 10 cm. Survival was similar to that observed in laboratory controls; however, all collected sediment (from test and reference sites) reduced growth slightly (reductions of about 25% for wet weight and 9% for length) compared to laboratory controls. The controlled exposures of embryos and larval fish in the laboratory showed that sediment collected from the SMR had no chronic effects on fathead minnows. There were no significant changes in survival, growth, or development of the larval fish compared to the upstream SMR reference sites. There were no differences between SMR sediments and Georgian Bay Reference site sediments. In addition, there were no differences in response at the two depths of sediment collection."

5.2.3 Fish Population Surveys

Schaeffer et al. (2011) examined trends in fish abundance from 1975 through 2006 and in size and age from 1995 through 2006. By comparing 2006 gill net survey data to a creel survey conducted the same year, the authors were able to interpret population trends over a 69-year timeframe for which creel survey data were available. Schaeffer et al. (2011) characterized the SMR fish community as a coolwater fish community with apparent little variation in species composition and only slight variation in fish abundance since 1975. Most individual captured were native (74%) and exotics represented 26% of all fish caught. Nearly all exotic fish species caught had been intentionally introduced (salmonids) or had been established in SMR prior to the first survey (alewife, rainbow smelt, common carp). Yellow perch, white sucker, cisco, and rock bass were consistently the four most abundant species in each gill net survey. In recent years, populations of some target species sought by anglers have increased (e.g., centrarchids), while others have been stable (percids) or declined (northern pike, cisco). On an annual average basis, northern pike and cisco declined 1.5% and 2.5%, respectively. Though trends were statistically significant, temporal variation explained no more than 5% of the variability in abundance. The authors hypothesized possible reasons for declines in northern pike and cisco populations (e.g., high exploitation, variation in recruitment, or a combination of both factors). Interpretation of trends for walleye abundance is complicated by the contribution of hatchery-reared stocks, which represent 30% of the walleye harvested from SMR. Overall, Schaeffer et al. (2011) conclude that the SMR fish community is "remarkably stable."

In conclusion, the three lines of evidence for fish—liver tumour survey data, larval fathead minnow toxicity tests, and long-term fish population surveys—together indicate negligible risk of adverse effects on fish populations from contaminated sediment in SMR. No further action is warranted to protect fish in the AOC.

5.3 Wildlife

In the SMR AOC, wildlife effects related to bird/animal deformities, reproductive problems or degradation of wildlife populations have never been designated as BUIs. Such effects have, however, been designated as requiring further assessment to determine their status. Risks to wildlife (i.e., birds and mammals) due to contaminated sediments in the AOC appear to be minimal, although a comprehensive risk assessment has not been conducted. A 1998 study of common terns on Lime Island (at the far downstream end of the AOC) found a small number of cross-billed chicks (3 of 120 chicks examined) and determined that dioxins, furans, and PCBs in tern eggs were of concern (Selzer 2007). However, dioxin and furan concentrations in surface sediment collected from SMR in 1992 and 1995 (Bedard and Petro 1997, Kilgour et al. 2001) were very low. Also, fish tissue monitoring for public health purposes (i.e., Ontario Fish Contaminant Monitoring Program) has not identified dioxins or furans as a significant issue in SMR (MECP 2009, Awad 2010) and the levels of these contaminants in fish tissue continue to show a declining trend. Therefore, it appears that any significant dioxin and furan exposure of common terns originated from sources outside of SMR AOC, as these birds have a large migratory range, spanning from the northern tip of South America to the Northwest Territories.

ECCC studied the breeding colonies of herring gulls and common terns within the SMR AOC in 2011 and 2012 (Hughes et al. 2014a). Eggs from the gull and tern colonies were collected for artificial incubation in the laboratory and for chemical analysis. Reproduction and development were also assessed in wild populations. During the laboratory study, embryos were examined for physical deformities and embryonic viability was calculated. In addition, ten embryos from each colony were selected for analysis of contaminants. Embryos were analyzed for organochlorine compounds, polybrominated diphenyl ethers (PBDEs), and mercury. Results of the chemical analysis showed that concentrations of organochlorines, PBDEs, and mercury were not sufficiently elevated to impair reproductive success of herring gulls or common terns foraging within the AOC. Embryonic viability was high within herring gull and common tern colonies in the AOC. In addition, herring gull productivity at AOC colonies was high. Common tern productivity within the AOC was low, but consistent with productivity of other colonies in the region. Overall, Hughes et al. (2014a) found little evidence of impaired reproduction or deformities in herring gull or common tern populations attributable to contaminants within the SMR AOC.

Although Hughes et al. (2014a) found little evidence of impairment to colonial waterbird populations within the AOC during the 2011 and 2012 study, the researchers observed deformed embryos at the AOC colonies, while no deformed embryos were found at reference colonies. Given this finding, the same authors conducted a further assessment on embryo deformities in 2013 and 2014 (Hughes et al. 2014b). During the follow-up study, ECCC collected herring gull eggs from two AOC colonies and common tern eggs from one AOC colony, as well as herring gull and common tern eggs from downstream Lake Huron reference colonies. ECCC found that the frequency of embryonic deformities was comparable between AOC colonies and downstream reference colonies for both species and concluded that there was no link between frequency of deformities and either geographical area where the eggs were collected or to contaminant burdens.

In a related study, ECCC assessed breeding populations of common tern (*Sterna hirundo*) and black tern (*Chlidonias niger*) within the SMR AOC between 2010 and 2013 (Hughes et al 2014c). Using nest count data collected during the 2011 and 2012 herring gull and common tern study described above (Hughes et al. 2014a), as well as nest count data collected by the Canadian Wildlife Survey decadal survey of colonial waterbird survey, ECCC found that common terns have nested consistently both within and just beyond the AOC boundary for the last decade. While the black tern populations were low, ECCC concluded that the low populations were reflective of low population density throughout Ontario and was not specific to conditions within the AOC.

5.4 Human Health

A formal human health risk assessment has not been conducted for the AOC; however, qualitative observations are possible based on the available information. As described in Section 3.3, risks associated with contaminant bioaccumulation from AOC sediments appear minimal, due to the low bioaccumulation potential of the primary contaminants. Fish consumption advisories for SMR are driven by mercury and PCBs. The assessment of the fish consumption BUI within the AOC that is currently being finalized by the MECP in consultation with the University of Toronto demonstrates that restrictions on eating fish from the AOC are mild and are typically either similar to or better than the other non-AOC areas of Lakes Superior and Huron. Mercury and PCBs in sediments are generally present at low concentrations and appear to be associated with diffuse regional sources.

The Bellevue Marine Park area supports a lot of recreation, such as sailing and kayaking, and direct contact with environmental media is the most relevant human exposure pathway. ECCC et al. (2002) cited anecdotal reports of floating masses of wood fibre and oil, presumed to be associated with eruptions of gas generated through biodegradation processes in subsurface sediment. In the late 1970s, "mats of oily fibrous material mixed with fine wood chips [were] noticed only occasionally on the Sault Ste. Marie waterfront extending as far as the Lake George Channel" (Hamdy et al. 1978). Such conditions have vastly improved, such that a study conducted from 2013 through 2015 on degradation of aesthetics and eutrophication and undesirable algae supported the delisting of those BUIs in 2018 (Ginou 2016). Thus, it appears that human health concerns do not warrant significant consideration in development of a sediment management strategy.

6. SUMMARY OF KEY FINDINGS AND KNOWLEDGE GAPS

Development of a comprehensive sediment management strategy for SMR requires consideration of several interrelated issues, as described in this CSM. Key issues include definition of risk drivers, status of contaminant sources, fate and transport processes, natural recovery, and specific sediment management option considerations. This section provides an overview of key findings and knowledge gaps.

6.1 Key Findings

The benthic invertebrate community of SMR has largely recovered over the past three decades. There is, however, some evidence of benthic toxicity in localized areas of EBMP and Lake George Channel and the cause(s) of the observed toxicity warrants further evaluation. Findings and recommendations specific to Bellevue Marine Park, EBMP, Lake George Channel, Algoma Boat Slip and the Transport Canada water lot are as follows:

- Bellevue Marine Park: No management action required.
- EBMP: Based on the community structure line of evidence, conditions at EBMP are similar to reference conditions. Therefore, management action to reduce ecological risks in surface sediments is not required. Continued monitoring is recommended, given conflicting outcomes from toxicity and community structure lines of evidence. Also, sediment management measures to avoid accidental exposure of buried contaminants are likely warranted.
- Lake George Channel: Following the 2010 survey, Milani (2012) concluded that further BEAST assessment is not required, given that the two stations requiring management are small localized areas. Continued monitoring at two stations (EC46, EC48) is recommended as a means of improving understanding of the cause(s) of toxicity.
- Algoma Boat Slip: Contaminated sediment was dredged in 2017 to 2019. Post-dredge monitoring is underway.
- Transport Canada water lot: Monitoring is ongoing.

Risks to fish generally are not predicted in SMR, except within the Algoma Boat Slip. The prevalence of liver tumours in fish collected within the AOC has decreased to 6%, which only marginally exceeds the RAP's delisting criteria. In addition, a sediment toxicity assessment on fish found no impacts on fish survival or growth from sediment collected from Bellevue Marine Park or EBMP; the one exception is sediment from Algoma Boat Slip, which was toxic to larval fish. The Algoma Boat Slip was dredged in 2017 and 2019 and post-dredge monitoring is underway.

Because contaminants of concern in SMR sediment are not bioaccumulative, risks to wildlife and human health from AOC sediments are not expected. The primary contaminants of concern in SMR sediment are PAHs and petroleum hydrocarbons, neither of which accumulates in fish tissue. As such, risks to piscivorous wildlife and humans that consume fish from the AOC are not expected.

Sediment stability evaluations suggest near-shore sediments and buried sediments (deeper than 5 cm) are generally stable. Sediment transport modelling in the AOC indicated that under a range of flow conditions, contaminated sediment deposits are stable at sediment depths greater than approximately 5 cm. In addition, there is no evidence of significant ice scour in Bellevue Marine Park or EBMP.

Natural attenuation processes may be occurring, which reduce potential sediment-related risks in the AOC. Generally lower concentrations of metals and PAHs in the top 5 cm of sediment cores collected in 2018 compared to previous sampling indicate that cleaner sediment is being deposited in EBMP. Over time, fewer locations are showing benthic community impacts. Such temporal trends in biological effects and chemistry of surface sediments indicate substantial recovery in sediment quality, likely due to a combination of contaminant biodegradation and burial processes.

6.2 Knowledge Gaps

The following knowledge gaps were identified in this CSM. Improved understanding of these topics would support a more detailed and accurate evaluation of risks and better inform the selection of an appropriate management strategy.

- *Are contaminant sources associated with the Algoma Boat Slip effectively controlled?* Review post-dredge monitoring results to assess the potential for sediment re-contamination.
- *Are conditions at the Transport Canada water lot continuing to improve?* Continue monitoring every five years at nine stations. The next round of monitoring will be conducted in 2023. Monitoring includes bulk sediment chemistry and toxicity testing to confirm that conditions are not worsening based on bulk chemistry and toxicity testing. At that point, the frequency of future monitoring will be reassessed; if conditions are improving, monitoring may be less frequent. If dredging is contemplated in the future, characterization of deeper sediment will be considered.

7. CONCLUSIONS

The purpose of this report was to assemble and synthesize existing information collected within SMR AOC, in order to assist with the development of a contaminated sediment management strategy for the river. As such, a central goal of this report was to reach one of the following three possible conclusions:

1. Sufficient evidence exists to conclude that current conditions in the AOC do not pose a significant risk to the environment and therefore risk management actions are not warranted; OR
2. Insufficient evidence exists to draw conclusions regarding risks to the environment under current conditions and further investigation and/or monitoring is warranted; OR
3. Sufficient evidence exists to conclude that current conditions pose significant risks to the environment and therefore risk management actions are warranted in specific locations.

Of these three alternative conclusions, the first best describes the conclusions of this CSM report. Based on the foregoing analysis, there is low potential for adverse effects in benthic invertebrates, fish, wildlife and humans that are exposed to contaminants of concern in the AOC. Risk management actions are not warranted. Periodic monitoring is recommended to confirm continued natural recovery and address the knowledge gaps listed in Section 6. Additionally, institutional controls to safeguard against accidentally exposing buried contaminants may be warranted.

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TABLES

Table 1. Summary of Sediment Chemistry Results from 2018, East of Bellevue Marine Park (EBMP).

Polycyclic Aromatic Hydrocarbon	Lowest Detection Limit	ISQG[a] (µg/g)	PEL[b] (µg/g)	EC31-5CM	EC31-10CM	EC34-5CM	EC34-10CM	EC35-5CM	EC35-10CM	EC64-5CM	EC64-10CM	EC26-10CM[c]	CS6-10CM[c]	EC52-10CM	EC54-10CM	Units	CS6-1	CS6-2	CS6-3	EC26-1	EC26-2	EC26-3
Acenaphthene	0.0050	0.007	0.089	0.0364	0.0562	0.0613	0.0744	**0.0993**	0.0613	0.0578	0.0697	0.067	0.064	**0.2010**	0.0753	mg/kg	0.0421	0.0543	0.0959	0.0735	0.0713	0.0571
Acenaphthylene	0.0050	0.006	0.128	**0.165**	**0.215**	**0.194**	**0.242**	**0.302**	**0.176**	**0.191**	**0.224**	**0.179**	**0.235**	**0.543**	**0.299**	mg/kg	0.155	0.211	0.339	0.174	0.209	0.155
Anthracene	0.0040	0.047	0.245	0.153	0.216	0.193	**0.269**	**0.448**	0.218	0.204	0.230	0.226	0.213	**1.360**	**0.312**	mg/kg	0.147	0.186	0.306	0.225	0.236	0.217
Benz(a)anthracene	0.010	0.032	0.385	**0.774**	**1.05**	**0.959**	**1.33**	**1.82**	**1.02**	**0.978**	**1.23**	**0.923**	**1.195**	**2.03**	**1.41**	mg/kg	0.77	1.03	1.790	0.942	0.99	0.833
Benzo(a)pyrene	0.010	0.032	0.782	**1.03**	**1.35**	**1.31**	**1.63**	**2.04**	**1.30**	**1.27**	**1.55**	**1.137**	**1.518**	**2.03**	**1.61**	mg/kg	0.99	1.36	2.20	1.19	1.25	0.97
Benzo(b&j)fluoranthene	0.010			1.45	1.91	1.81	2.25	3.01	1.85	1.86	2.26	1.527	2.177	2.47	2.27	mg/kg	1.41	2.05	3.07	1.59	1.69	1.30
Benzo(e)pyrene	0.010			0.824	1.09	1.06	1.27	1.55	1.04	1.04	1.27	0.857	1.253	1.37	1.25	mg/kg	0.81	1.14	1.81	0.899	0.94	0.73
Benzo(g,h,i)perylene	0.010			0.612	0.852	0.832	0.912	1.14	0.759	0.740	0.894	0.768	0.821	1.300	0.883	mg/kg	0.57	0.721	1.170	0.787	0.869	0.647
Benzo(k)fluoranthene	0.010	0.240[d]		**0.471**	**0.610**	**0.591**	**0.713**	**0.991**	**0.604**	**0.576**	**0.708**	**0.522**	**0.715**	**0.917**	**0.788**	mg/kg	0.451	0.673	1.020	0.569	0.571	0.426
Chrysene	0.010	0.057	0.862	0.856	**1.13**	**1.06**	**1.37**	**1.82**	**1.10**	**1.09**	**1.35**	**1.014**	**1.276**	**1.88**	**1.47**	mg/kg	0.81	1.12	1.90	1.060	1.10	0.88
Dibenz(a,h)anthracene	0.0050	0.006	0.135	**0.146**	**0.202**	**0.183**	**0.208**	**0.292**	**0.183**	**0.185**	**0.223**	**0.172**	**0.212**	**0.275**	**0.236**	mg/kg	0.141	0.181	0.314	0.166	0.200	0.151
Fluoranthene	0.010	0.111	2.355	1.12	1.53	1.46	1.83	**3.11**	1.47	1.43	1.83	1.423	1.797	**4.90**	2.02	mg/kg	1.11	1.54	2.74	1.50	1.54	1.23
Fluorene	0.010	0.021	0.144	0.066	0.094	0.103	0.125	**0.186**	0.112	0.098	0.111	0.109	0.100	**0.469**	0.124	mg/kg	0.072	0.092	0.137	0.109	0.117	0.100
Indeno(1,2,3-cd)pyrene	0.010	0.200[d]		**0.603**	**0.820**	**0.765**	**0.934**	**1.11**	**0.742**	**0.723**	**0.942**	**0.781**	**0.831**	**1.620**	**0.828**	mg/kg	0.55	0.708	1.240	0.854	0.856	0.634
Naphthalene	0.010	0.035	0.391	**0.450**	**0.597**	**0.628**	**0.659**	**0.729**	**0.705**	**0.607**	**0.599**	**0.815**	**0.423**	**0.652**	**0.812**	mg/kg	0.359	0.391	0.519	0.861	0.893	0.691
Perylene	0.010			0.338	0.435	0.458	0.544	0.670	0.425	0.425	0.494	0.386	0.520	0.753	0.533	mg/kg	0.314	0.480	0.766	0.409	0.429	0.321
Phenanthrene	0.010	0.042	0.515	0.492	**0.671**	**0.674**	**0.831**	**1.22**	**0.697**	**0.642**	**0.734**	**0.729**	**0.643**	**4.270**	**0.811**	mg/kg	0.45	0.575	0.909	0.781	0.781	0.626
Pyrene	0.010	0.053	0.875	**0.988**	**1.33**	**1.34**	**1.71**	**2.59**	**1.24**	**1.25**	**1.61**	**1.220**	**1.603**	**4.37**	**1.79**	mg/kg	0.99	1.38	2.44	1.270	1.31	1.08
SUM 18 PARENT PAHs		4[e]		**10.57**	**14.16**	**13.68**	**16.90**	**23.13**	**13.70**	**13.37**	**16.33**	**12.86**	**15.60**	**31.41**	**17.52**	**mg/kg**	**10.13**	**13.89**	**22.77**	**13.46**	**14.05**	**11.05**
1-Methylnaphthalene	0.010			0.042	0.055	0.059	0.069	0.083	0.068	0.061	0.059	0.066	0.049	0.081	0.066	mg/kg	0.039	0.045	0.063	0.070	0.072	0.056
2-Methylnaphthalene	0.010	0.020	0.201	0.072	0.094	0.106	0.119	0.127	0.111	0.103	0.098	0.123	0.082	0.106	0.124	mg/kg	0.063	0.077	0.105	0.129	0.135	0.104
C2 sub'd B(a)A/chrysene	0.040			0.217	0.270	0.305	0.457	0.523	0.244	0.289	0.387	0.218	0.427	0.264	0.462	mg/kg	0.259	0.401	0.620	0.227	0.236	0.190
C2 Fluorenes	0.040			0.060	0.057	0.086	0.107	0.308	0.083	0.087	0.106	0.092	0.126	0.121	0.130	mg/kg	0.080	0.108	0.191	0.100	0.098	0.079
C2 Naphthalenes	0.0052			0.162	0.211	0.198	0.230	0.357	0.369	0.315	0.341	0.228	0.230	0.239	0.231	mg/kg	0.152	0.200	0.337	0.261	0.226	0.197
C2 Phenanthrenes/Anthracenes	0.040			0.130	0.182	0.221	0.305	0.579	0.184	0.166	0.239	0.182	0.299	0.359	0.305	mg/kg	0.167	0.257	0.473	0.178	0.208	0.161
C3 Benzanthracenes/Chrysenes	0.040			0.218	0.303	0.339	0.452	0.421	0.275	0.313	0.435	0.185	0.426	0.169	0.476	mg/kg	0.260	0.398	0.621	0.169	0.191	0.196
C3 Fluorenes	0.040			0.083	0.108	0.154	0.251	0.388	0.125	0.110	0.173	0.095	0.241	0.114	0.271	mg/kg	0.139	0.187	0.397	0.076	0.108	0.100
C3 Naphthalenes	0.0052			0.060	0.076	0.091	0.160	0.188	0.094	0.083	0.108	0.091	0.120	0.159	0.108	mg/kg	0.074	0.095	0.191	0.099	0.098	0.075
C3 Phenanthrenes/Anthracenes	0.040			0.105	0.118	0.206	0.307	0.516	0.168	0.145	0.250	0.105	0.298	0.153	0.358	mg/kg	0.173	0.243	0.478	0.097	0.125	0.094
C4 Benzanthracenes/Chrysenes	0.040			0.188	0.211	0.253	0.273	0.279	0.157	0.224	0.303	0.141	0.369	0.138	0.325	mg/kg	0.208	0.329	0.569	0.162	0.146	0.114
C4 Naphthalenes	0.0052			0.003	0.066	0.067	0.087	0.211	0.030	0.003	0.090	0.062	0.135	0.096	0.114	mg/kg	0.069	0.116	0.219	0.062	0.075	0.048
C4 Phenanthrenes/Anthracenes	0.040			0.152	0.177	0.219	0.319	0.480	0.235	0.154	0.250	0.289	0.227	0.317	0.289	mg/kg	0.146	0.161	0.375	0.296	0.280	0.291
C1 Benz(a)Anthracenes/Chrysenes	0.040			0.283	0.394	0.382	0.491	0.683	0.386	0.381	0.462	0.347	0.510	0.530	0.566	mg/kg	0.301	0.490	0.740	0.366	0.380	0.294
C1 Fluoranthenes/Pyrenes	0.040			0.463	0.655	0.611	0.779	1.35	0.602	0.598	0.743	0.585	0.803	1.550	0.855	mg/kg	0.49	0.666	1.250	0.597	0.635	0.523
C1 Fluorenes	0.040			0.003	0.003	0.003	0.069	0.118	0.061	0.003	0.003	0.058	0.061	0.139	0.057	mg/kg	<0.060	0.046	0.075	0.055	0.063	0.056
C1 Phenanthrenes/Anthracenes	0.040			0.176	0.247	0.260	0.355	0.615	0.262	0.230	0.293	0.264	0.297	0.830	0.316	mg/kg	0.178	0.245	0.469	0.271	0.300	0.221
SUM 16 ALKY PAHs				**2.42**	**3.23**	**3.56**	**4.83**	**7.23**	**3.45**	**3.26**	**4.34**	**3.13**	**4.70**	**5.37**	**5.05**	mg/kg	**2.80**	**4.06**	**7.17**	**3.22**	**3.38**	**2.80**
EPA 34 (18 PAR + 16 ALKY)				**12.99**	**17.39**	**17.24**	**21.73**	**30.35**	**17.16**	**16.63**	**20.67**	**15.99**	**20.30**	**36.78**	**22.57**	mg/kg	**12.93**	**17.96**	**29.94**	**16.67**	**17.43**	**13.85**
% of 18 PAH18 to PAH34				**81.4**	**81.4**	**79.4**	**77.8**	**76.2**	**79.9**	**80.4**	**79.0**	**80.4**	**76.8**	**85.4**	**77.6**		**78.4**	**77.4**	**76.0**	**80.7**	**80.6**	
Retene	0.010			0.448	0.555	0.542	0.680	1.23	0.731	0.400	0.513	0.981	0.275	1.060	0.614	mg/kg	0.19	0.215	0.425	1.070	0.984	0.888

[a] Canadian interim sediment quality guideline; [b] Canadian probable effect level; [c] QA/QC site - values represent mean of 3 samples

Source: Milani and Grapentine 2019c

Values in red are 1/2 the lowest detection limit. Bolded values exceed probable effect level (PEL) or lowest effect level if PEL not available.

