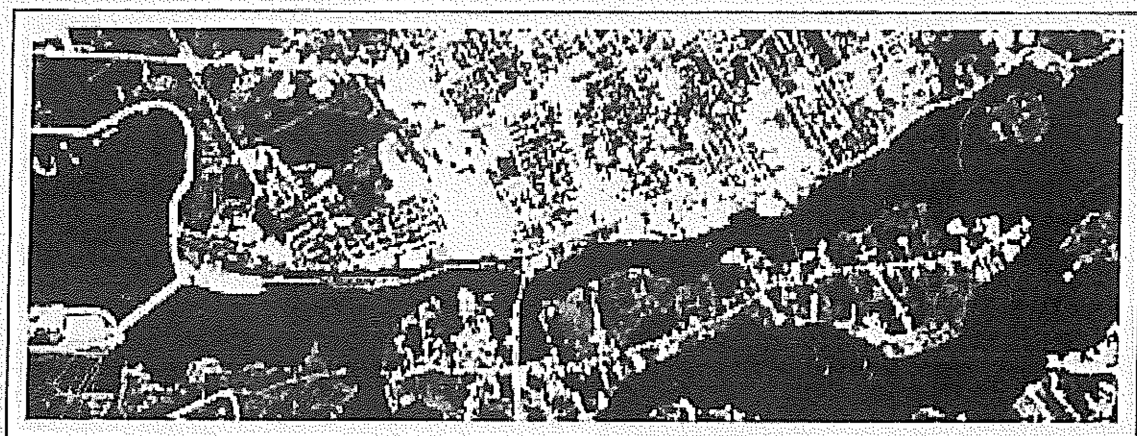
Cornwall, Ontario Waterfront Sediment

Review of Environmental Studies from 1970 to 1999



S. I. Dreier
Ontario Ministry of the Environment
Eastern Region

May 2000

Joint Project of

Ontario Ministry of the Environment
and
Environment Canada

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Page #	Section	Location	Correction / Addition
11	2.2.3.3 ZONE 3 – Tank Farm/Oil Tank Storage Area	text	Add: " **When winds are from the east, Zone 3 is a high energy area subject to wave action and ice scouring (Richard David, Mohawk Governments of Akwesasne, pers. comm.).**"
33-34	2.4.2 Contaminants in Suspended Matter	text	The contaminant concentrations provided are all dry weight concentrations (change "µg/g" to "µg/g **dry weight**").
		Table 23	Change units in column heading "Mean total Hg in suspended matter" from µg/g wet wt to µg/g **dry wt.**
51	2.6 Upstream Loading of Mercury	Table 28	In the 1995-1996 column: • mean mercury concentration (particulate) should read **218 ng/g** (not 218 µg/g). • annual mercury load (dissolved and particulate) at Moses-Saunders Dam should read **116 kg/year** (not 112 kg/day) • calculated upstream mercury loading to north channel at Cornwall should read **108.2 g/day** (not 104.3 g/d).
56	3.1 Mercury in Sport Fish	para. 4 & Table 31	Delete "M. Eckersley, MNR, pers. comm. cited in". The reference should read "Dreier et al. 1997".
		Table 31	Change value for the age of a 50 cm walleye in Lake St. Lawrence from 63 to **3-4** years.
55-68	3.1 Mercury in Sport Fish	text & Tables 33A, 33B	Change Lalonde (1999) to Lalonde (**1998**).
74-80	3.3.1 Toxicity & Bio-accumulation (Bedard 1999)	Table 35, 36, 37, 39 & 40	Bedard (1998) should read Bedard (**1999**).
		Table 38	To the table title, add "**Source: Bedard (1999).**"

Page #	Section	Location	Correction / Addition
		Table 39	End of second line of the table title should read "...associated biota-sediment accumulation **factor (BSAF)**".
91	3.3.5 Benthic Invertebrate Toxicity and Contaminants, 1991	Table 46	Reword column heading "Ratio of total mercury/TOC in sediment" to "**Total mercury/TOC ratio in sediment**" for clarity (the ratio uses sediment values of total mercury and TOC).
97	5. References		Change "Lalonde, J. 1999." to "Lalonde, J. **1998**."
n/a	Figure 103	Figure 103	Labels are missing for two stations (368A, 370A). Station 368A should be the red dot immediately inshore from and mostly overlapped by the dot for 368; station 370A is the unlabelled red dot shown inshore from 370.