Table 2. Comparison of 2009 and 2018 COA Framework Results for Sediment in East of Bellevue Marine Park (EBMP)

Standard Indicators (0-10 cm)									Additional Indicators											
Site	Year	Sediment Chemistry	Overall Toxicity[1]	Benthos Alteration	2018 Metal >high SQG; PAH, PHC > Reference mean by 20%[2]	$\sum PAH_{34}$ ($\sum PAH_{par16}$)	$\sum PHC_{F2\text{-}F4}$	Assessment Outcome	Sediment Chemistry (0-5 cm)	Overall Toxicity[1] (0-5 cm)	Larval Fish Survival (0-5 & 0-10 cm)	Larval Fish Growth (0-5 & 0-10 cm)	Larval Fish Hatching[3] (0-5 & 0-10 cm)	Larval Fish Deformities[4] (0-5 & 0-10 cm)	*H. azteca* 42-d Survival (0-5 cm)	*H. azteca* 42-d Survival[5] (0-10 cm)	*H. azteca* 42-d Growth[5] (0-5 cm)	*H. azteca* 42-d Growth[5] (0-10 cm)	*H. azteca* 42-d Reproduction[5] (0-5 cm)	*H. azteca* 42-d Reproduction[5] (0-10 cm)
CS6	2018	■	▣	□	PAHs, PHC, Fe, Pb, Zn	20.3 (13.8)	6266	Determine reason(s) for sediment toxicity	-	-	-	-	-	-	-	-	-	-	-	-
CS6	2008	■	▣	□	PAHs, Fe, Pb, Zn	(16.2)	750	Determine reason(s) for sediment toxicity	-	-	-	-	-	-	-	-	-	-	-	-
EC26	2018	■	□	□	PAHs, PHC	16.0 (11.6)	1392	No further action	-	-	-	-	-	-	-	-	-	-	-	-
EC26	2008	■	■	▣	PAHs	(17.8)	1417	Management action required	-	-	-	-	-	-	-	-	-	-	-	-
EC31	2018	■	■	□	PAHs, PHC, Fe	17.4 (12.6)	3679	Determine reason(s) for sediment toxicity	■	■	□	□	□	□	□	▣	□	□	□	□
EC31	2009	■	▣	□	PAHs, Fe	31.7	4910	Determine reason(s) for sediment toxicity	-	-	-	-	-	-	-	-	-	-	-	-
EC34	2018	■	■	□	PAHs, PHC, Fe, Zn	21.7 (15.1)	5316	Determine reason(s) for sediment toxicity	■	■	□	□	□	□	-	-	-	-	-	-
EC34	2009	■	▣	□	PAHs, PHC, Fe, Zn	(19.8)	2730	Determine reason(s) for sediment toxicity	-	-	-	-	-	-	-	-	-	-	-	-
EC35	2018	■	■	□	PAHs, PHC, Fe, Pb, Zn	30.4* (12.2)*	5173*	Determine reason(s) for sediment toxicity	■	■	□	□	□	□	□	▣	□	□	□	□
EC35	2009	■	▣	□	PAHs, PHC, Fe	(25.0)	5055	Determine reason(s) for sediment toxicity	-	-	-	-	-	-	-	-	-	-	-	-
EC52	2018	■	□	□	PAHs, PHC	36.8 (29.3)	320	No further action	-	-	-	-	-	-	-	-	-	-	-	-
EC52	2010	■	▣	▣	PAHs, PHC	43.6	1031	Management action required	-	-	-	-	-	-	-	-	-	-	-	-
EC54	2018	■	▣	□	PAHs, PHC	22.6 (15.7)	5207	Determine reason(s) for sediment toxicity	-	-					-	-	-	-	-	-
EC54	2010	■	▣	▣	PAHs, PHC, Fe	32.1	4901	Management action required	-	-	-	-	-	-	-	-	-	-	-	-
EC64	2018	■	■	□	PAHs, PHC, Fe, Zn	20.7 (14.6)	5381	Determine reason(s) for sediment toxicity	■	■	□	□	□	□	-	-	-	-	-	-
EC64	2010	■	■	▣	PAHs, PHC, Fe, Pb, Zn	(22.1)	9055	Management action required	-	-	-	-	-	-	-	-	-	-	-	-

[1] Integrated toxicity (10 endpoints)

[2] see Reference conc > 20% worksheet

[3] Includes various endpoints (see Parrott and Milani 2019)

[4] Includes various deformites (see Parrott and Milani 2019)

[5] Bartlett et al. 2019

* highest at 0-5 cm

FIGURES

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RAMBOLL

DRAFTED BY: ARG

DATE: 3/21/2018

Area of Concern

St. Marys River

Ontario, Canada

FIGURE

1

PROJECT:

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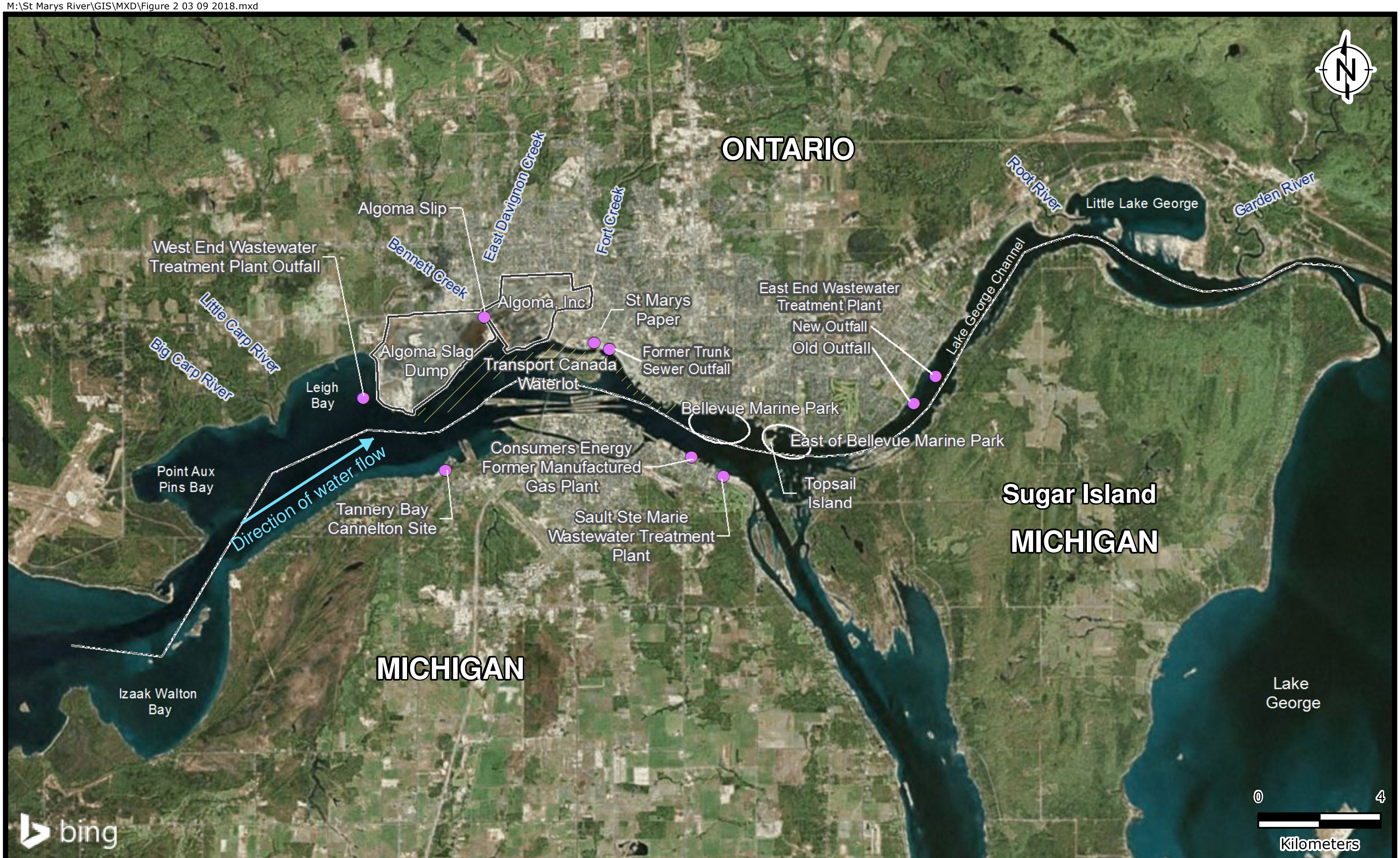


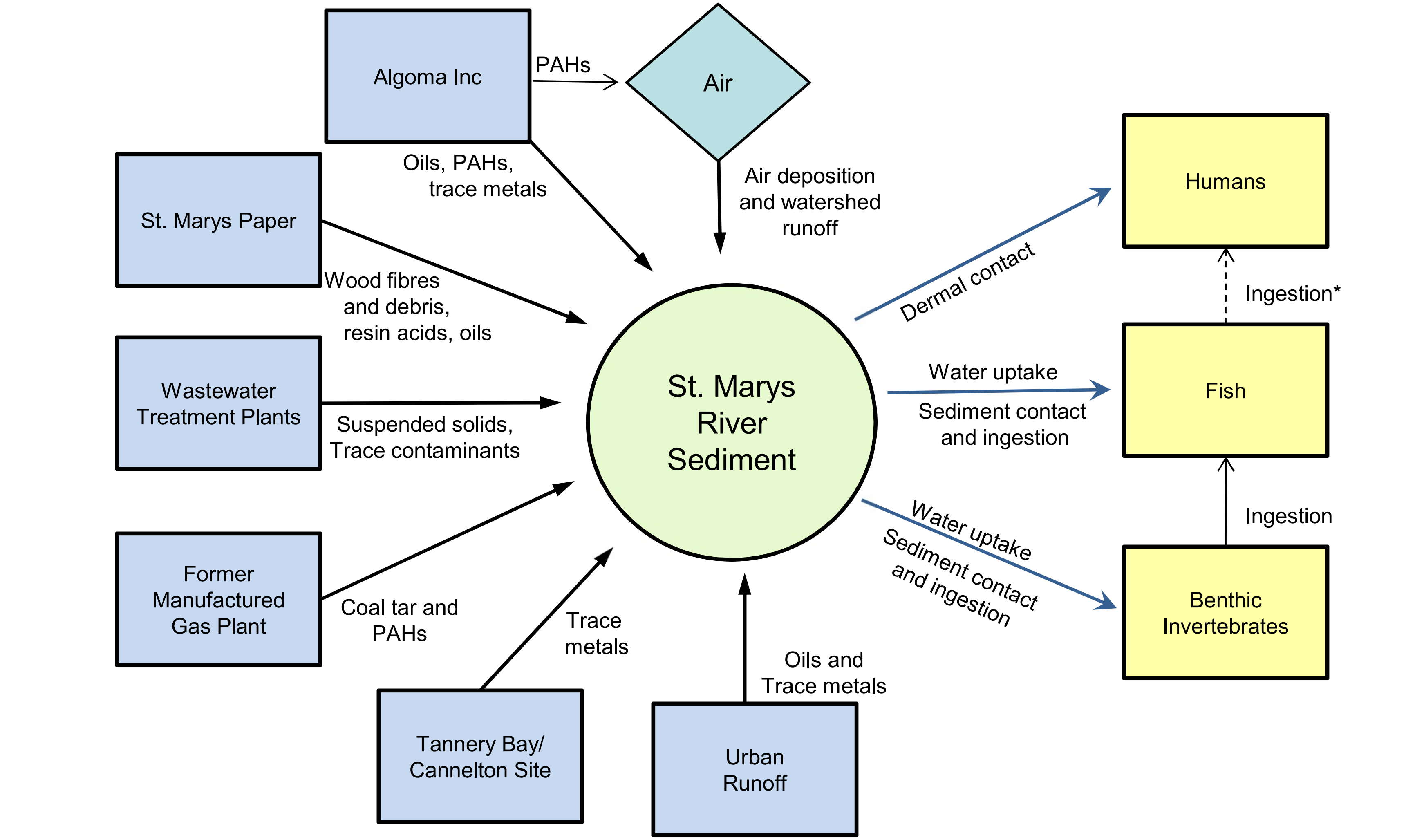
RAMBOLL

DRAFTED BY: ARG | DATE: 3/26/2019

Site Features
St. Marys River
Ontario, Canada

FIGURE
2

PROJECT:

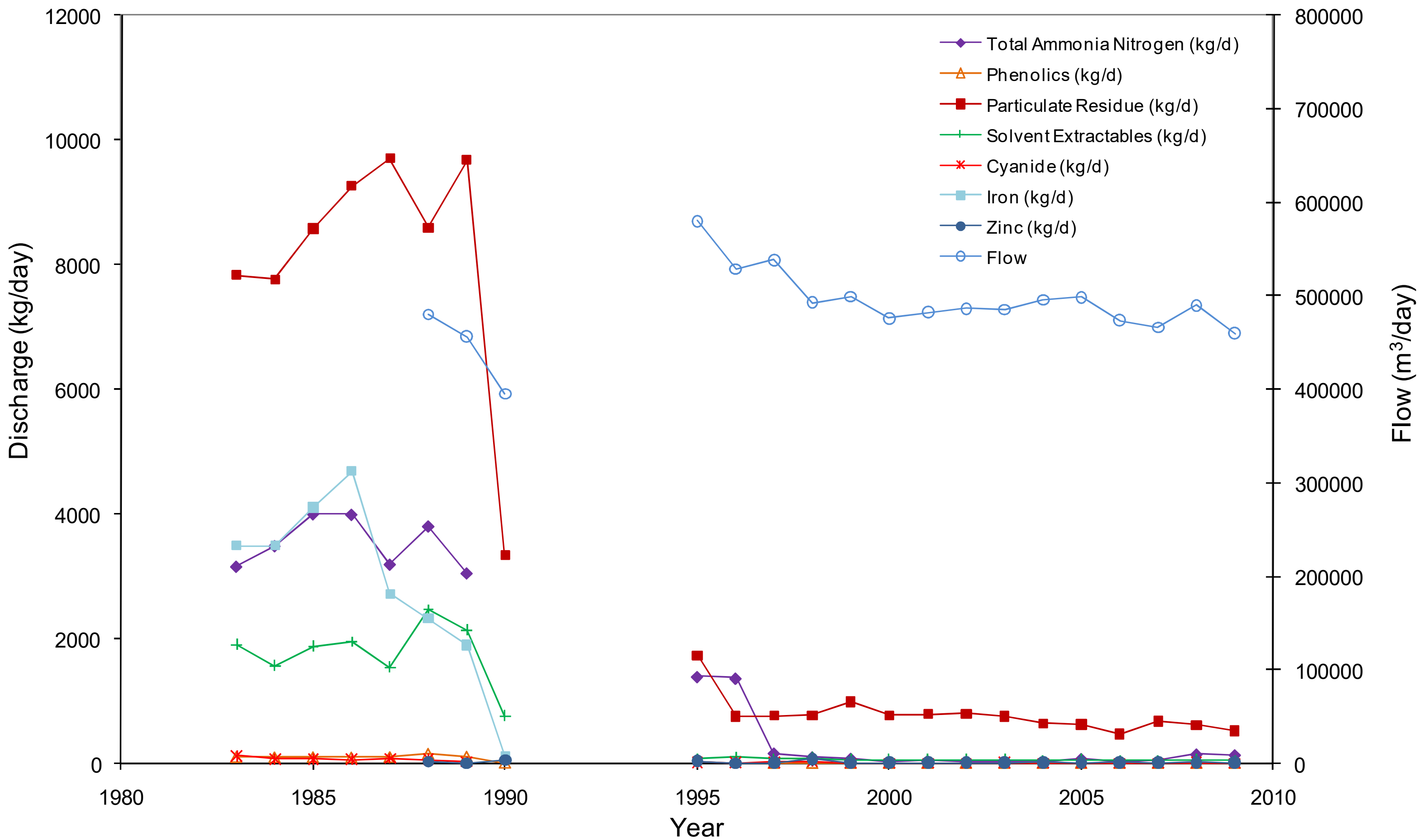


* Fish ingestion occurs but it is not a significant exposure pathway for the major sediment contaminants of interest (petroleum, hydrocarbons, PAHs), which do not bioaccumulate in fish.

RAMBOLL

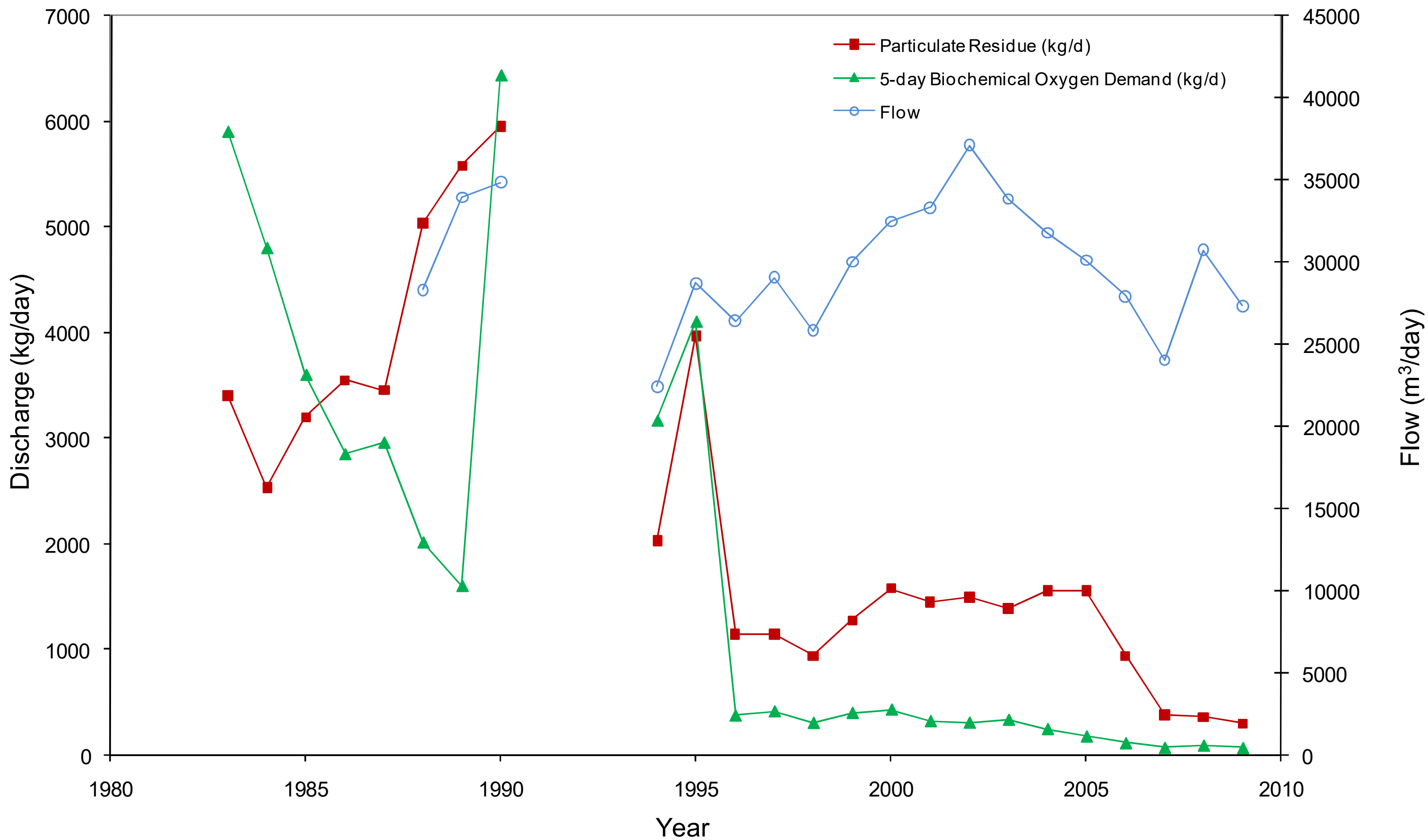
Contaminant Sources and Pathways
St. Marys River
Ontario, Canada

Figure 3



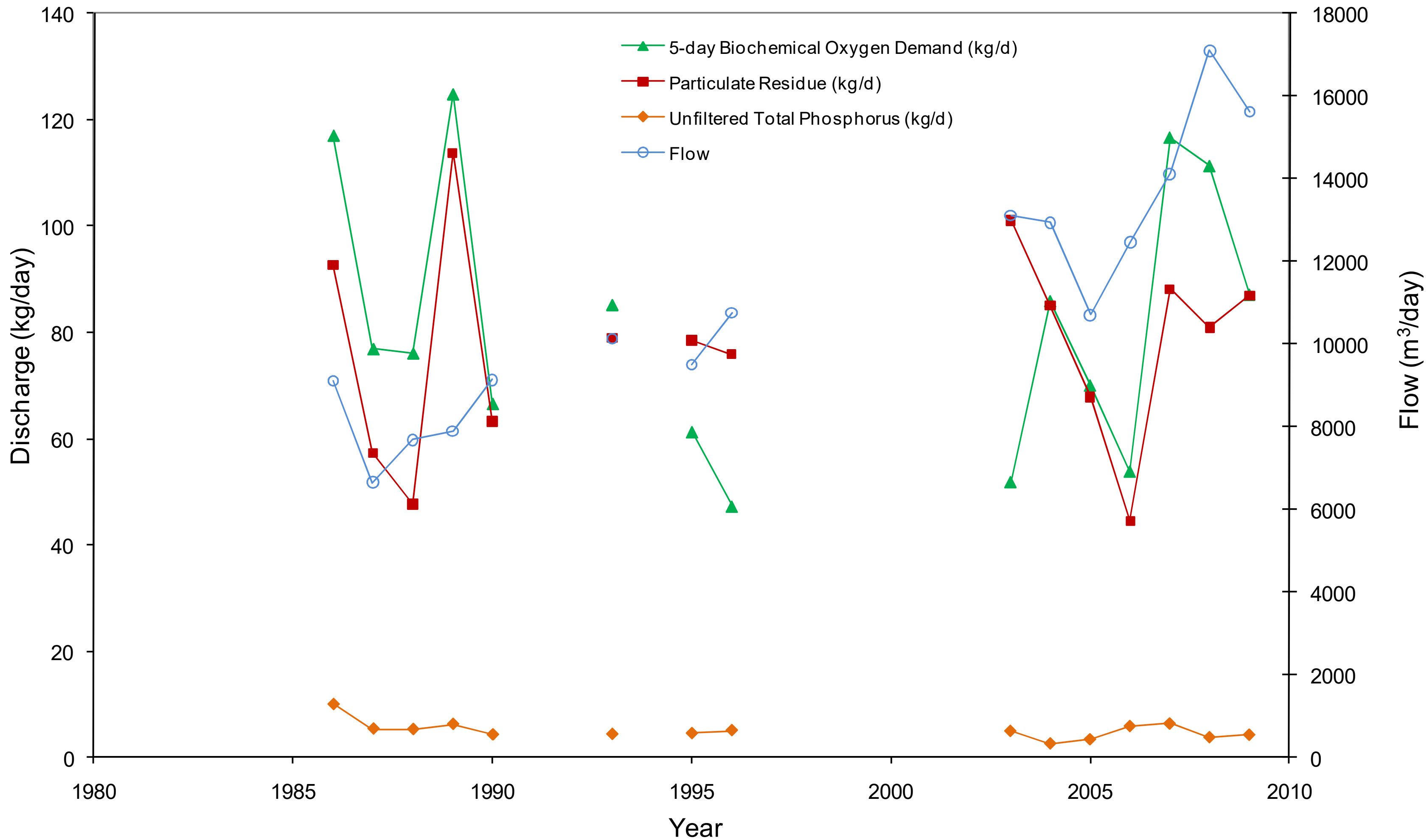
Data compiled by MECP. No data were reported between 1991 and1994.

RAMBOLL	**Algoma Effluent Discharges** St. Marys River Ontario, Canada	Figure 4



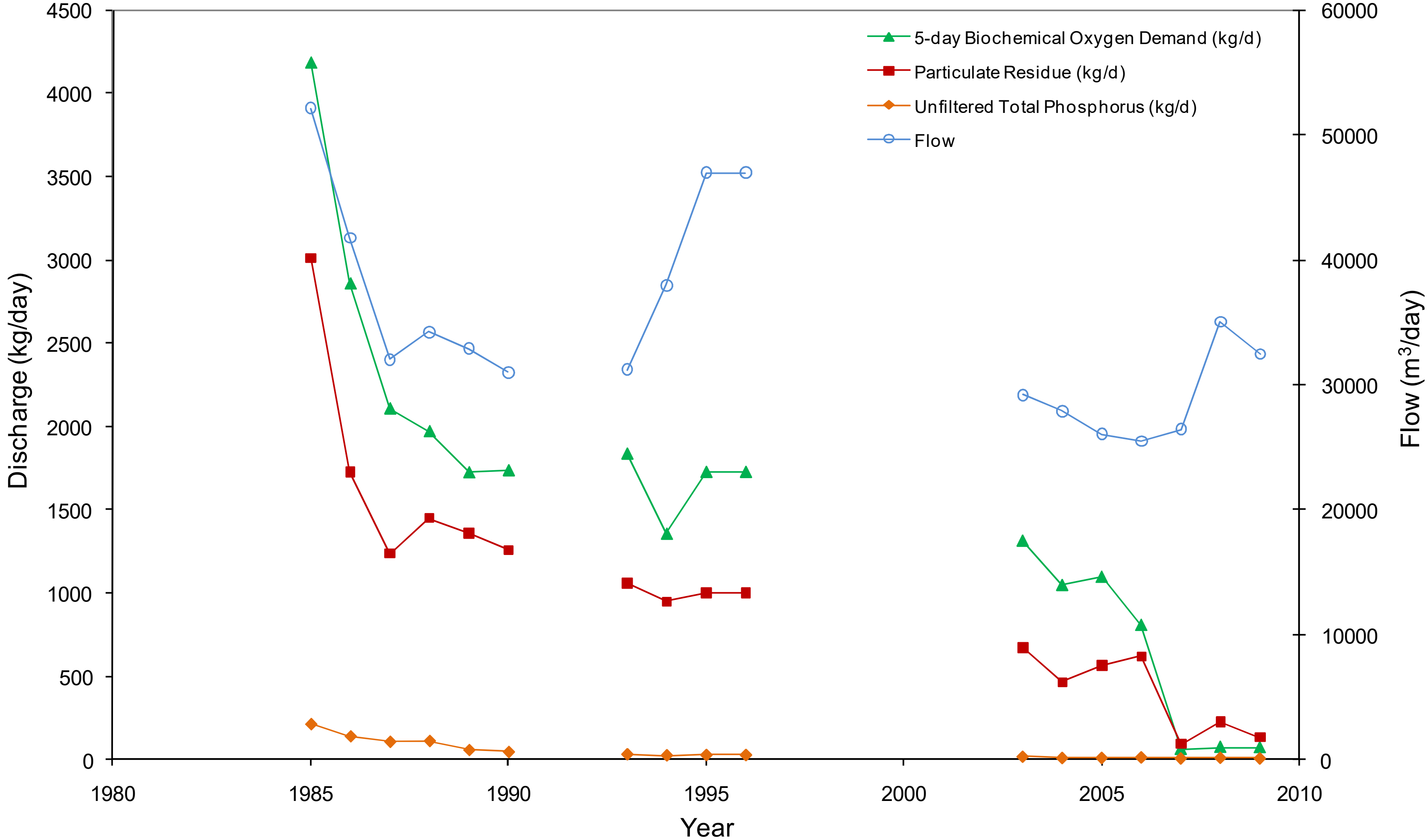
Data compiled by MECP. No data were reported between 1991 and1993.

RAMBOLL	**St. Marys Paper Effluent Discharges** St. Marys River Ontario, Canada	Figure 5



Data compiled by MECP. Data were not available for some years at the time of this report.

RAMBOLL	**West End Water Pollution Control Plant Effluent Discharges** St. Marys River, Ontario, Canada	Figure 6



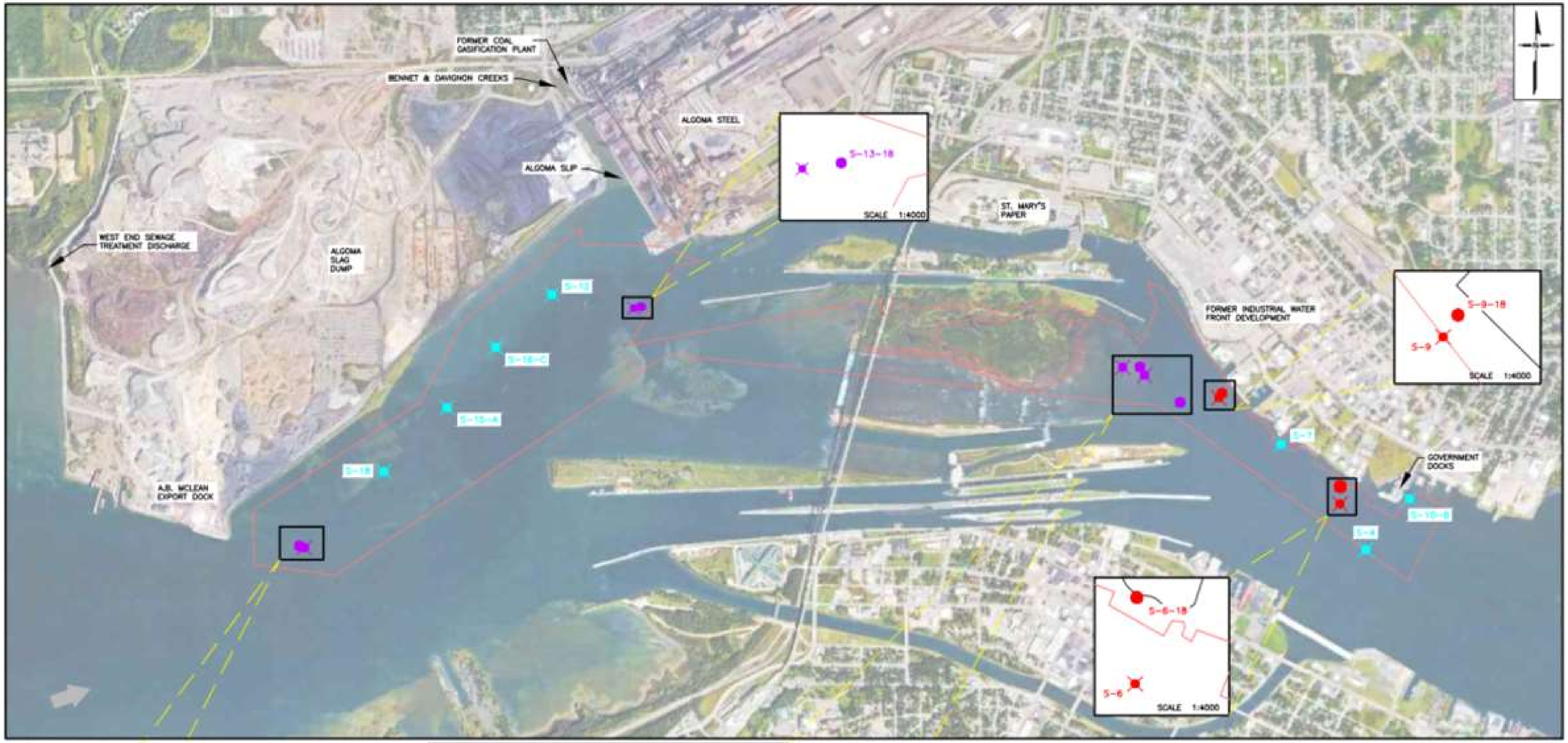
Data compiled by MECP. Data were not available for some years at the time of this report.

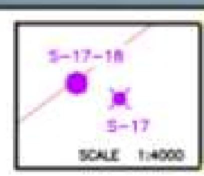
RAMBOLL

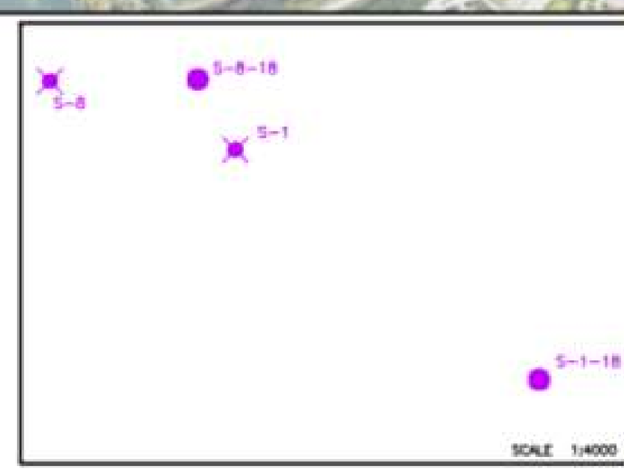
East End Wastewater Plant Effluent Discharges

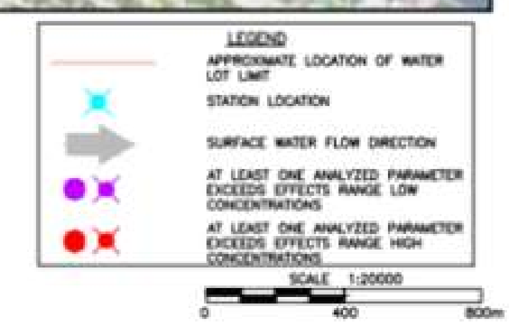
St. Marys River, Ontario, Canada

Figure 7









NOTES:

1. Scale and site infrastructure locations are approximate
2. Information on this figure may be lost if it is photocopied, faxed or printed in other than its original size and colors
3. 'm': meters

SOURCE:

SNC-Lavalin. 2019. Update to Step 6 and Decision 5 of the Canada-Ontario Decision-Making Framework for Assessment of Great Lakes Contaminated Sediment – Sediment Quality Evaluation, Transport Canada, Sault Ste. Marie Harbour. Prepared by SNC-Lavalin Environment and Geoscience. March 29.

RAMBOLL

DATE: 8/26/2019

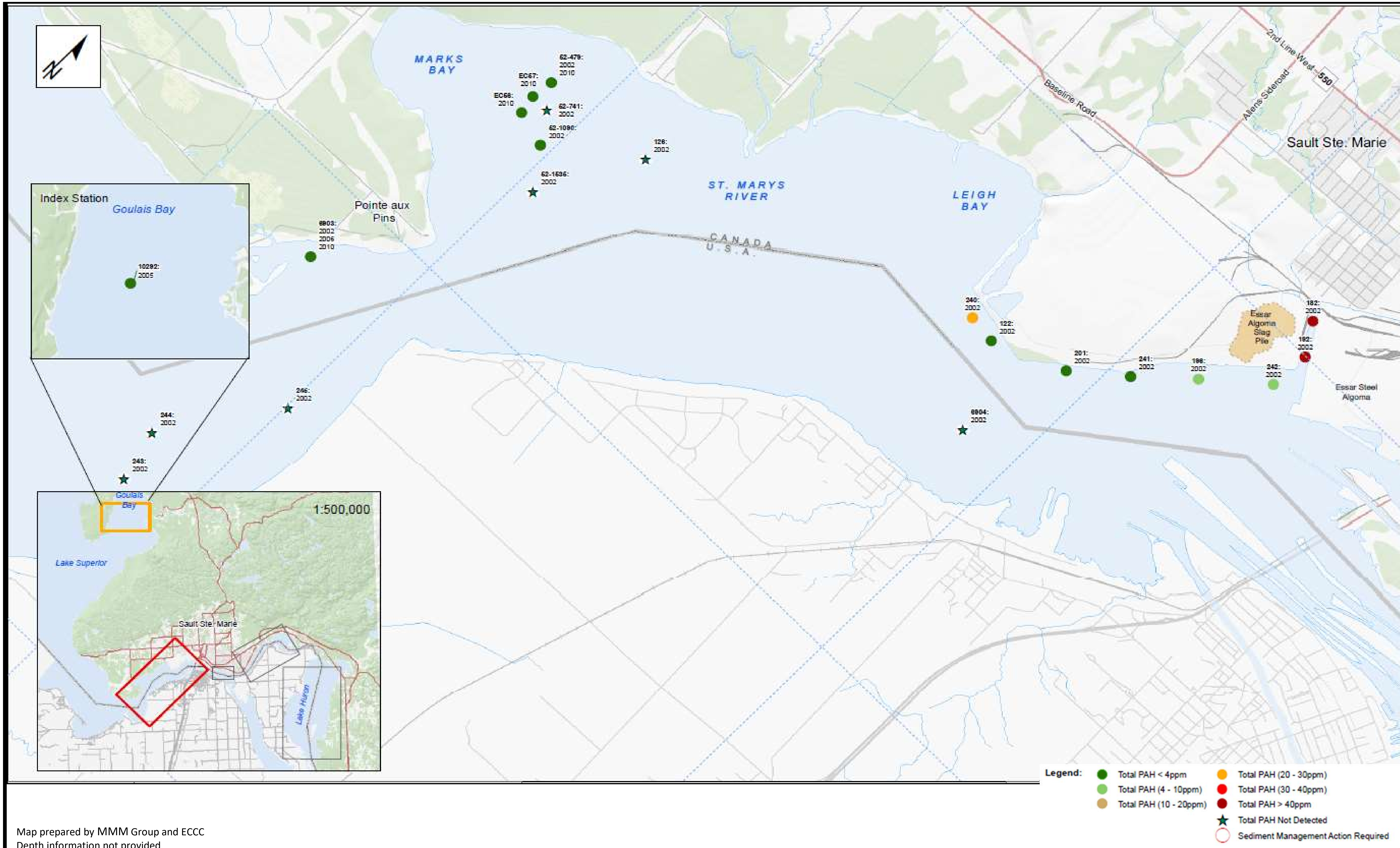
2018 Sampling Locations and Results at Transport Canada Water Lot

St. Marys River

Ontario, Canada

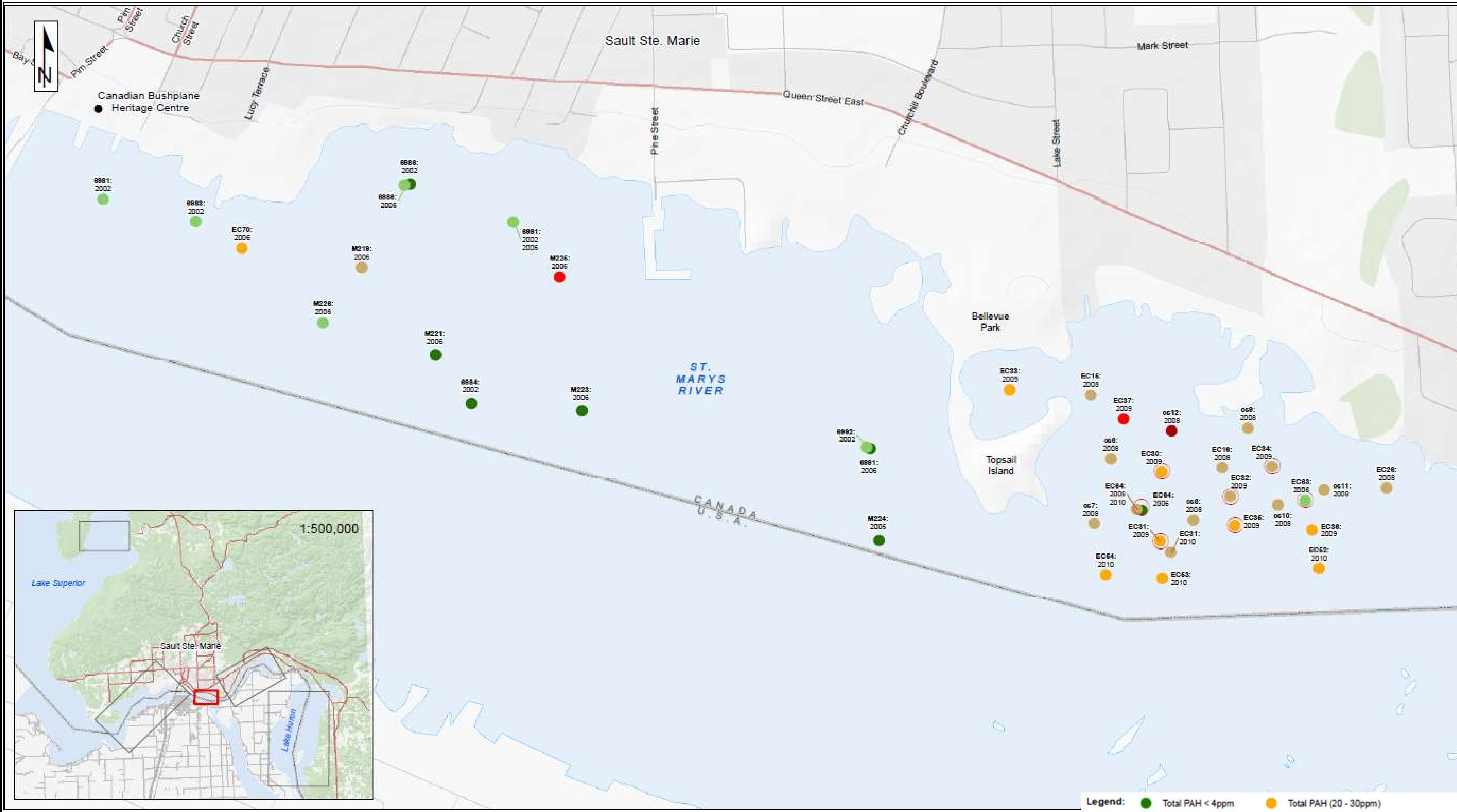
FIGURE

8



Map prepared by MMM Group and ECCC
Depth information not provided

RAMBOLL	**Total PAHs in Sediment (2002-2010)** St. Marys River Ontario, Canada	FIGURE **9a**
DATE: 1/10/2020		1690014264