CORNWALL WATERFRONT SEDIMENT: REVIEW OF ENVIRONMENTAL STUDIES FROM 1970 TO 1999

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Table 20 continued:

ENV.CAN STN #	AL u g/g	CD u g/g	CR u g/g	CU u g/g	FE u g/g	HG u g/g	MN u g/g	NI u g/g	PB u g/g	ZN u g/g	Total P u g/g	TOC %
CS171	18600	0.690	36.5	31.3	19100	0.44	359	20.6	29.6	118	918	2.016
CS172	31200	1.120	58.8	46.1	26600	0.62	439	34.1	44.0	350	1120	3.340
CS173	32700	1.210	58.4	46.1	28900	0.13	489	34.8	39.3	155	1150	3.130
CS175	29200	1.020	52.4	39.7	26500	0.08	464	30.4	37.5	136	1080	2.532
CS175-B	23000	1.380	48.5	36.4	21600	0.24	359	30.8	46.8	131	931	2.246
CS176	33300	1.150	59.1	46.3	30000	0.15	592	35.4	39.0	156	1180	3.099
CS176-S	27500	1.100	48.7	41.5	26200	0.15	665	38.4	29.5	136	1110	3.229
CS177	28600	0.976	49.5	38.1	26000	0.12	495	30.8	32.8	132	1020	2.599
CS179	28200	1.020	50.6	38.9	26500	0.14	488	31.5	35.2	136	1090	2.227
CS-179-B	20400	1.070	41.5	30.0	20300	0.16	344	26.9	36.2	108	909	1.945
CS-179-S	23900	1.040	42.9	34.0	22900	0.12	445	33.7	26.5	120	956	3.079
CS181	25500	1.010	47.9	35.6	24300	0.13	435	29.9	33.2	126	996	2.030
CS-181-B	22900	1.290	47.7	35.9	22300	0.20	366	31.2	43.3	124	952	2.141
CS-181-S	21700	0.916	39.7	30.6	20800	0.13	386	28.4	27.0	110	698	2.528
CS182	24300	0.901	43.8	31.8	23500	0.12	492	28.4	27.2	117	1090	2.507
CS-182-S	23500	0.939	42.4	33.4	23000	0.12	513	32.3	26.3	116	986	2.827
Lowest Effect Level		0.6	26	16	2%	0.2	460	16	31	120	600	1
Severe Effect Level **		10	110	110	4%	2	1100	75	250	820	2000	10

Table 20. Metal, total phosphorus and TOC concentrations (µg/g dry weight) in sediment samples (top 10 cm core sample) collected from the St. Lawrence River, 1997. (n=1). B - bottom 10 cm of the core. S - surface sample (top 3 cm). (Source: Richman 1999).