Legend:
- Total PAH < 4ppm
- Total PAH (4 - 10ppm)
- Total PAH (10 - 20ppm)
- Total PAH (20 - 30ppm)
- Total PAH (30 - 40ppm)
- Total PAH > 40ppm
- Total PAH Not Detected
- Sediment Management Action Required

Map prepared by MMM Group and ECCC
Depth information not provided

RAMBOLL

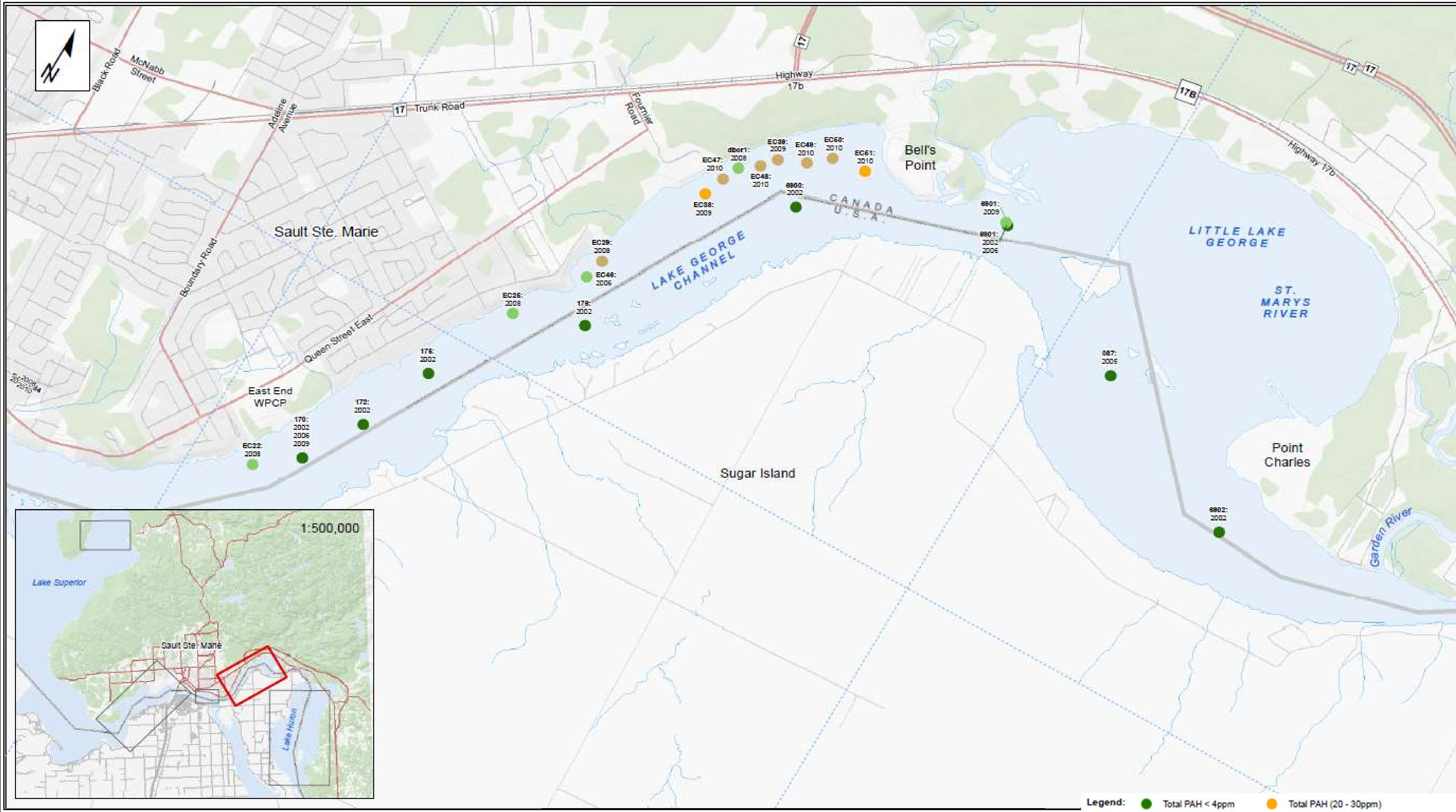
DATE: 1/10/2020

Total PAHs in Sediment (2002-2010)

St. Marys River
Ontario, Canada

FIGURE
9b

1690014264



Legend:
- Total PAH < 4ppm
- Total PAH (4 - 10ppm)
- Total PAH (10 - 20ppm)
- Total PAH (20 - 30ppm)
- Total PAH (30 - 40ppm)
- Total PAH > 40ppm
- Total PAH Not Detected
- Sediment Management Action Required

Map prepared by MMM Group and ECCC
Depth information not provided

RAMBOLL

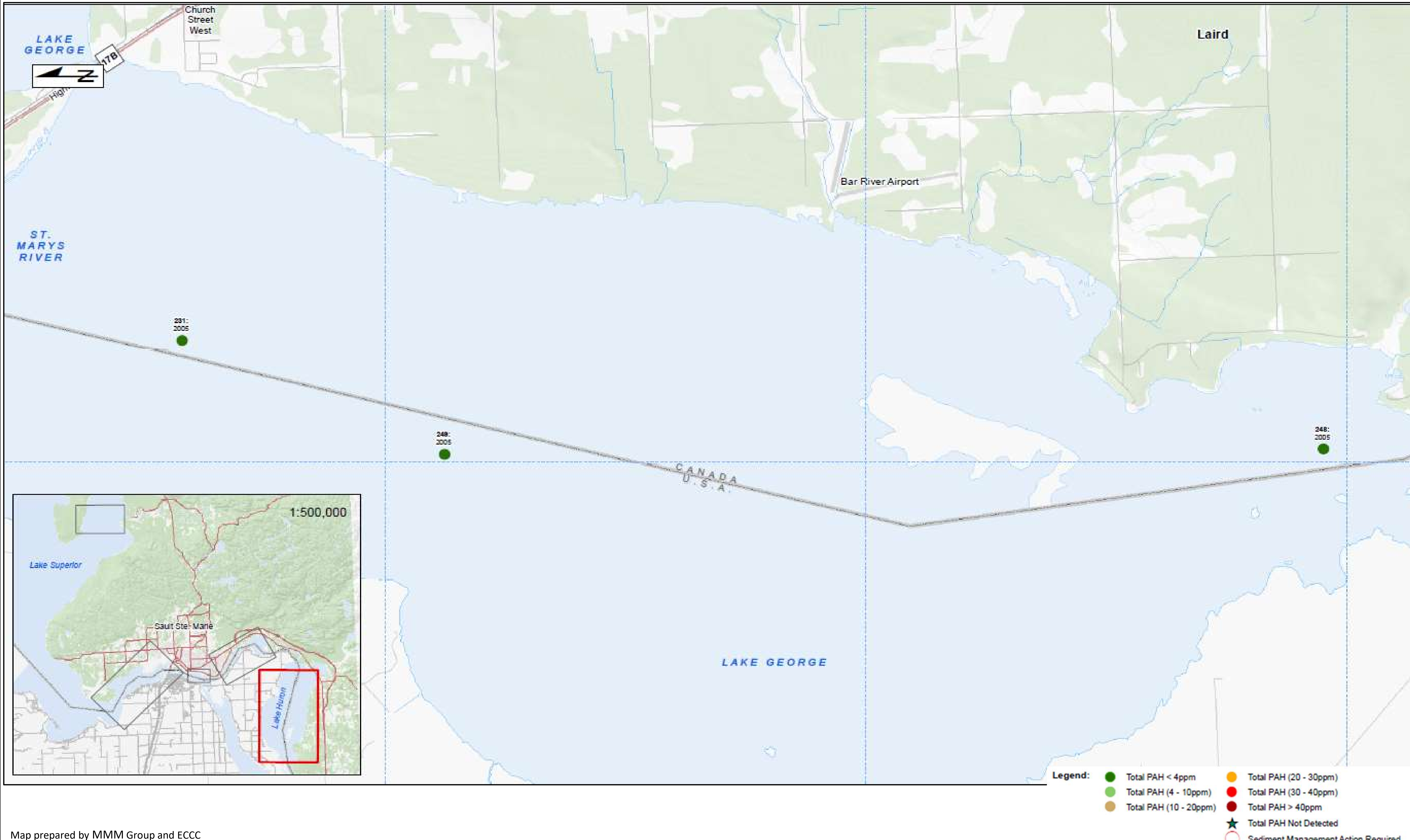
DATE: 1/10/2020

Total PAHs in Sediment (2002-2010)

St. Marys River
Ontario, Canada

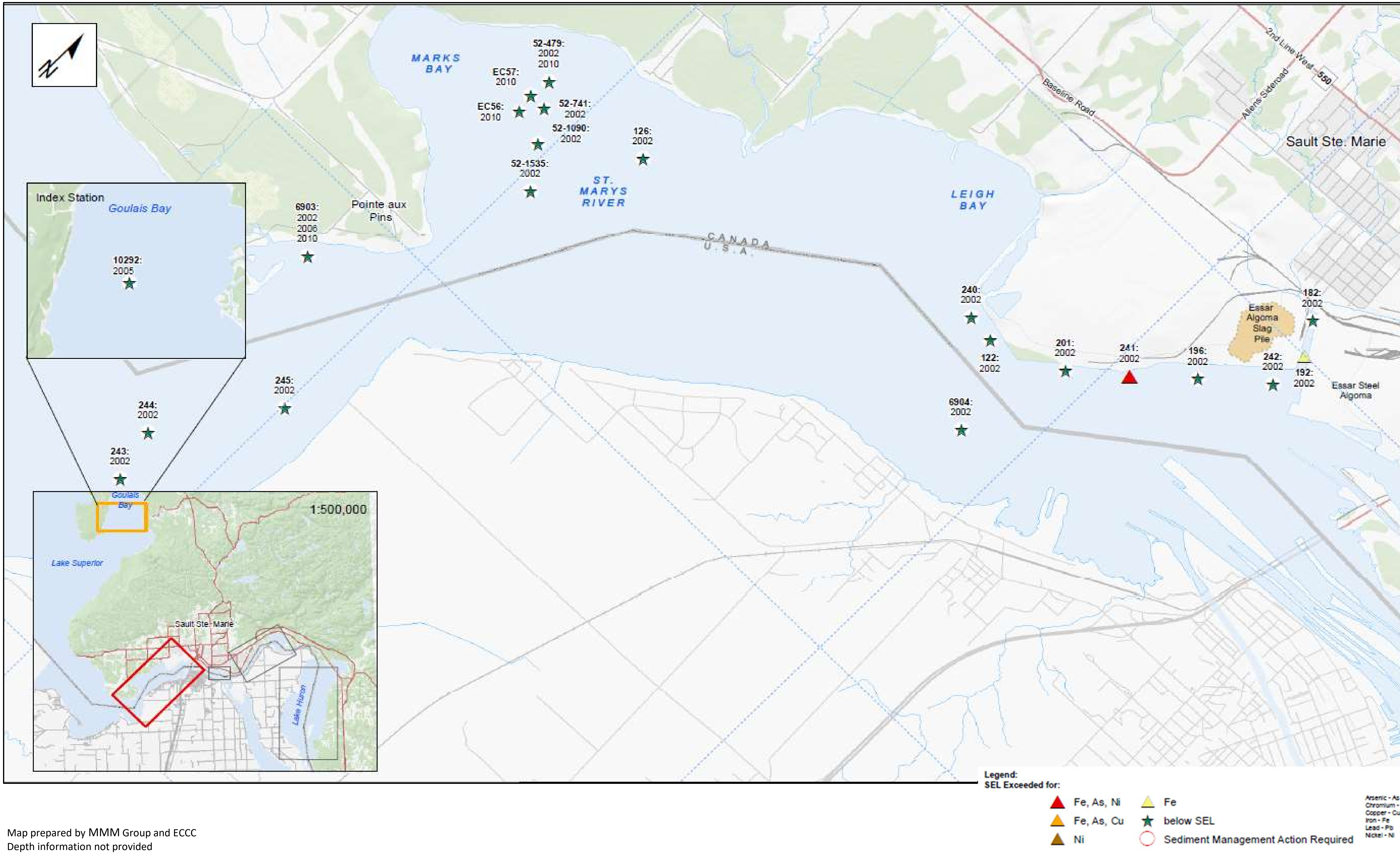
FIGURE
9c

1690014264



Map prepared by MMM Group and ECCC
Depth information not provided

RAMBOLL	**Total PAHs in Sediment (2002-2010)** St. Marys River Ontario, Canada	FIGURE **9d**
DATE: 1/10/2020		1690014264



Map prepared by MMM Group and ECCC
Depth information not provided

RAMBOLL

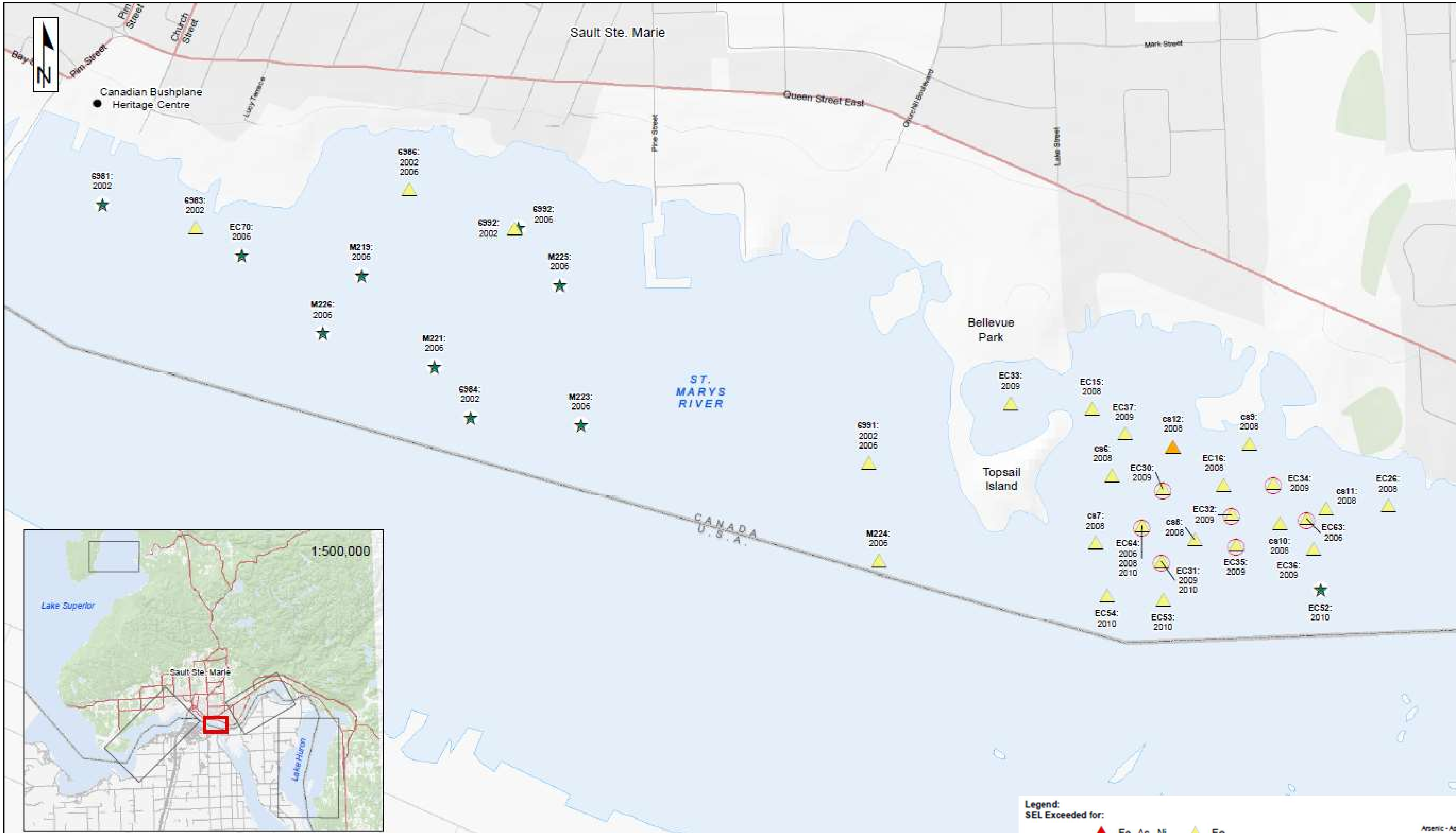
DATE: 1/10/2020

Exceedances of the Severe Effect Level for Metals in Sediment (2002-2010)

St. Marys River, Ontario, Canada

FIGURE **10a**

1690014264



Map prepared by MMM Group and ECCC
Depth information not provided

RAMBOLL

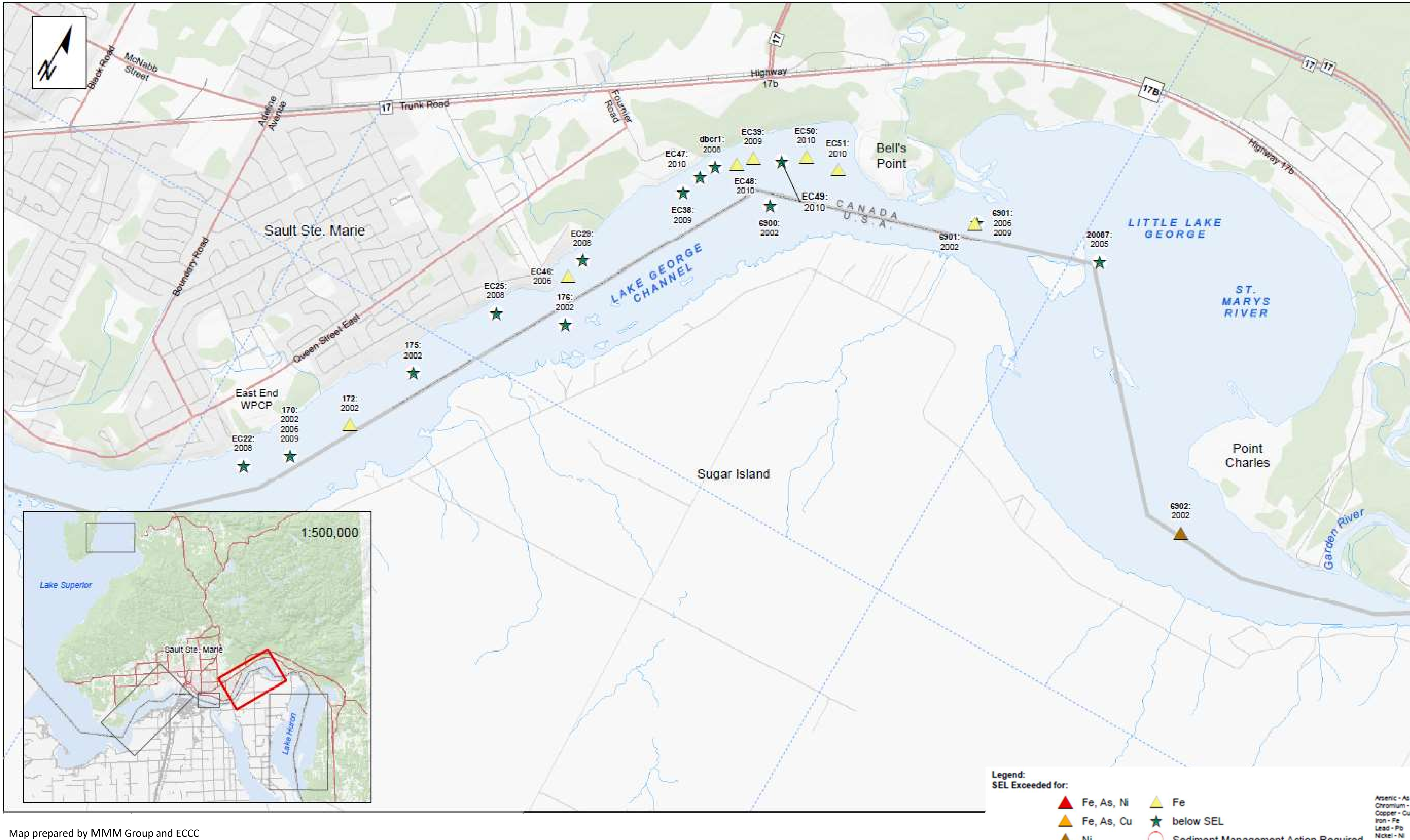
Exceedances of the Severe Effect Level for Metals in Sediment (2002-2010)