31.45 ppm   3.4 ppm   67.39 ppm   557 ppm

ENV.CAN STN #	AL ug/g	CD ug/g	CR ug/g	CU ug/g	FE ug/g	HG ug/g	MN ug/g	NI ug/g	PB ug/g	ZN ug/g	Total P ug/g	TOC %
CS105	19600	0.952	46.8	45.1	19300	1.67	295	26.8	63.3	600	761	2.926
CS109	16500	1.170	44.0	55.4	16600	**4.83 ****	252	26.2	136.0	759	638	2.872
CS109-S	19000	0.929	49.1	55.6	18600	**5.79 ****	317	26.1	156.0	673	1020	2.753
CS115	15400	1.450	41.8	57.9	16400	**3.66 ****	281	26.6	99.8	603	678	2.446
CS115-S	19400	1.080	43.7	57.6	19400	1.63	339	28.0	60.5	335	977	2.266
CS117	11500	0.608	31.4	34.9	13300	**2.00 ****	245	19.9	83.5	279	549	2.353
CS117-S	18700	0.869	40.6	42.8	18600	1.35	333	25.8	60.8	270	965	2.049
CS126	9630	0.720	23.8	26.6	12700	**5.20 ****	201	19.1	25.2	125	457	3.076
CS126-S	12200	0.424	28.5	21.8	18800	1.20	357	16.0	19.7	82	852	2.980
CS127	13500	0.473	27.8	29.4	13600	**4.32 ****	280	13.8	53.6	152	548	1.681
CS128	16800	0.637	32.7	33.6	17900	**11.20 ****	345	18.0	37.3	177	681	2.850
CS128-S	10300	0.359	25.1	21.1	15200	**3.44 ****	293	14.5	23.6	69	765	1.901
CS131	17000	0.645	33.3	42.5	18300	**19.50 ****	347	17.8	40.6	186	659	4.037
CS131-S	14900	0.758	32.0	37.6	14700	**14.70 ****	288	24.9	33.7	251	853	2.737
CS132	21800	0.949	43.8	60.9	17300	**6.78 ****	329	24.6	63.3	474	818	2.516
CS135	19000	0.984	40.3	45.5	18500	**3.75 ****	331	22.9	58.8	234	660	2.108
CS135-S	12300	0.624	27.2	19.3	14300	1.22	288	21.1	27.2	117	758	1.753
CS156	11400	0.490	26.2	16.9	14400	0.80	272	14.6	26.7	134	745	1.740
CS164	16000	0.888	48.7	59.9	17700	**3.09 ****	292	24.9	136.0	669	847	2.454
CS164-S	14800	0.795	38.4	47.3	15900	1.98	291	25.2	100.0	395	841	2.822
CS166	24400	1.220	47.5	43.2	23000	0.79	360	27.5	36.0	132	1150	3.236
CS166-B	27000	1.290	53.1	46.1	23500	0.12	328	32.8	44.2	142	1140	3.263
CS166-S	23400	1.120	45.3	43.4	22200	1.06	335	30.2	33.7	133	986	3.491
CS167	16800	0.910	37.0	39.7	16300	1.19	265	29.9	36.6	107	867	2.970
CS167-B	24700	1.420	44.0	64.4	15000	**4.63 ****	247	38.8	58.3	158	1120	6.745
CS167-S	12200	0.687	26.7	23.6	13700	1.72	239	19.5	21.9	76.9	653	2.370
CS168	17000	0.877	37.5	31.7	17000	1.71	290	24.0	38.0	102	947	1.813
CS168-B	25600	1.590	51.5	75.1	17000	**13.70 ****	286	35.9	73.6	187	1090	5.818
CS168-S	11800	0.754	26.8	21.2	13100	0.70	222	20.8	24.7	80.7	724	2.178
Lowest Effect Level	0.6	26	16	2%	0.2	460	16	31	120	600	1	
Severe Effect Level **	10	110	110	4%	2	1100	75	250	820	2000	10	

H-P 9.9 ppm   30.75 ppm

ESTIMATED MASS OF MERCURY, ZINC, COPPER AND LEAD IN ZONES 1, 2, 3 AND 4

The table below contains the estimated mass of mercury, zinc, copper and lead for each of zones 1, 2, 3 and 4.

The calculations are based on the following assumptions.

1. For **Zones 1, 2, 3 and 4** all of the metal concentrations are assumed to be in the finer sediments, i.e. mud and muddy sands. For this reason only the combined volume of mud and muddy sands was used in the calculations.

2. For **Zones 1, 2, 3 and 4**,the density of the mud and muddy sands is assumed to be **1.37 kg/l** (or 1370 Kg/m^3) and this density is assumed to be similar throughout the depth of finer sediment.

3. For **Zones 1, 2 and 4**,three concentrations (mean, min. and max) for each of the four elements were assumed for the entire mud and muddy sand deposits. These concentrations were derived on a zone by zone basis using all 1997 data which includes concentrations of metals in surface grab, core top and core bottom samples. (Note: no coring in Zone 2 was performed in 1997. Estimates of elemental mass in this zone are based on surface grab and top 10 cm samples and for this reason these estimates of elemental mass may be low. Additional work is needed to perform similar estimates of elemental mass based on the1994 data, which includes extensive coring).

4. For **Zone 3**, estimates of elemental mass were calculated using mean concentrations derived from samples collected in 1992, '93 and '99 and the maximum concentrations from all of the combined years (i.e., '92 to '99). (Note: the '99 samples were composite samples of a 20 cm deep swath of sediment as compared to the '92 and '93 samples which were 2-5 cm surface samples. For this reason the '99 sampling detected higher mercury concentrations).

Zone 1		
ELEMENT		**MASS in Kg**
MERCURY	mean	78
	min.	3
	Max	3741
ZINC	mean	3386
	min	2100
	max	5106
COPPER	mean	1179

[Hg] × (Vol mud/muddy sand) × Bulk density = Hg mass.

	min	579
	max	2051
LEAD	mean	1114
	min	598
	max.	2010
ZONE 2		
Element		**Mass in Kg**
MERCURY	mean	799
	min	130
	max	3173
ZINC	mean	53568
	min	11228
	max	123505
Copper	mean	6605
	min	2750
	max	9910
LEAD	Mean	10622
	min.	3206
	Max	25384
ZONE 3		
Element		**Mass in Kg**
MERCURY	'92 mean	1.6
	'93 mean	2.1
	'99 mean	29.7
	max.	72.4
Zinc	'92 mean	522
	'93 mean	538
	'99 mean	449
	max.	1316

Copper	'92 mean	200
	'93 mean	250
	'99 mean	148
	max.	1053
LEAD	'92 mean	176
	'93 mean	268
	'99 mean	132
	max.	1053
ZONE 4		
Element		**Mass in Kg**
MERCURY	mean	27.31
	min	15.61
	max	46.82
ZINC	mean	25123
	min	21067
	max	30431
Copper	mean	7221.55
	min	5852.15
	max	9032
LEAD	Mean	6728
	min.	5130
	Max	9129

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ACKNOWLEDGEMENTS

This report summarizes decades of research on the St. Lawrence River at Cornwall from the 1970s onwards. A dedicated group of researchers is responsible for the data reviewed herein. Their generous help throughout the writing of this report was greatly appreciated. In particular, I would like to thank Lisa Richman, Norm Rukavina, Serge Legage, Peter Nettleton, Al Hayton, Donna Bedard and Trefor Reynoldson. In addition, Norm Rukavina and Peter Nettleton generously provided full colour maps of their data. Janette Anderson, Hans Biberhofer, Conrad deBarros, Rein Jaagumagi, Keith Somers and Mike Whittle also provided valuable input to the report. Editorial comments from all of the above individuals as well as Heather Hawthorne and Barbara Theman were very much appreciated.

Mapping was done by Linda Hua and Rebecca Kay, Environment Canada, Geomatics Unit. Air photos (1991) were provided by Ontario Ministry of Natural Resources. Digital mosaic of photos and satellite imagery (including the cover image) was provided by Environment Canada.

1. INTRODUCTION

1.1 Purpose of the Review

The purpose of this review is to provide a state of the environment report on sediment in the St. Lawrence River at the Cornwall waterfront. It is meant to be used as a handbook for deciding on the best approach to managing areas of contaminated sediment at the waterfront. The report compiles and summarizes the results of numerous studies of sediment quality and related environmental conditions that have been completed in the north channel of the St. Lawrence River at Cornwall since the 1970s. Figure 1 shows the area under consideration, which stretches from the upstream (west) end of Cornwall Island to the west side of Pilon Island.