St. Marys River, Ontario, Canada

FIGURE **10b**

DATE: 1/10/2020

1690014264



Map prepared by MMM Group and ECCC
Depth information not provided

RAMBOLL

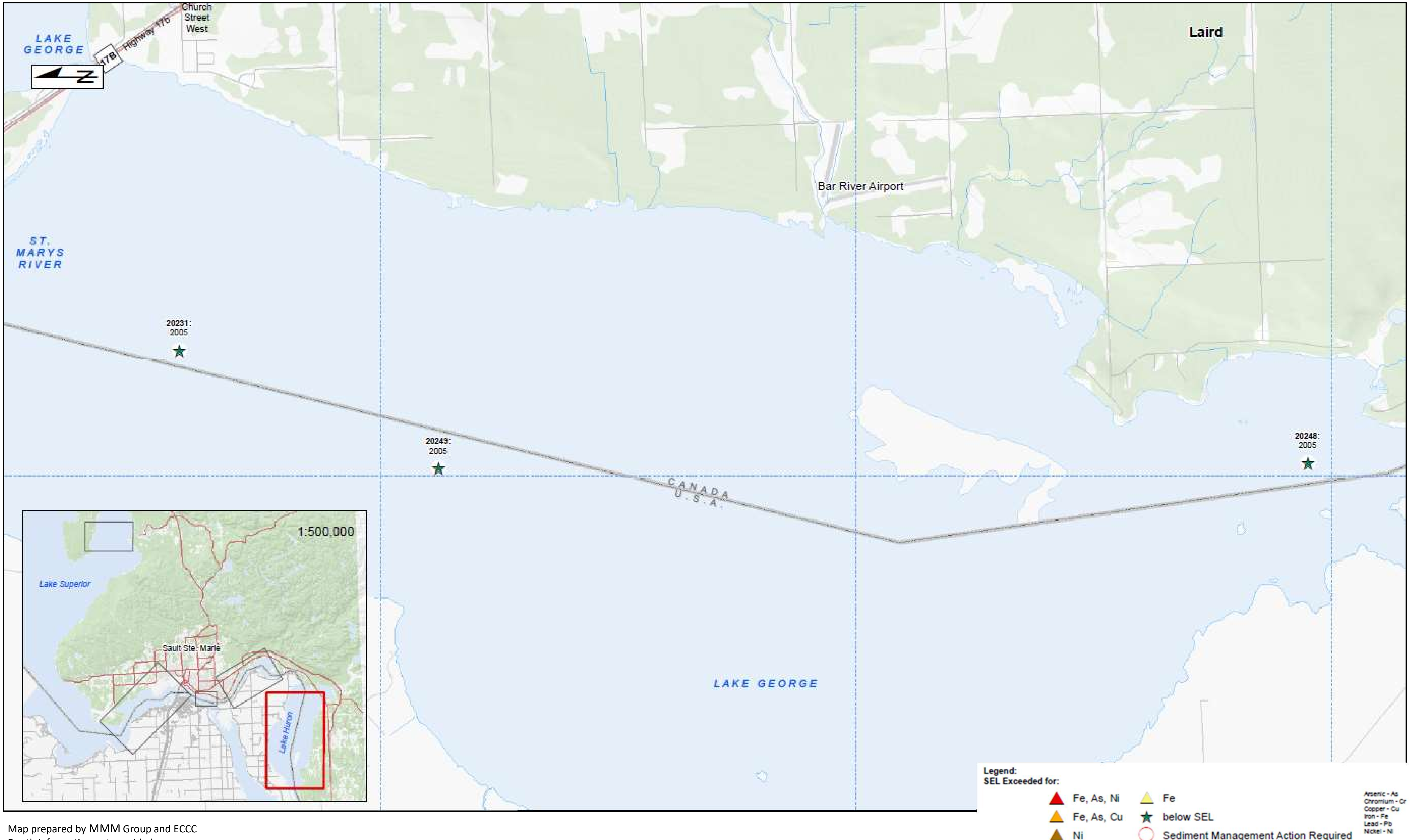
DATE: 1/10/2020

Exceedances of the Severe Effect Level for Metals in Sediment (2002-2010)

St. Marys River, Ontario, Canada

FIGURE **10c**

1690014264



Map prepared by MMM Group and ECCC
Depth information not provided

RAMBOLL

DATE: 1/10/2020

Exceedances of the Severe Effect Level for Metals in Sediment (2002-2010)

St. Marys River, Ontario, Canada

FIGURE
10d

1690014264

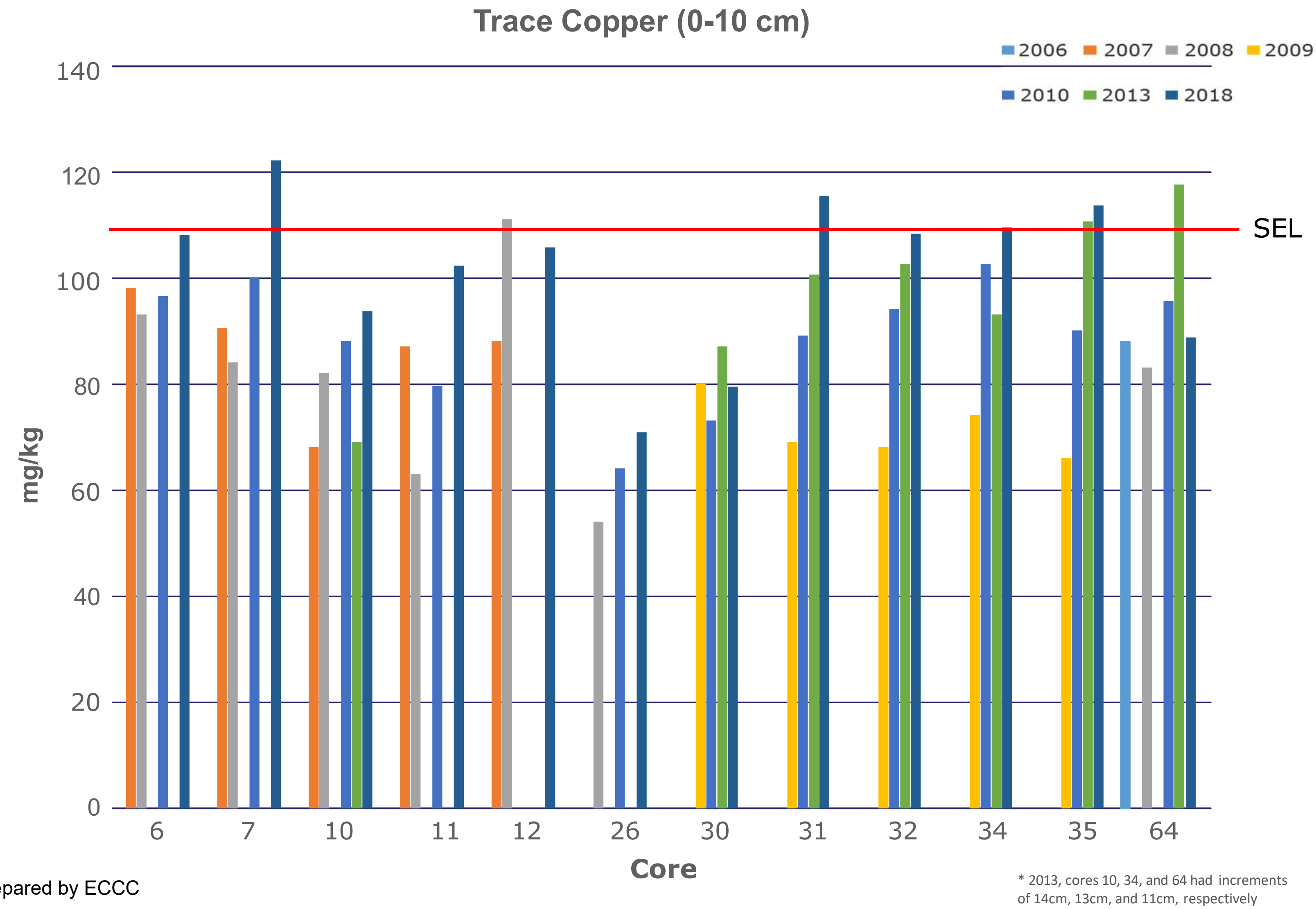


Figure prepared by ECCC

RAMBOLL

Temporal Trends in Metal Concentrations in EBMP Sediment (2006 – 2018)
St. Marys River, Ontario, Canada

Figure 11a

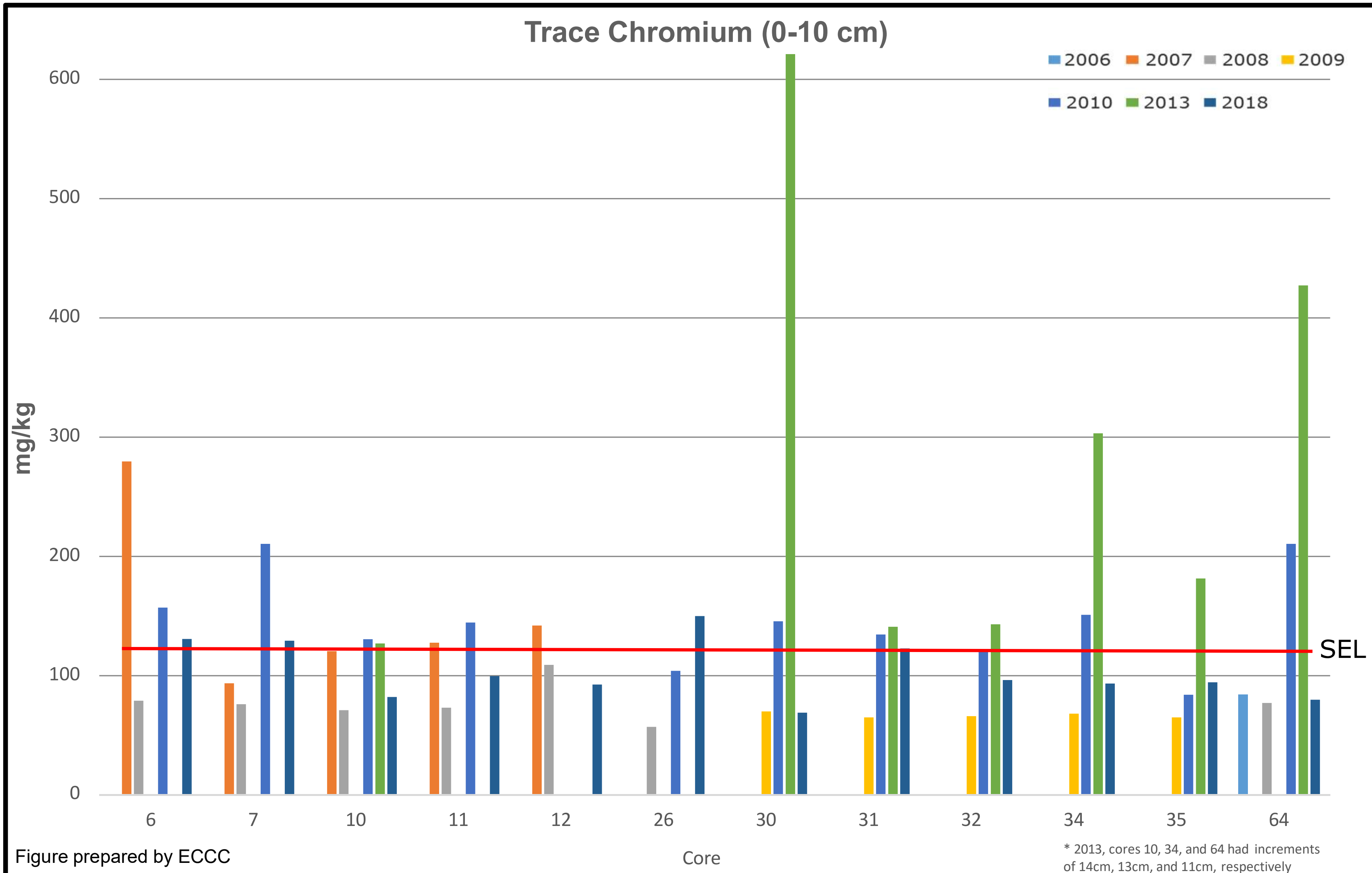


Figure prepared by ECCC

RAMBOLL	**Temporal Trends in Metal Concentrations in EBMP Sediment (2006 – 2018)** St. Marys River, Ontario, Canada	Figure 11b

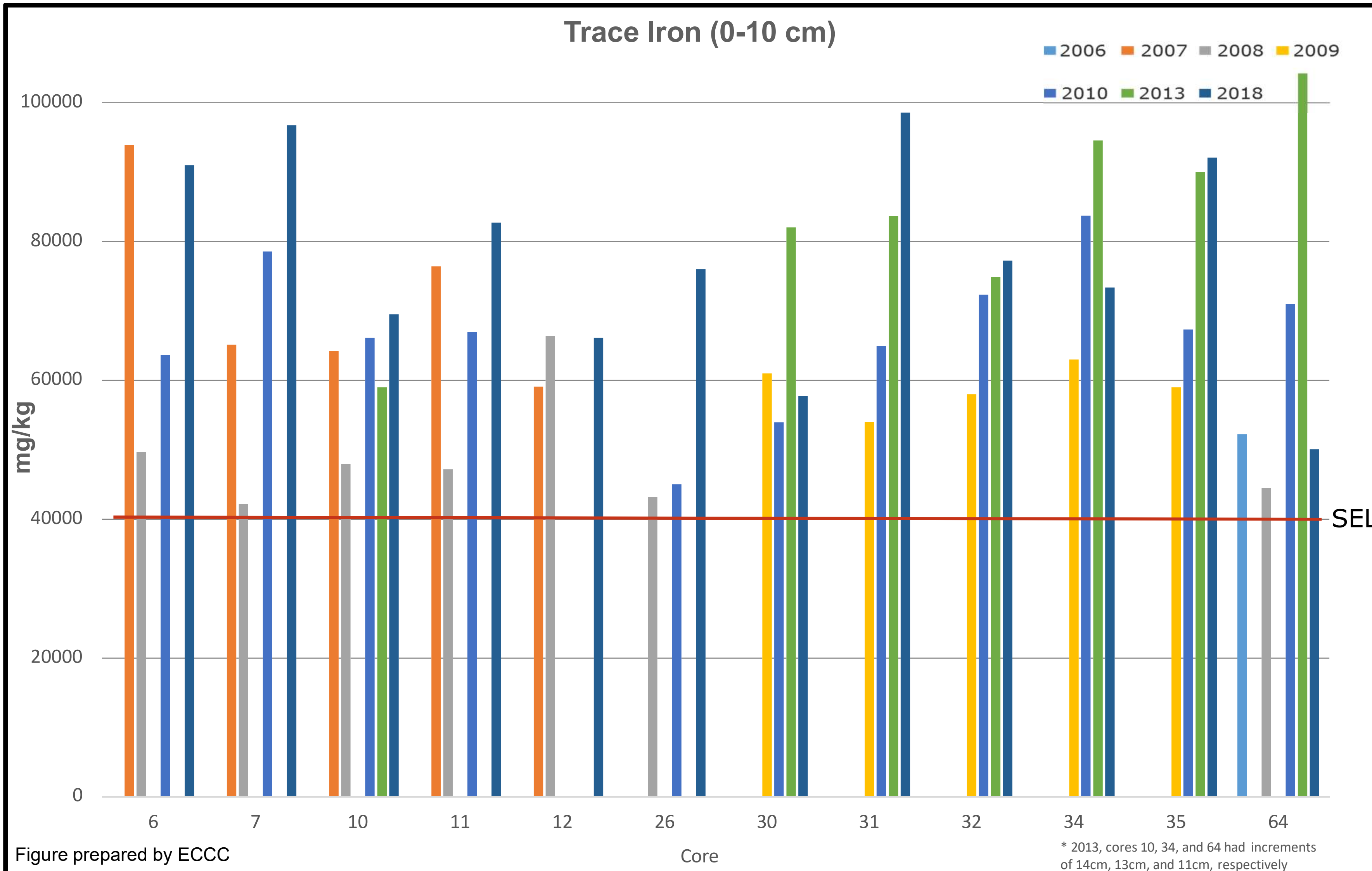


Figure prepared by ECCC

RAMBOLL	**Temporal Trends in Metal Concentrations in EBMP Sediment (2006 – 2018)** St. Marys River, Ontario, Canada	Figure 11c

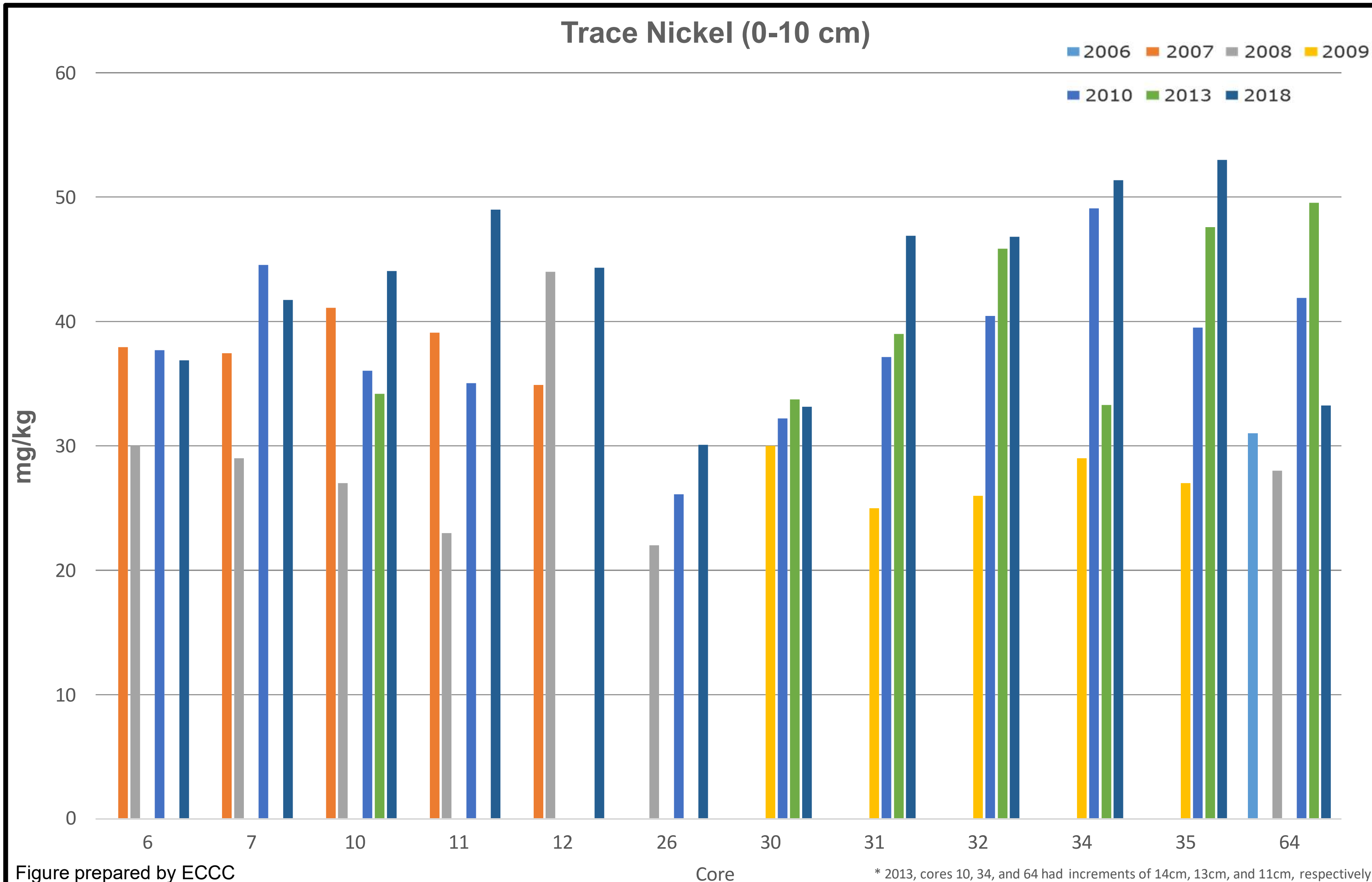


Figure prepared by ECCC

* 2013, cores 10, 34, and 64 had increments of 14cm, 13cm, and 11cm, respectively
* SEL for Nickel is 75 mg/kg

RAMBOLL

Temporal Trends in Metal Concentrations in EBMP Sediment (2006 – 2018)
St. Marys River, Ontario, Canada