1.2 Background

Cornwall, Ontario has been a centre of industrial activity since around the turn of the 20th century. A cluster of industries developed here on the north shore of the St. Lawrence, taking advantage of the large volume of fresh water available for industrial processes and effluent dilution and dispersion. Three main industries operated plants at the Cornwall waterfront—a pulp and paper mill (1881-present), a chlor-alkali manufacturer (1935-1995) and a rayon manufacturer (1925-1992). The former two produced materials (pulp and sodium hydroxide) used in the rayon manufacturing process of the latter. Company names have varied somewhat over time but for clarity the following names are used throughout this document: Domtar (pulp & paper); ICI or ICI Forest Products (chlor-alkali); and Courtaulds or Courtaulds Fibres (rayon).

Over the decades of industrial operations at Cornwall, persistent toxic contaminants such as mercury that were discharged into the St. Lawrence River have accumulated in sediment along the Cornwall waterfront. The sediment also contains contaminants from non-point sources such as urban and rural surface runoff and contaminants of upstream origin transported from Lake Ontario in St. Lawrence River water and suspended solids. The presence of contaminated sediment in the Cornwall area has been documented since 1970, the date of the first sediment quality survey. Since then there have been many studies of sediment quality and related environmental conditions along the Cornwall waterfront and downstream into Lake St. Francis.

In 1985, the Cornwall-Massena section of the St. Lawrence River was designated a Great Lakes-St. Lawrence Area of Concern (AOC) by the International Joint Commission. Separate Remedial Action Plans (RAPs) were developed for Canadian and US areas of concern in this section of the St. Lawrence River. The Canadian AOC stretches from the Moses-Saunders Power Dam in the west, through the north channel of the river at Cornwall and downstream throughout the northern portion of Lake St. Francis to Valleyfield, Quebec.

Recommendations of the St. Lawrence River (Cornwall) RAP were documented in the Stage 2 Report which was released in 1997 (Dreier et al. 1997). The RAP was developed through an in-depth process of public, inter-agency and international consultation and includes recommendations for addressing problems of contaminated sediment. Recommendations #16 and #17 concern contaminated sediment in the Canadian section of the St. Lawrence River:

#16 In areas where contaminant levels in sediment are below the severe effect level but above the lowest effect level, implement source control measures to prevent further contamination of sediment and allow remediation of contaminated sediment to occur by means of burial by the natural sedimentation process.

#17 In areas where contaminant levels in sediment exceed the severe effect level for mercury, PCBs or other persistent toxic contaminants or where the sediment is found to be acutely toxic (i.e., the "hot spots"), prevent further contamination by implementing source control measures and remediate sediment by the most appropriate state-of-the-art technology (e.g., dredging, capping, *in situ* treatment).

In the 1990s, further contamination of sediment from major local point sources was stopped with the closure of the rayon and chlor-alkali plants and the installation of secondary treatment of wastewater at the pulp and paper plant. Serious consideration can therefore now be given to implementing the remainder of Recommendations #16 and #17 as appropriate.

The first step is to review and interpret as a whole the large body of technical information collected since 1970 in relation to sediment contamination at Cornwall. This handbook reviews the information to provide a framework for making decisions regarding appropriate management of contaminated sediment along the Cornwall waterfront.

2. WATER, SEDIMENT, AIR

2.1 Rates, Distribution and Direction of Flow

The St. Lawrence River flow rate immediately upstream of the Moses-Saunders Power Dam ranged from 5660-9905 m^3/s between 1960 and 1980 (MOE unpublished data). Mean annual flow was 7657 m^3/s over the 1985-1995 period (St. Lawrence River Committee on Gauging 1985-1995). In the period between July 15, 1998 and July 15, 1999 the flow ranged from 4690-10,240 m^3/s, with a mean flow of 7465 m^3/s (Rukavina 2000).

Below the dam, the river splits into two channels at the upstream end of Cornwall Island and rejoins into a single channel at the downstream end of Isle St. Regis. About one-third of the river flows through the north channel at Cornwall and two-thirds flows through the south channel near Massena as shown in Figure 2. Table 1 shows rates and distribution of flow.

Table 1. Rates and distribution of flow in the St. Lawrence River, Cornwall-Massena area.