Figure 11d

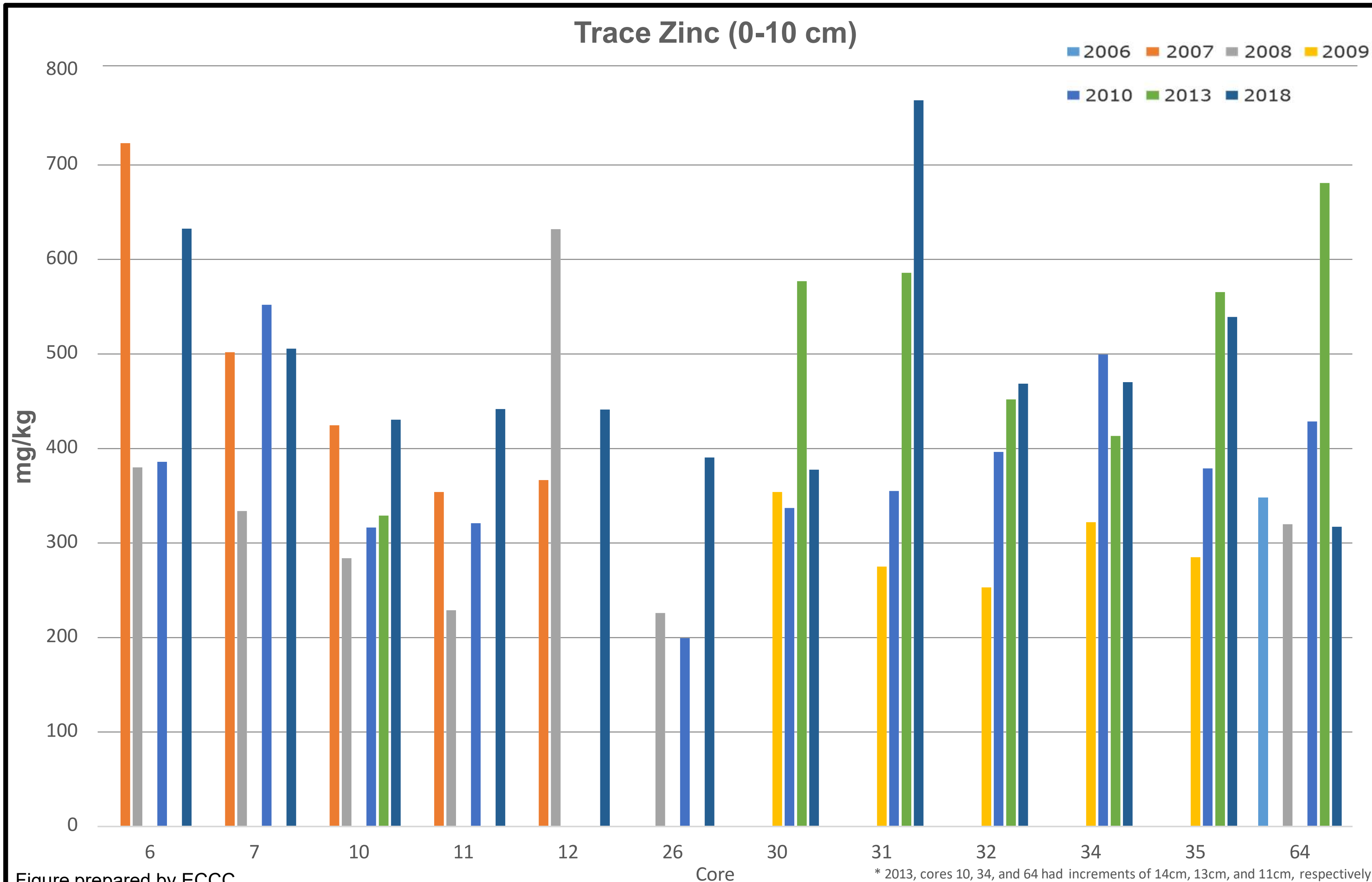


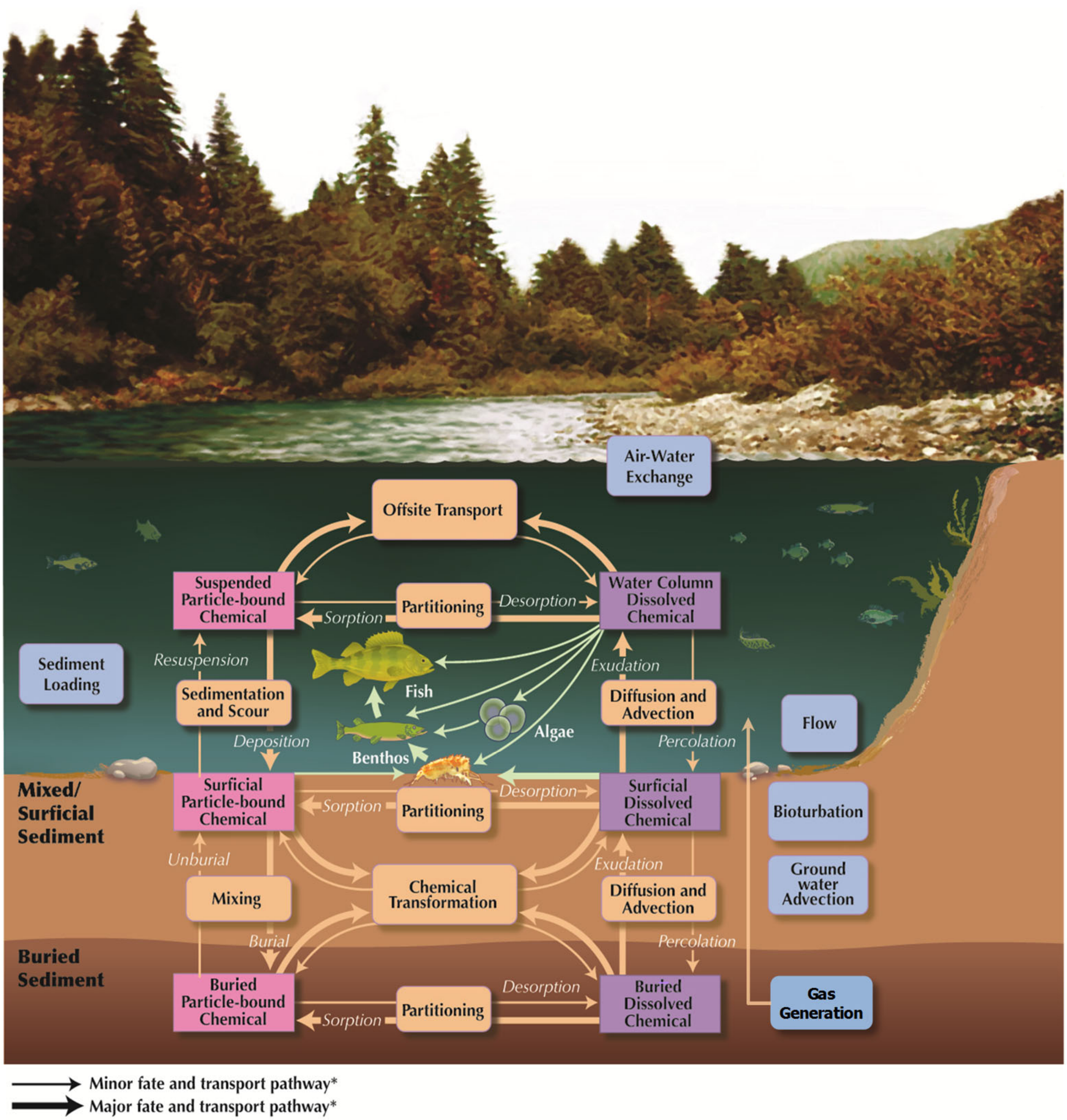
Figure prepared by ECCC

* 2013, cores 10, 34, and 64 had increments of 14cm, 13cm, and 11cm, respectively
* SEL for Zinc is 820 mg/kg

RAMBOLL

Temporal Trends in Metal Concentrations in EBMP Sediment (2006 – 2018)
St. Marys River, Ontario, Canada

Figure 11e



RAMBOLL

Fate and Transport Processes Affecting Contaminated Sediments

St. Marys River, Ontario, Canada

Figure 12

Topsail Island vicinity, April 22, 2011

Marina east of Topsail Island, April 12, 2011

Inside tip of Topsail Island facing west, April 12, 2011

Facing east of Shingwauk Island, April 12, 2011

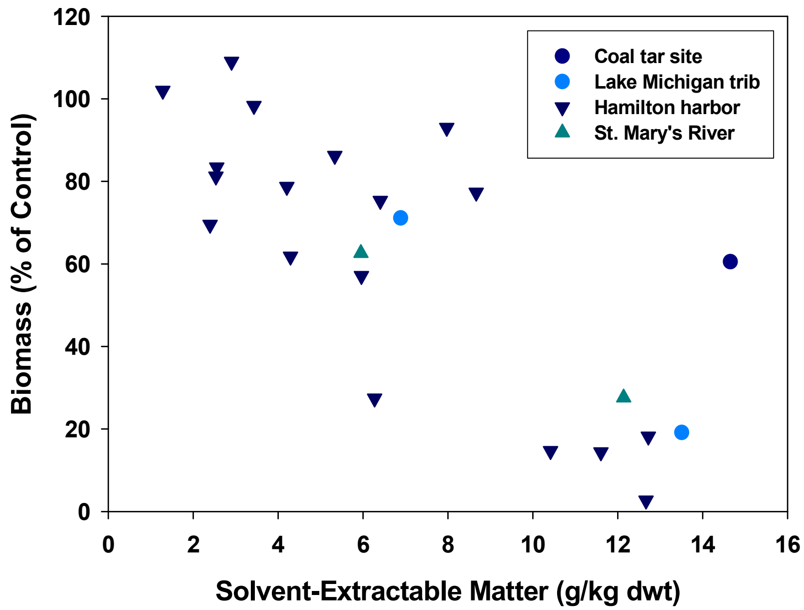
Photographs taken by Corrina Barrett, Sault Ste. Marie Innovation Centre.

RAMBØLL

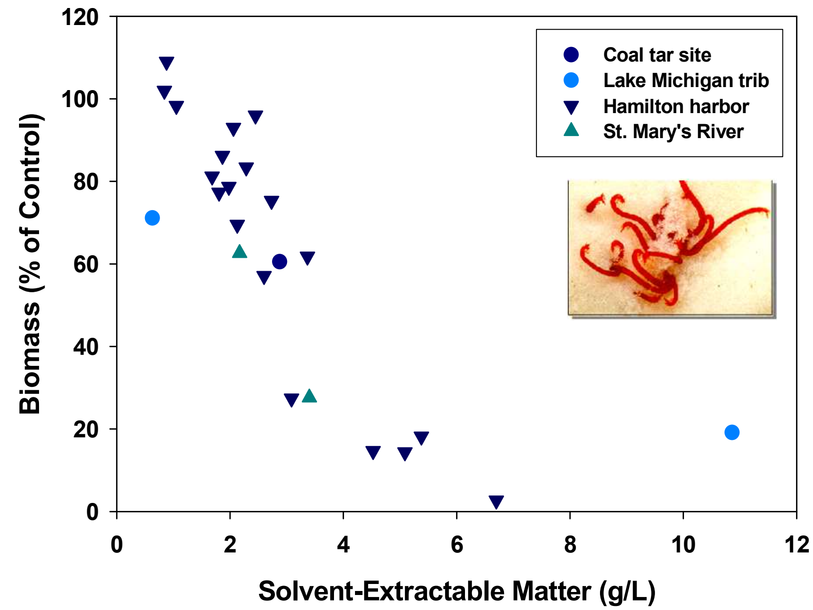
Observations of Ice Melt Conditions, EBMP Area, April 2011

Figure 13

Normalized to Dry Weight



Normalized to Volume



Relationship between oil & grease exposures and *Chironomus dilutus* biomass for St. Marys River sediments and sediments from other sites where toxicity due to physical effects of oil is suspected. Source: D. Mount (personal communication, 2011). Presented at Society for Environmental Toxicology and Chemistry Annual Meeting (Mount et al. 2009).

RAMBOLL

Relationship Between Oil and Grease Exposures and Midge Biomass for Multiple Sites
St. Marys River, Ontario, Canada

Figure 14

APPENDIX A
REVIEW OF 2018 EAST OF BELLEVUE MARINE PARK DATA & RELATED CASE STUDIES

1 Introduction

This appendix to the Conceptual Site Model (CSM) for St. Marys River sediments reviews data collected in 2018 to characterize sediment conditions in the East of Belleview Marine Park (EBMP) area (Milani and Grapentine 2019a,b), in the context of past site-specific investigations and relevant case studies from other contaminated sediment sites. This review is consistent with the Canada-Ontario Contaminated Sediments Decision-Making Framework developed under the 2002 Canada-Ontario Agreement (COA) Respecting the Great Lakes Ecosystem ("COA Framework"; ECCC and MECP 2008).

A key premise of the COA Framework is that "lines of evidence … that contradict the results of properly conducted field surveys with appropriate power to detect changes … 'are clearly incorrect' … to the extent that other [lines of evidence] are not indicative of adverse biological effects in the field." On the other hand, in cases where sediment chemistry and laboratory toxicity tests indicate toxicity to invertebrates but the benthic invertebrate community is not altered, the COA Framework identifies the assessment outcome as "Determine reason(s) for sediment toxicity." The specific rationale for this outcome is not spelled out in the Framework, but the implication is that a more detailed analysis is needed to verify that the benthic invertebrate community is in fact adequately protected under prevailing conditions.

Accordingly, the objectives of this review are:

- To identify likely and/or potential causes of toxicity observed in invertebrates exposed to EBMP sediments in the laboratory, to the extent possible based on available data
- To evaluate how the lack of apparent adverse effects on the EBMP benthic invertebrate community affects the need for additional information about causes of laboratory toxicity
- To develop recommendations for additional data needs, if any

The remainder of this appendix is organized according to the above objectives.

2 Potential Causes of Laboratory Toxicity

Below we provide a brief summary of the toxicity and chemistry data set (Section 2.1), followed by discussions of the potential role of petroleum hydrocarbons (Section 2.2) and other stressors and/or testing artifacts (Section 2.3).

2.1 Data Summary

In 2018, Environment and Climate Change Canada (ECCC) performed sediment toxicity testing using sediment collected from eight EBMP stations, four upstream reference stations within the St. Marys River, and two Great Lakes references stations. A subset of these stations were represented through side-by-side testing of sediment samples from two depth intervals: 0-5 centimeters (cm) and 0-10 cm. Toxicity was judged in comparison to an ECCC data set of toxicity test results from 27 Great Lakes reference sediment samples tested between 2011 and 2018. Sediments were considered nontoxic to a given endpoint if the test result was within two standard deviations of the Great Lakes reference mean and toxic if the test result deviated from the Great Lakes reference mean by more than three standard deviations; intermediate results were considered "potentially toxic."

The 2018 toxicity test results are summarized as follows:

- Mayfly nymphs (*Hexagenia* spp.) exhibited toxicity in 2 of 12 sediment samples representing 2 of 8 EBMP stations; both survival and growth were affected. Two additional EBMP samples were classified as potentially toxic, although the magnitude of effect was small (4% mortality). No reference sediment samples were toxic to *Hexagenia*.

- Sludge worms (*Tubifex tubifex*) exhibited toxicity or potential toxicity in 10 of 12 EBMP sediment samples representing 6 of 8 EBMP stations. The most sensitive endpoint was cocoon hatching success, leading to lower numbers of young per adult. No reference sediment samples were toxic to *Tubifex*.
- Amphipod (*Hyalella azteca*) mortality indicated toxicity or potential toxicity in 8 of 12 EBMP sediment samples and 4 of 10 references sediment samples, representing 5 of 8 EBMP stations and 3 of 6 reference stations. *Hyalella* growth was less sensitive than survival in these tests.
- Midge larvae (*Chironomus riparius*) exhibited toxicity in 1 of 12 EBMP sediment samples and 1 of 10 reference sediment samples, based on reduced survival. Four additional EBMP sample results were classified as potentially toxic based on an intermediate level of survival (76.7—80%). A total of 5 of 8 EBMP stations were thus classified as toxic or potentially toxic. None of the stations represented by sediment from two depth intervals showed agreement between results for both depths. No adverse effect on *Chironomus* growth was observed in any EBMP or reference sediment sample.

The sediment samples were analyzed for petroleum hydrocarbons, polycyclic aromatic hydrocarbons (PAHs), metals, nutrients (phosphorus and nitrogen), total organic carbon, and grain size distribution. Additionally, during the toxicity tests, overlying water quality was monitored by measuring temperature, conductivity, dissolved oxygen, pH, and ammonia. Physico-chemical data were reported for the EBMP and upstream reference samples but not for the Great Lakes reference samples.

2.2 Petroleum Hydrocarbons

Historically, petroleum hydrocarbons have been identified as the primary cause of toxicity in St. Marys River sediments, as described in Section 5.1 of the main CSM report. Petroleum hydrocarbons can affect benthic invertebrates through several mechanisms. While PAHs are the most toxic component of petroleum hydrocarbons, other hydrocarbons can also exert toxicity through the same nonpolar narcosis mechanism as PAHs, if they are present at sufficiently high concentrations (Redman et al. 2017). Additionally, petroleum hydrocarbons can affect organisms through physical oiling of gills and membranes (Verbruggen 2004, Pettigrove and Hoffman 2005); this physical effect was identified as a potential cause of toxicity during a toxicity identification evaluation of St. Marys River sediments (Milani et al. 2008). Petroleum hydrocarbons can also interact with PAHs to promote higher PAH bioavailability (and thus toxicity) than would otherwise be predicted from bulk sediment PAH concentrations (Mount et al. 2009); this effect can be detected by measuring freely dissolved PAH concentrations in sediment porewater. Although porewater PAHs were not measured in the 2018 sediment samples, such analyses were performed on 20 sediment samples collected in 2010. At that time, the PAH concentrations in sediment porewater were too low to be a primary cause of sediment toxicity (Hawthorne 2010; USEPA 2003).

The 2018 toxicity test results for *Hexagenia* and *Tubifex* are generally consistent with effects due to petroleum hydrocarbons. Measured concentrations of hydrocarbons are compared to *Hexagenia* survival in Figure 1.[1] The survival endpoint is shown because the growth endpoint was more variable in the

[1] Petroleum hydrocarbons are difficult to characterize analytically due to the extreme diversity of petroleum compounds, as well as potential interference from natural hydrocarbon compounds. In the 2018 St. Marys River sediment samples, hydrocarbons are reported as fractions 2 through 4, based on increasing number of carbons. Fraction 4 was measured chemically and was also measured gravimetrically, where the gravimetric analysis was intended to capture compounds with more than 50 carbons. Such heavy compounds are associated with solid

Hexagenia tests, and more samples were identified as toxic or potentially toxic based on survival than based on growth. The samples with the three highest hydrocarbon concentrations encompass both samples that were toxic to *Hexagenia* and one of the two samples that were potentially toxic to *Hexagenia*. Thus, consistent with historical investigations, petroleum hydrocarbons are a likely cause of the few cases to *Hexagenia* toxicity observed in EBMP sediment samples in 2018.

For *Tubifex*, most EBMP sediment samples were toxic or potentially toxic. The two nontoxic EMBP samples had the two lowest reported hydrocarbon concentrations (Figure 2). However, among the toxic samples, there is no further correlation between hydrocarbon exposure and magnitude of effect. When physical oiling is the mechanism of effect, stronger correlations between exposure and effect may be obtained by expressing hydrocarbon concentrations on a volume-normalized basis (Mount et al. 2009). However, volume normalization cannot be implemented for the 2018 data because sediment density was not measured. Confounding factors may also have played a role in the results, as discussed below for *Hyalella* and *Chironomus.* Despite these limitations, it seems reasonable to identify petroleum hydrocarbons as a likely cause of or contributor to the observed *Tubifex* toxicity.

2.3 Other Factors Potentially Contributing to Observed Toxicity Test Results

The toxicity test results for *Hyalella* and *Chironomus* are not well explained based on petroleum hydrocarbons or any other contaminants measured in the St. Marys River sediment samples. Also, for these test species, toxicity (or some other type of adverse effect) was observed in sediments from both EBMP and reference areas. In cases where a given EBMP or reference station was tested using sediment from two depth intervals, the *Hyalella* results were generally more similar within each station than among stations. This pattern was not evident for *Chironomus*, but the overall range of responses (i.e., percent mortality) was much less for *Chironomus* than for *Hyalella*, which would make such a pattern harder to discern. These observations suggest that the test organisms were apparently responding to some characteristic(s) of the tested sediments, but not necessarily to EBMP-related sediment contaminants.

As discussed below, we hypothesize that factors contributing to the observed results may have included the following:

- Stress related to low overlying water pH observed in certain *Hyalella* tests
- Increased bioavailability of metals related to sample handling, aeration, and decreased overlying water pH
- Competition with or predation by native organisms in the sediment samples
- Unmeasured contaminants such as legacy or current-use pesticides

As noted by Milani et al. (2019a), overlying water pH declined during the *Hyalella* tests for some samples, with pH falling below 6.0 in some cases. Figure 3 shows the relationship between minimum overlying water pH and *Hyalella* survival. The four samples with minimum pH less than 6.0 had particularly low survival, although other cases of low survival were not associated with low pH. The reason for the observed pH changes is not known, but it could have been related to changes in metal geochemistry within the samples. Specifically, formation of iron and manganese oxides and oxyhydroxides following oxidation of previously anoxic sediment would be expected to reduce overlying

asphalt or pitch-like substances and would not be expected to cause physical oiling. The characteristics of the chemically measured fraction 4 compounds are uncertain (could be solid or oily). Therefore, for purposes of evaluating relationships between toxicity and chemistry, we consider both the sum of fractions 2 through 4 and the sum of fractions 2 and 3 only.

water pH. Low pH could have directly affected the test organisms, and it could also have increased the bioavailability of metals in the sediment samples (Simpson and Batley 2007, Costello et al. 2015).