Observed distribution of flow	Year observed	Calculated mean annual flow rate (m^3/s)	Reference for % flow measurements
N. channel 34.5% S. channel 65.6%	1989	N. channel 2642	Tsanis et al. (1991)
N. channel 30.2% S. channel 69.8%	1991	N. channel 2312	P. Nettleton, MOE (unpublished 1991 data)

1 Calculated using an observed ten year mean flow rate (1985-1995) of 7657 m^3/s (St. Lawrence River Committee on Gauging 1985-1995)

Current velocities and direction immediately offshore from Courtaulds were investigated in 1976. This was done to determine design criteria and location for new waste outfall facilities (diffusers) for the Courtaulds plant, which were installed in 1978. Using on-shore instrument stations, a motorboat in the river tracked and mapped the routes of flagged floats attached to large sheet metal vanes extending several feet below water. Winds were light at the time of observation, varying from negligible to 3-4 knots (5.6-7.4 km/h) from the west. The floats were started upstream of the Courtaulds property at varying distances from shore (Richards & Associates 1976).

According to Richards & Associates (1976),

> the float tracking showed up dramatically the fact that at the location of the present [shore-based] outfall pipes and for several hundred feet from shore there exists a quiescent or backwater zone with only very slight water movement. One float...which was started 150 feet from shore upstream of the outfall drifted slowly upstream and toward shore. Another..., at 275 feet from shore, moved downstream at the extremely slow rate of 0.15 feet per second, or about 1/10 of a mile per hour. At 600 feet from shore the velocity was still only 0.5 feet per second.

A "marked change in velocity at about 700 feet from shore" was explained by the presence of Windmill Point which "deflects the river current and creates a sheltered quiescent area along the Courtaulds waterfront". The report also notes that additional observations made in conditions of "strong east winds showed that the waste plume from the broken discharge pipes [i.e., the shore-based outfalls] extended several hundred feet upstream along the shoreline. Beyond the 600-foot zone, east winds would have no significant effect on currents" (Richards & Associates 1976).

1999 Flow Direction and Velocity Measurements

In 1999, further measurements of flow rates and directions through the water column at the Cornwall waterfront were done using state-of-the-art acoustic equipment. Environment Canada, National Water Research Institute staff collected data on currents in the St. Lawrence River adjacent to the former Courtaulds Fibres site (Figure 3) using an acoustic Doppler current profiler (ADCP). Internal processing by the ADCP resulted in estimates of current velocity within several one meter thick layers ("bins") in the water column from near the surface to near the bottom. A total of 98 data sets were collected through holes in the ice on February 16, 17 and 18, 1999. Each data set represents approximately 5 minutes of data.

The data were processed, screened for quality and averaged over time and over all valid depths. For each site, profiles of velocity (east/west, north/south, and vertical) were calculated by averaging all of the data ensembles within each bin to obtain mean current speeds, error velocities and associated standard deviations. Data collection, processing and results are described in detail in Coastal Ocean Associates (1999).

Visual examination of the processed velocity data (89 profiles) showed that vertical current was very weak. Horizontal currents were nearly uniform from top to bottom, so the depth averaged values are representative of the entire water column at each observation site. Depth and time averaged currents are mapped in Figure 4. The currents were rather weak (< 10 cm/s) and the weakest currents occurred near the shore. Moreover, flow was in the upstream direction in many instances as shown in Figure 4 (Coastal Ocean Associates 1999).

2.2 Bottom Sediment Type, Thickness and Stability

The physical properties and stability of Cornwall waterfront sediment were intensively studied for six years beginning in 1993 by N. Rukavina, Environment Canada, National Water Research Institute (NWRI). Information on the location, thickness, volume and stability of fine-grained sediments was obtained from a combination of acoustic mapping, sediment grab and core sampling, underwater television and diver observations, and acoustic monitoring of sediment stability (Rukavina 2000).

Areas of fine grained sediment were of primary interest because contaminants tend to be associated with the organic matter of fine, silty sediments. This type of sediment accumulates in depositional zones of the river where slower currents allow the fine particles to settle and remain on the bottom.

2.2.1 Acoustic Mapping of Bottom Sediment Type

Sediment type along the Cornwall waterfront is so variable that it has been difficult to delineate areas of contaminated sediment by means of sediment sampling alone. An impossibly detailed grid of sampling points would have been required to accurately map the areas of fine-grained sediment and investigate contaminant concentrations in these areas. From 1993-1998, Rukavina (2000) tested and refined an acoustic seabed classification system called RoxAnn™ which provided the high-density data required. The results of the RoxAnn mapping were used to design intensive sampling grids of selected depositional areas for sediment surveys done in 1994 (Richman 1996) and 1997 (Richman 1999).

Rukavina (2000) combined RoxAnn acoustic data with data from sediment sampling and underwater television observations to classify Cornwall waterfront sediment (top 3 cm) into one of 8 types: mud, muddy sand, sand, coarse sand, gravel, boulders/hard, weeds on soft, weeds on hard. The mapped area extended from just upstream of the Lamoureux Park boat launch downstream to the west side of Pilon Island, and from the north shore of the St. Lawrence River (the mainland) across to the north shore of Cornwall Island (Rukavina 2000). RoxAnn bottom sediment types are mapped in Figure 5. Six depositional zones were identified in the north channel of the St. Lawrence River (Figure 6 and Table 2). The two zones downstream of Pilon Island are not discussed in this report because they are located beyond the eastern end of the Cornwall waterfront area.

The RoxAnn system collected echo-sounder data on acoustic hardness and roughness and converted it into a real-time georeferenced map of bottom sediment type. Data were collected at 1-second intervals from a boat moving at 2-3 m/s, with over 450,000 data records of depth and bottom type logged between 1993 and 1998. Transducer specifications suggest that RoxAnn's footprint on the river bottom has a maximum diameter approximately equal to the water depth at which the reading is taken. RoxAnn averages the returns from the footprint and cannot discriminate smaller features which could cause heterogeneous sediments to be misrepresented (Rukavina 2000). Water depths within the depositional zones 1-4 are shown in Figures 9, 13, 18 and 22 respectively. A discussion of some limitations of RoxAnn and its difficulties with gassy sediments appears in Rukavina and Caddell (1997) and Rukavina (1998).

The system was calibrated by comparing the RoxAnn acoustic bottom types with particle size data from sediment samples and underwater television observations. Shipek grab samples were collected at 217 sites and cores at 132 sites. Over 3000 georeferenced bottom images were recorded by underwater television (Rukavina 2000). Sediment thickness data for 975 sites was obtained from underwater television records, cores and diver observations. Rukavina (2000) reported that "the accuracy of the RoxAnn classification...varied from site to site but was typically about 55% good, 35% fair and 20% poor. This was good agreement considering that it was based on comparison of samples and television data that had a much smaller footprint than the sounder."

Table 2. Zones of fine-grained sediment deposition in the north channel of the St. Lawrence River identified by Rukavina (2000) and shown in Figure 6.

Location of depositional zone	Zone # within this report
Lamoureux Park boat launch area	Zone 1
Windmill Point to Pilon Island, adjacent to the north shore	Zone 2
embayment immediately upstream of Windmill Point near the oil tank storage area	Zone 3
northeast shore of Cornwall Island	Zone 4
Farlingers Point (across from northeast side of Pilon Island) to Flanigans Point	not numbered
south shore of Pilon Island	not numbered

The distribution of RoxAnn acoustic bottom types across the study area and within individual depositional zones is discussed in Section 2.2.3 and shown in Figures 5 (entire study area), 10 (Zone 1), 14 (Zone 2), 19 (Zone 3) and 23 (Zone 4).

2.2.2 Grain Size Type, [illegible]

[illegible] 705 755 4316

George email team list

Sediment sampling data was a[illegible]
thickness in the more intensive[illegible]
(Rukavina 2000). These result[illegible]
volumes in each of the four de[illegible]
maps corresponded well with t[illegible]
was "not surprising given the r[illegible]
data" that was observed (Ruka[illegible]

GIS-computed polygons of gra[illegible]
and the individual depositional [illegible]
calculated from field data obtai[illegible]
sites; core samples from 132 s[illegible]
because a polygon may be ext[illegible]
site). The grain-size distribution [illegible] individual depositional zones show GIS polygons superimposed on the actual sampling locations or data points from which they were extrapolated.