Further, the same redox changes that are hypothesized as affecting pH can also directly affect metal bioavailability apart from the pH changes. Metal bioavailability has been shown to increase when sediments with stable vertical redox gradients become mixed during sample handling (i.e., during collection, sieving, and homogenization) and/or are oxygenated through aeration during toxicity testing (Costello et al. 2019). The bioavailability of metals can be low under both oxygenated and anoxic conditions, due to sorption to iron and manganese oxides and oxyhydroxides under oxygenated conditions and precipitation of insoluble metal sulfides under anoxic conditions. When redox gradients are disrupted, metal bioavailability can increase temporarily, as the time required to reestablish stable redox gradients and metal sequestration can be on the order of weeks to months (Costello et al. 2015, 2016, 2019).

The hypothesis of elevated metal bioavailability is supported by overlying water analyses conducted by Parrott et al. (2018). In that study, sediment collected from St. Marys River in 2016—including samples from some of the same EBMP stations evaluated in 2018—was tested for toxicity based on *Hyalella* survival, growth, and reproduction. Metals were measured in overlying water from the toxicity test exposure chambers, as an indicator of metal bioavailability. In the water overlying sediment from EBMP station EC64, the concentration of cadmium was reportedly greater than the 25% lethal concentration (LC25) for *Hyalella*, and the zinc concentration was 40% of the respective LC25. Comparable analyses were not performed for the 2018 sediment samples. No adverse effects on *Hyalella* survival were observed in the 2016 EBMP samples, but in the absence of metal bioavailability data for the 2018 samples, it is not possible to determine whether exposures were greater in 2018 or whether only *Hyalella* responses to the sediment exposures were greater.

Elevated exposures to bioavailable metals in St. Marys River sediments seem more likely to be a testing artifact than an accurate representation of field conditions, because (1) unlike petroleum hydrocarbons, metals have not been identified as a primary toxicant in this system over the last several decades, as discussed in Section 5.1 of the main CSM; (2) benthic invertebrate community composition does not appear adversely affected compared to reference conditions; and (3) the mechanism of such a laboratory artifact has been demonstrated, as described above. Nonetheless, if metals are readily released from the sediment under toxicity test conditions, it is reasonable to expect that localized sediment disturbances (e.g., related to boat traffic) could also cause release of metals. However, the extent and rapidity of dilution by overlying water would be far greater within the river than in a toxicity test chamber, such that occasional minor disturbances may have little impact on the benthic community.

In the 2018 sediment samples, sieving was attempted to remove native organisms before toxicity testing, but organic matter in the sediment would not pass through the sieve, so the sieving process was not completed. Although sieving (and presumably attempted sieving) disrupts redox gradients, it is often implemented because native organisms can adversely affect toxicity test organisms through competition and/or predation. For example, Reynoldson et al. (1994) found that survival and growth of *Hyalella*, *Chironomus*, and *Hexagenia* decreased with increasing numbers of worms added to toxicity test sediments. Native worms are a major component of the St. Marys River benthic community in EBMP and reference areas (Milani et al. 2019b) and were present in the toxicity test samples (D. Milani, personal communication). The moderate degree of mortality in the *Chironomus* tests is consistent with effects observed by Reynoldson et al. (1994) related to presence of native organisms. Additionally, ENVIRON (2013) reported that in sediments collected from a Lake Erie tributary (Otter Creek, Toledo, Ohio), native planaria (flatworms) appeared to have eaten the *Chironomus* test organisms in certain toxicity test replicates. Based on the EBMP benthic invertebrate community characterization (Milani et al. 2019b), planaria were reported from several EBMP stations, though not any reference stations, and could potentially have been present in some toxicity test samples.

Lastly, it is possible that the toxicity tests were affected by unmeasured contaminants. The pyrethroid pesticide bifenthrin, in particular, has been identified as a widespread cause of sediment toxicity to *Hyalella.* Other pyrethroid pesticides and chlordanes have also been flagged as likely contributors to *Hyalella* toxicity in sediments collected over wide geographic areas (e.g., Nowell et al. 2016, Moran et al. 2017, 2020). *Hyalella* is generally more sensitive than *Chironomus* to current-use pesticides (Nowell et al. 2016). Although not related to persistent sediment contamination in EBMP, unmeasured pesticide exposures thus have the potential to confound sediment toxicity assessments with *Hyalella.*

3 Benthic Invertebrate Community Quality versus Laboratory Toxicity

The benthic invertebrate community in EBMP was characterized in 2018 for the same stations tested for sediment toxicity. Data collection involved collecting sediment samples, extracting and preserving benthic organisms found in the sediment, taxonomically identifying the organisms at the family level, and estimating the abundance of each family at each sampling station. Benthic community composition at the EBMP stations was compared to observations from local (upstream St. Marys River) and regional (North Channel and Georgian Bay) reference stations. The reference stations were matched to EBMP based on similar physical habitat conditions. Multivariate statistical analyses were conducted to determine whether benthic community composition was significantly different between EBMP stations and reference conditions. No significant differences were detected (Milani et al. 2019b).

Although toxicity observed in laboratory tests is often predictive of benthic community alterations in the field (e.g., Day et al. 1995), exceptions can occur for reasons such as the following:

- Sediments are not actually toxic under field conditions, and laboratory toxicity (or appearance of toxicity) was due to one or more study artifacts
- Sediments are toxic and benthos is altered, but the benthic community alteration was not detected
- Sediments are toxic, but there is a mismatch between toxicity test sensitivity and the sensitivity of the benthic community, such that toxicity is not manifested in the benthic community characteristics targeted for protection

Each of these explanations may have some applicability to the EBMP data interpretation. Potential explanations for the difference between toxicity and benthic community findings in the 2018 data set are discussed below (Section 3.1), followed by a discussion of related case studies (Section 3.2).

3.1 EBMP Evaluation

As discussed in Section 2.3, observed effects on *Hyalella* and *Chironomus* in toxicity tests did not appear to be due to EBMP-related sediment contaminants. Sediment handling and aeration procedures during toxicity testing may have artificially increased sediment toxicity to metal-sensitive species (especially *Hyalella*) by lowering pH and increasing metal bioavailability. Also, interactions between toxicity test organisms and native organisms in the sediment samples may have created the appearance of toxicity to both test species, when effects were actually caused by competition or predation.

Toxicity to *Hexagenia* and *Tubifex* was associated with hydrocarbon exposure and thus seems more likely to be representative of effects potentially occurring within the EBMP area. The mayfly *Hexagenia* is a member of the family Ephemeridae, which was a major component of the benthic invertebrate community at all four upstream St. Marys River reference stations. However, Ephemerid mayflies were less abundant or absent at most of the North Channel and Georgian Bay reference stations. In the EBMP area, Ephemerid mayflies were present in numbers comparable to upstream reference locations at 4 of 8 stations and were absent from 3 of 8 stations, with intermediate abundance observed at the remaining EBMP location. The location associated with the greatest adverse effect on *Hexagenia* (CS6, 16% mortality and growth <10% of the Great Lakes reference mean) had no Ephemerid mayflies in the paired benthic community sample. This result could reflect sediment toxicity, or it could reflect natural variation in Ephemerid mayfly abundance. Even if mayflies in the field are adversely affected at this location, any toxicity was not pronounced enough in 2018 to be evident at the level of the overall benthic invertebrate community.

Tubifex is a member of the family Naididae, one of the two most abundant invertebrate families throughout EMBP and the reference areas. There are several reasons that may explain why the sensitivity of *Tubifex* reproduction in toxicity tests is less than that of Naidid worms in the field. For example, other Naidid species may be less sensitive to petroleum hydrocarbons than *Tubifex*. In that case, the toxicity test endpoint would be more sensitive than the community attribute targeted for protection, since worm diversity at the species level is not likely to be a societally valued ecosystem attribute. Another possible explanation is that local Naidid worms may be genetically or phenotypically adapted to tolerate petroleum hydrocarbons better than laboratory-cultured *Tubifex*. An additional possible explanation lies in density-dependent limitations to population growth, whereby most offspring of an abundant and highly fecund species may not reach maturity regardless of any toxicity (REF). These possible explanations fall in the category of sensitivity mismatches between toxicity test and benthic community assessment endpoints.

The species selected for toxicity testing are intended not only as representatives of their taxonomic families but as sensitive indicators of potential effects on a wider variety of benthic invertebrate species. Some jurisdictions place a high value on the prevalence of pollution-sensitive species versus pollution-tolerant species (e.g., REFs). ECCC's approach to assessing benthic invertebrate community quality focuses instead on the overall family-level similarity to the community composition that could be expected in the absence of sediment contamination. This approach implicitly gives a higher value to the overall ecological function of the benthic community, rather than the presence or absence of individual species. Thus, the assessment endpoint applied for benthic invertebrate community quality allows for a minor degree of community alteration to be considered acceptable.

3.2 Case Studies

Three case studies are reviewed for contaminated sediment sites in or near the Great Lakes region. Each case study has publicly available data indicating significant sediment toxicity in the absence of significant benthic community alteration. The case studies include:

- Lyons Creek East, located in Welland, Ontario (Milani et al. 2013)
- St. Lawrence River and tributaries in Massena, New York (Duffy et al. 2016)
- Hudson River in Hudson, New York (Azzolina et al. 2015)

A fourth case study, for Lake Champlain in Burlington, Vermont (Watzin et al. 1997), was also considered but is not discussed further in this appendix because the researchers used assessment methods that are now outdated and had a relatively high likelihood of failing to detect benthic invertebrate community alterations. The case studies reviewed here are intended to help determine the degree to which contaminated sediment conditions at EBMP are generally consistent with those in other systems and ways in which they differ. That context, in turn, can help inform decisions related to remediation and further study.

3.2.1 Lyons Creek East

Lyons Creek East is a moderate-sized (fourth-order) stream located in the Niagara River watershed. Sediment quality in the stream was first investigated due to a known polychlorinated biphenyl (PCB) spill from a hydro-transformer. The study area was located downstream from an urban area (Welland) and adjacent to a former pipe mill. Contaminants of interest included PCBs, PAHs, metals (especially zinc), and pesticides. Assessment methods were similar to those employed for the St. Marys River, as the sediment investigation was conducted in accordance with the COA Framework (Milani et al. 2013).

Severe toxicity was observed at three sampling stations, with test organism survival as low as 2% for *Hexagenia*, 35% for *Tubifex*, 35% for *Hyalella*, and 39% for *Chironomus*. Despite the severe toxicity observed in laboratory tests, overall benthic invertebrate community composition at these three stations was not significantly different than at the designated reference stations. However, one of the three stations with toxic sediment was devoid of caddisflies, mayflies, and amphipods and had the lowest taxonomic diversity of the 21 stations investigated. Because sediment management was required to address PCB bioaccumulation regardless of benthic community alterations, it was not necessary to formally determine whether the observation of sensitive species' absence from one station constituted an unacceptable impact. The observed loss of multiple sensitive taxa from the most impacted station—which was more pronounced than potential effects discussed above for EBMP—underscores that multivariate statistical analysis of overall benthic community composition is not designed to account for moderate to subtle effects related to species' sensitivity.

Milani et al. (2013) identified possible reasons for the discrepancy in severity or occurrence of benthic community alterations versus toxicity in Lyons Creek East sediment; definitive determinations could not be reached in part due to limited analyses of contaminant bioavailability. Increased bioavailability of organic contaminants related to sediment sieving and homogenization was one possible explanation, and potential toxicity due to current-use pesticides was another (Milani et al. 2013). Additionally, in light of recent research on effects of sediment handling on metal bioavailability (Costello et al. 2019), it seems likely that the bioavailability and toxicity of zinc was greater in the toxicity tests than in the field. With a maximum zinc concentration of nearly 8,000 mg/kg, metal concentrations in Lyons Creek East were much higher than in the St. Marys River. Thus, although Lyons Creek East was potentially more impacted, the pattern of significant sediment toxicity versus minimal benthic alterations and the possible explanations for the observed discrepancy are similar between the Lyons Creek East investigation and the 2018 EBMP investigation.

3.2.2 St. Lawrence River and Tributaries

Duffy et al. (2016) investigated sediment toxicity and benthic invertebrate community condition in the St. Lawrence River at Massena Area of Concern (AOC), to supporting delisting decision-making for the benthic invertebrate beneficial use impairment (BUI). Sediment chemistry was not investigated for this assessment, but chemicals of concern in the AOC include PCBs, mercury, pesticides, metals and excess nutrients.[2] Sediment toxicity was evaluated based on 20-day laboratory tests of *Chironomus dilutus* survival and growth. For the benthic community assessment, organisms were identified to the lowest practical taxonomic level, generally genus or species. Benthic results were analyzed using multivariate statistical analysis to evaluate community similarity (analogous to the ECCC approach applied in the St. Marys River), as well as using a multi-metric index (Biological Assessment Profile or BAP) that accounts for species-specific pollution tolerance ratings and taxonomic diversity.

[2] https://www.epa.gov/great-lakes-aocs/st-lawrence-river-area-concern-massenaakwesasne

Chironomus dilutus survival in the St. Lawrence River study was lower across the board (i.e., in AOC, reference, and control sediments) compared to that observed for *Chironomus riparius* in St. Marys River sediments or Great Lakes reference sediments evaluated by Milani et al. (2019a). *Chironomus* survival in sediment from AOC stations within the Raquette River (a St. Lawrence River tributary) was significantly lower than in reference sediment from the same tributary. However, this result was attributed to a particularly high result for *Chironomus* survival for one reference station. Differences in benthic invertebrate community composition were observed but were attributed to habitat differences (especially sediment grain size distribution) rather than sediment toxicity or station classification as AOC versus reference. Duffy et al. (2016) concluded that the study results supported removal of the benthic invertebrate BUI for this AOC, although the AOC website indicates the BUI has not yet been removed.

The potential to find statistically significant toxicity due to especially high reference performance rather than a biologically significant effect has some limited relevance to the 2018 EBMP data interpretation. Specifically, the Great Lakes reference data for *Hexagenia* and *Tubifex* survival indicates nearly 100% survival with low variability, such that even relatively low mortality rates (<10%) are flagged as toxic (Milani et al. 2019a). However, the few EBMP sediment samples that were flagged as toxic based on *Hexagenia* or *Tubifex* survival also adversely affected sublethal endpoints in the same species, such that this phenomenon has minimal effect on the overall data interpretation.

3.2.3 Hudson River

Azzolina et al. (2015) assessed sediment quality in the Hudson River adjacent to a former manufactured gas plant (MGP) in Hudson, New York. Contaminants of concern were PAHs and coal tar in the form of non-aqueous phase liquid (NAPL). Sediment chemistry, toxicity, and benthic invertebrate community condition were investigated at stations adjacent to the MGP site as well as upstream, downstream, and cross-river reference stations. Sediment toxicity was assessed using 28-day tests measuring *Hyalella azteca* survival and biomass. Benthic invertebrate community condition was evaluated using a multivariate similarity assessment and a suite of metrics related to characteristics such as taxonomic diversity, prevalence of pollution-sensitive or tolerant species, similarity to a generic model of reference conditions, and relative abundance of various functional and taxonomic groups.

Severe to moderate toxicity was observed in 8 of 32 site sediment samples, with median *Hyalella* survival rates of 0% among the toxic samples and 100% among all other samples. Consistent with prior studies at MGP sites (McDonough et al. 2010), sediment toxicity was strongly correlated with porewater PAH exposures. Despite the severity of observed toxicity, the only significant difference in the benthic community between toxic site stations and (nontoxic) reference stations was the relative abundance of predatory species. On the other hand, comparisons of toxic versus nontoxic stations within the site data set (i.e., excluding reference stations) yielded seven significant differences for a variety of metrics, including overall taxonomic richness and a multi-metric index similar to that applied by Duffy et al. (2016).

Azzolina et al. (2015) identified certain characteristics of benthic communities in large rivers that tend to impede detection of benthic community alterations, namely a lack of sensitive species, a limited number of species common to many locations, and small-scale heterogeneity among reference locations. However, the fact that benthic community differences could be detected between toxic and nontoxic samples within the MGP site area also points to challenges in the selection of appropriate reference stations. For example, the site and reference locations in this study were not closely matched with respect to water depth.

Some of the challenges faced by Azzolina et al. (2015) do not apply to the 2018 St. Marys River data set. Specifically, St. Marys River site and reference stations were carefully matched with respect to habitat conditions (including water depth), and sensitive species such as mayflies and caddisflies were found at the applicable reference stations. Nevertheless, like the Lyons Creek East case study, the Hudson River MGP case study underscores that there are limitations in the ability of benthic community investigations to detect moderate to subtle community alterations.

4 Recommendations

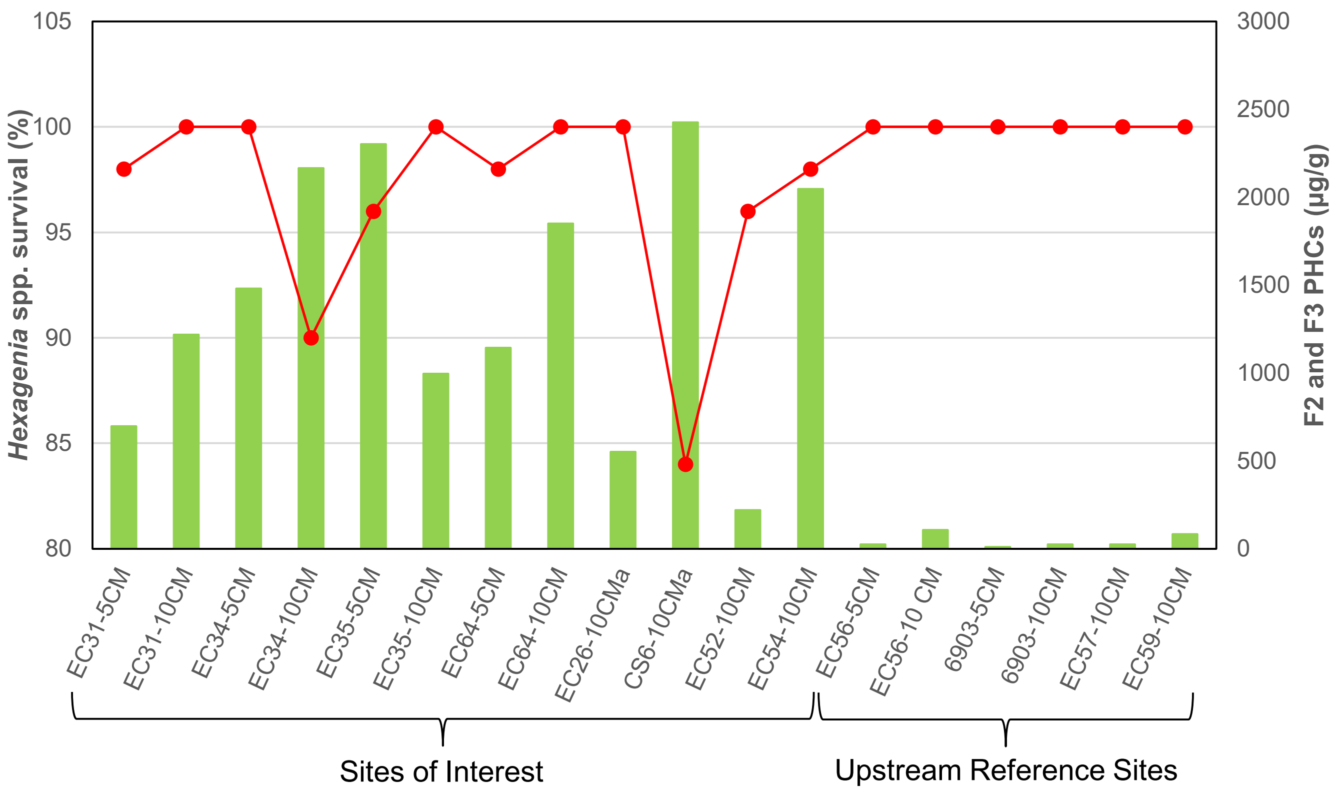
Based on the information reviewed in the main CSM and the 2018 data reviewed in this appendix, it is clear that sediment contamination and benthic invertebrate community conditions in the EBMP area have continued to recover over time. Recovery mechanisms likely include natural processes such as biodegradation and burial of contamination through deposition of cleaner incoming sediments. As of 2018, benthic community composition is similar to what would be expected in the absence of sediment contamination, although the possibility of moderate to subtle effects related to petroleum hydrocarbons at some EBMP locations cannot be fully ruled out.

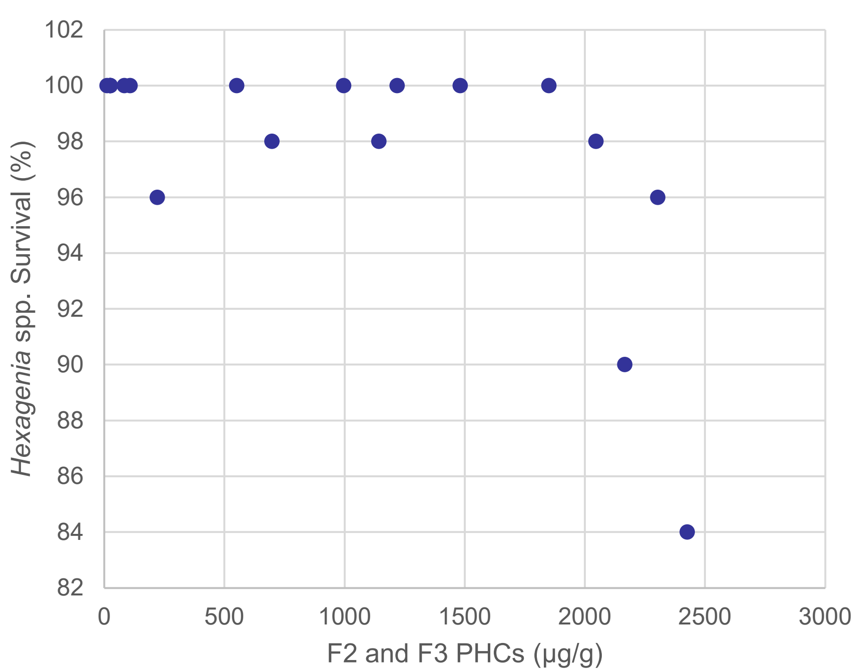
Based on these findings, together with the risk evaluations presented in the main CSM for fish, wildlife, and humans, we conclude that the available data are adequate to support an analysis of sediment management options for EBMP. Given the status of benthic community recovery, invasive sediment management measures such as dredging or capping do not appear to be warranted. However, past and current causes of sediment toxicity are sufficiently well understood to support planning activities such as monitoring natural recovery and safeguarding against unintended exposure of buried contaminants. For these purposes, it is not necessary to define precisely whether benthic community recovery as of 2018 was complete versus nearly complete.

To the extent that future monitoring efforts include sediment toxicity testing, several adjustments to the 2018 methods would be beneficial. A larger volume of overlying water should be used to lessen potential pH-related effects, as noted by Milani et al. (2019a). Analyses to clarify contaminant bioavailability would be particularly useful for interpreting toxicity test results. Specifically, sediment density could be measured to permit volume normalization of petroleum hydrocarbon concentrations. Metal bioavailability could be measured through one or more available techniques, such as porewater and/or overlying water analyses, test organism tissue analyses, measurement of acid volatile sulfide and simultaneously extracted metals, or possibly passive sampling techniques (currently an area of active research for metals). Additionally, at least some samples could potentially be evaluated through side-by-side tests with and without sediment sieving to remove native organisms. Such an approach might promote better understanding of sample handling artifacts with respect to contaminant bioavailability, as well as interactions between test organisms and native organisms. It might also be desirable in this context to collect and transport sediments in a manner that preserves intact redox gradients, to further minimize changes in metal bioavailability. Lastly, analyses of current-use pesticides could potentially clarify interpretation of toxicity test results, especially for *Hyalella*.

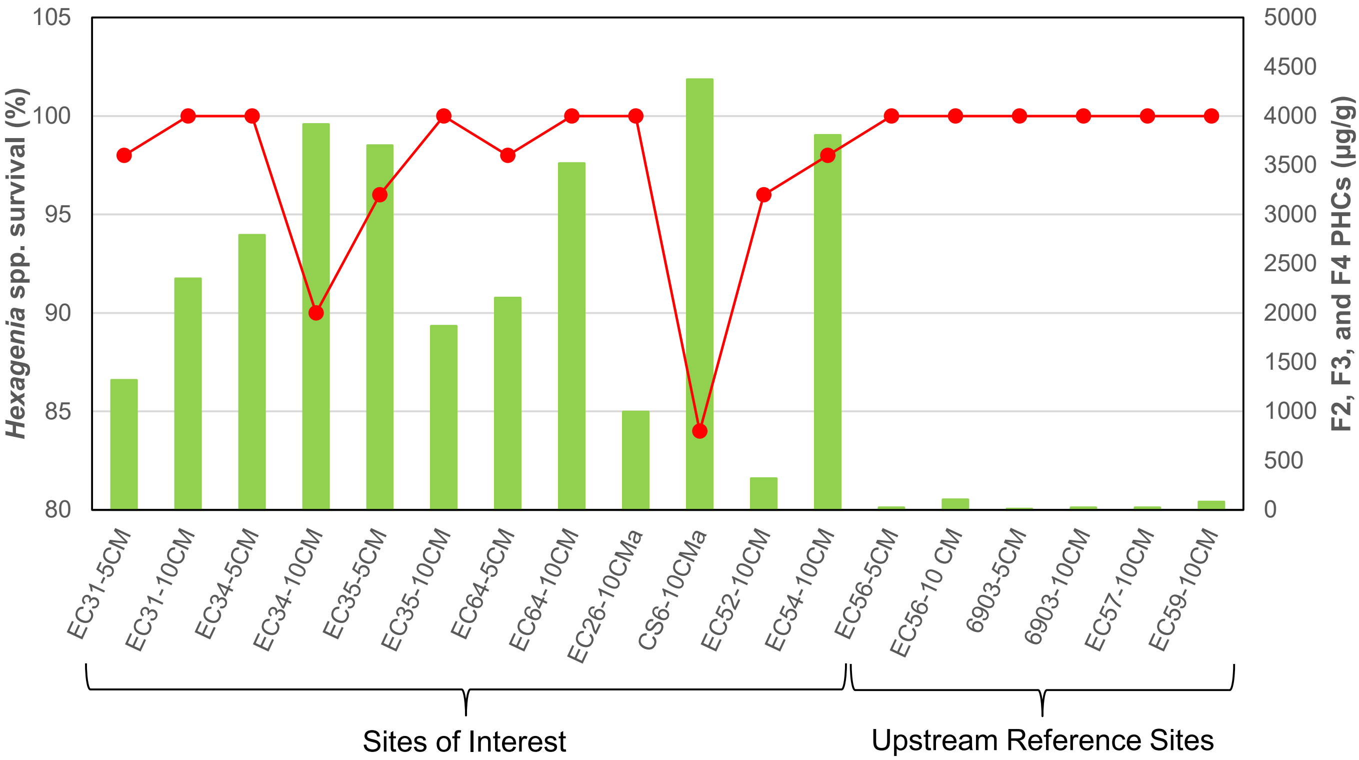
5 References

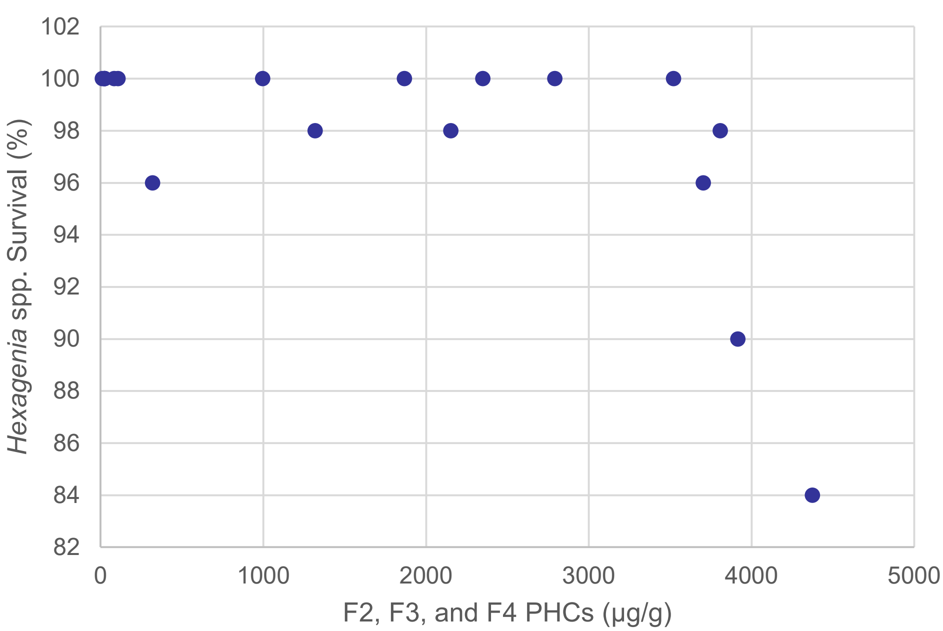
TO BE ADDED



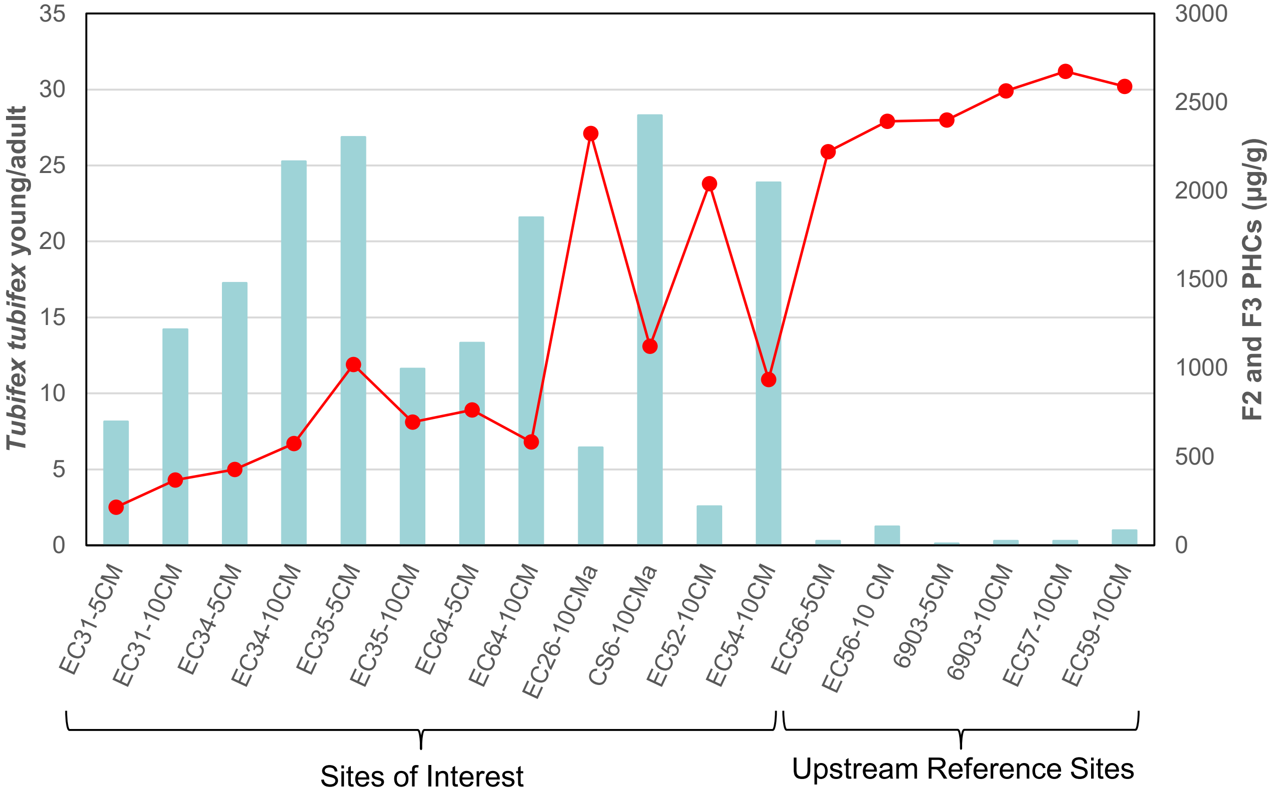


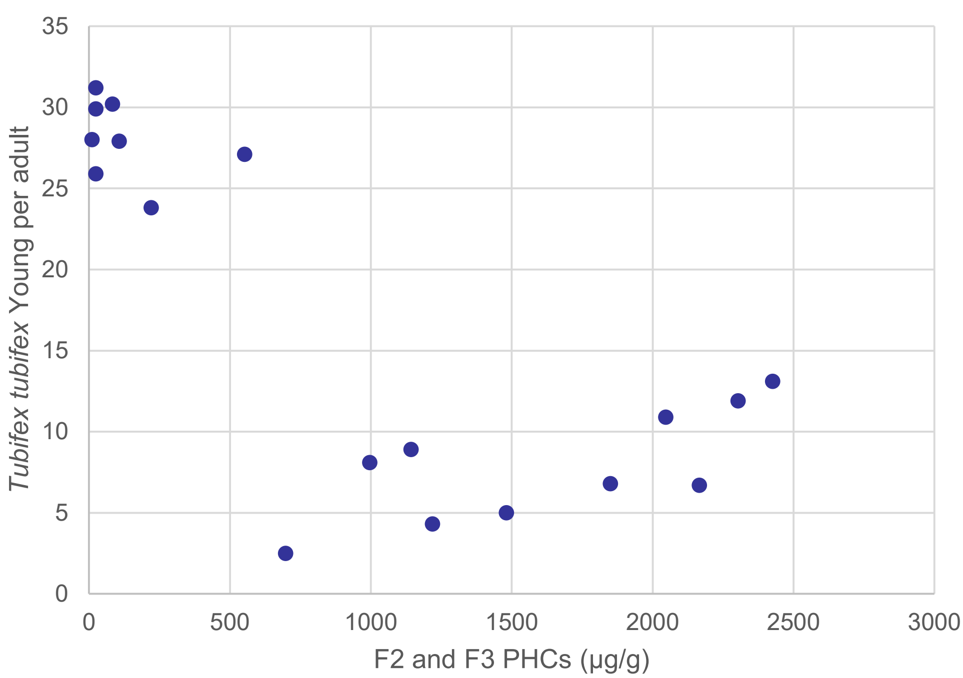
RAMBOLL	**Relationship Between Petroleum Hydrocarbon Concentrations and *Hexagenia* Survival (Fractions F2 & F3)** St. Marys River, Ontario, Canada	Figure A-1a





RAMBOLL	**Relationship Between Petroleum Hydrocarbon Concentrations and *Hexagenia* Survival (Fractions F2, F3, and F4)** St. Marys River, Ontario, Canada	Figure A-1b

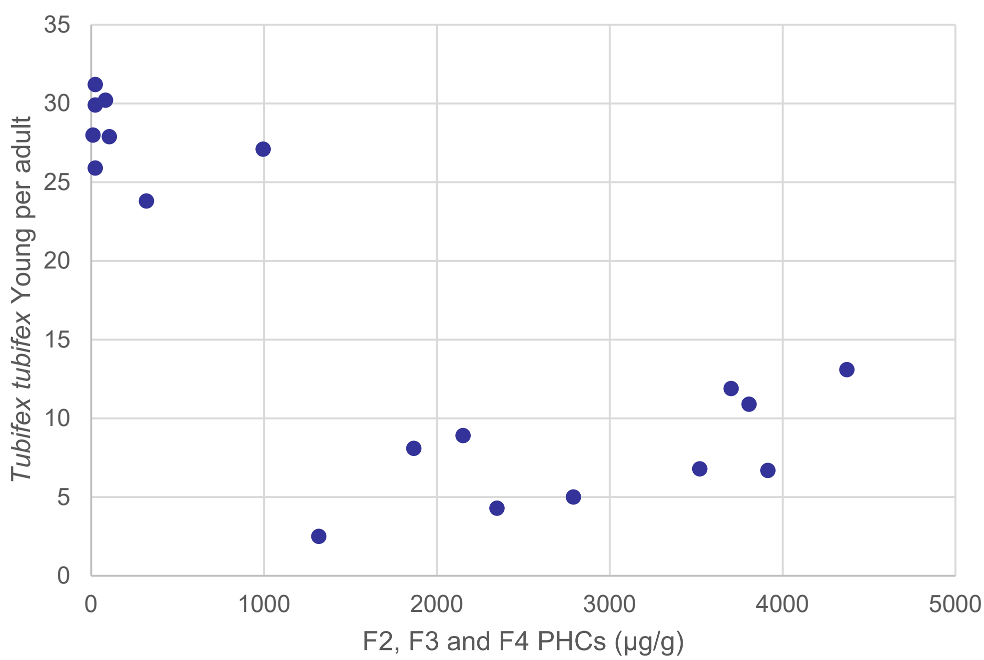




RAMBOLL

Relationship Between Petroleum Hydrocarbon Concentrations and Tubifex Reproduction (Fractions F2 and F3)
St. Marys River, Ontario, Canada

Figure A-2a

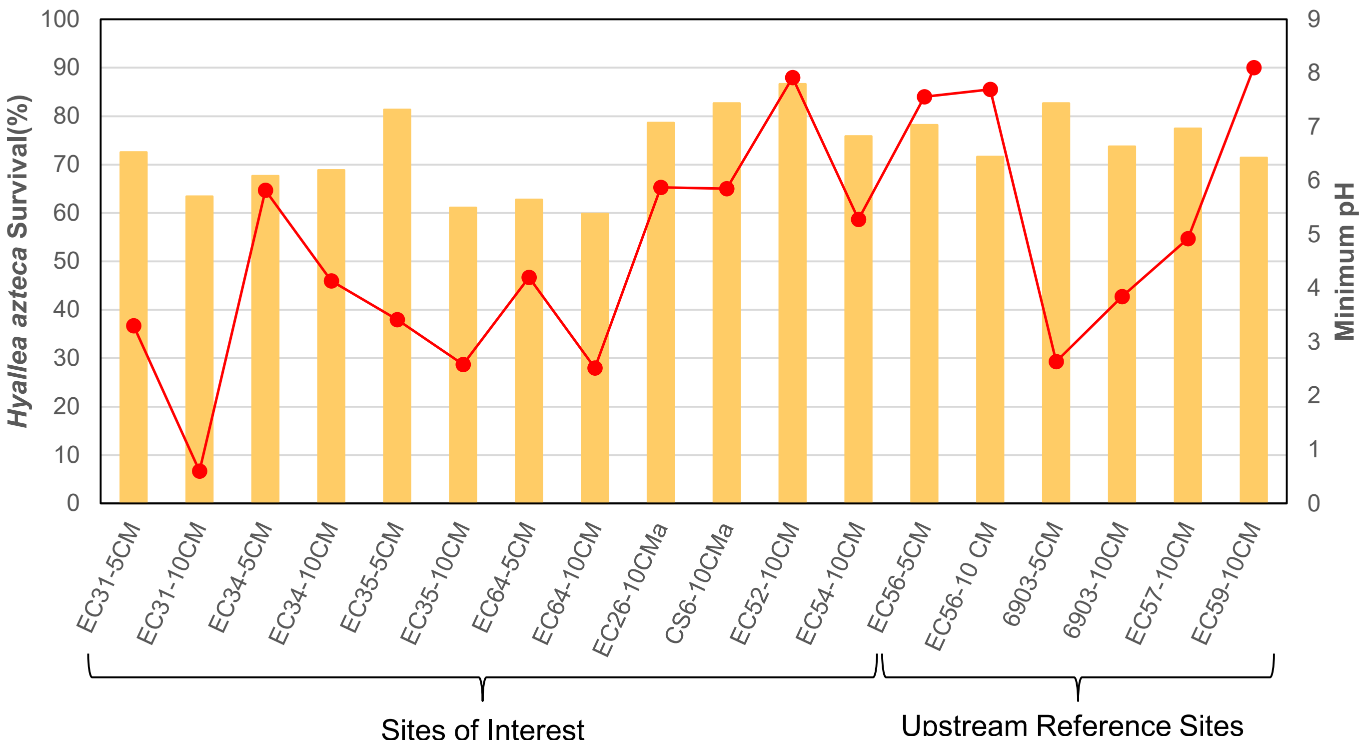


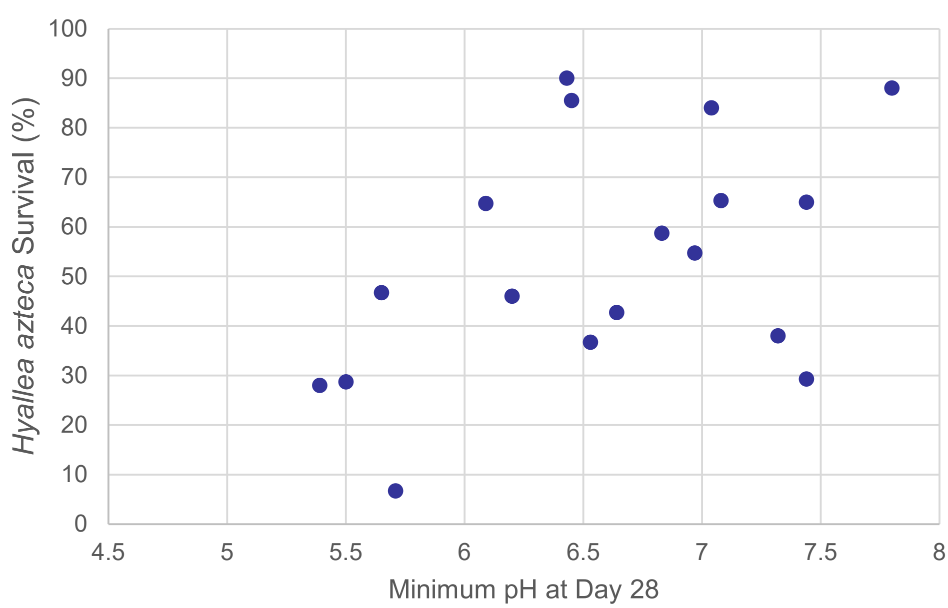
RAMBOLL

Relationship Between Petroleum Hydrocarbon Concentrations and Tubifex Reproduction (Fractions F2, F3 and F4)

St. Marys River, Ontario, Canada

Figure A-2b





RAMBOLL	**Relationship Between Minimum Overlying Water pH and *Hyalella* Survival** St. Marys River, Ontario, Canada	Figure A-3