GIS maps of sediment thickness were produced in a similar manner (Figures 8, 12, 17, 21, 25). Sediment thickness data (975 sites) was obtained by coring, diver probing and an underwater video system for recording sediment surface characteristics and measuring sediment thickness that was developed to calibrate the RoxAnn acoustic data on bottom sediment types.

The total volume of fine-grained sediments in the entire study area was estimated using the product of sediment area and average sediment thickness. The volumes of size classes for each of the four depositional zones were computed by superimposing size and thickness polygons in ARC/INFO (Rukavina 2000). Results of area and volume calculations are provided in Tables 4-17.

2.2.3 Individual Depositional Zones

Characteristics of the four fine-grained sediment deposits at the Cornwall waterfront are discussed below. Rukavina (2000) mapped bathymetry (water depth contours), RoxAnn acoustic bottom types, GIS grain-size distribution, GIS sediment thickness for the entire study area and each of four depositional zones within it. The maps are reproduced here (Figures 5, 7-25). Please note that because the maps for Zones 1 and 3 are based on a much lower density of sample points they are not as reliable as the Zone 2 and 4 maps and were produced to give a general idea of sediment distribution only.

For reference, Table 3 shows the figure numbering for Rukavina (2000) maps used in this report.

Table 3. Figure numbering in this report for Rukavina (2000) bathymetry and sediment maps.

Parameter mapped	Figure number in this report				
	Entire study area	Zone 1	Zone 2	Zone 3	Zone 4
bathymetry	-	9	13	18	22
RoxAnn acoustic bottom types	5	10	14	19	23
grain-size distribution (GIS)	7	11	16	20	24
sediment thickness (GIS)	8	12	17	21	25

2.2.3.1 ZONE 1 – Lamoureux Park Boat Launch area

Depositional Zone 1 consists of a "deep rectangular basin parallel to the shoreline and separated from it by an inshore shelf and shoal" (Rukavina 2000) as shown in Figure 9. The inshore shelf extends to a depth of about 4 m, beyond which the bottom falls steeply to basin depths of 8-11 m. The basin extends eastward beyond the area mapped and in the west it slopes upward into a "shoal area with water depths of less than 2 m" (Rukavina 2000).

The fine-grained sediments cover an area 500 m long and 100 m wide in water from 4-8 m deep, 50-100 m off shore (Figures 10 and 11). Sediment thickness is "mainly less than 10 cm but can exceed 70 cm in the area of fine sediments" (Rukavina 2000) (Figure 12).

As noted earlier, sediment maps of this deposit are based on a much lower density of sample points and are therefore not as reliable as the maps for Zones 2 and 4. Zone 1 maps were prepared to give a general idea of sediment types and thickness distributions. The data provided the best estimates currently available for areal coverage and volume in the boat launch area deposit (Tables 4-6).

Table 4. Areas of grain-size types, Zone 1 (Lamoureux Park boat launch), 1993-1998. Source: Rukavina (2000).

Grain-size type	Area (m^2)	Area (km^2)	Percent of total area
mud	27,190	0.0272	17.3
muddy sand	4,816	0.0048	3.1
sand	6,255	0.0063	4.0
hard/boulders/gravel	118,945	0.1189	75.7
Total (all grain-size types)	157, 206	0.1572	100.0

Table 5. Areal distribution by sediment thickness, Zone 1 (Lamoureux Park boat launch), 1993-1998. Source: Rukavina (2000).

Thickness (cm)	Area (m^2)	Area (km^2)	Percent of total area
0-10	110,422	0.1104	70.3
11-30	19,174	0.0192	12.2
31-50	8,825	0.0088	5.6
51-70	2,865	0.0029	1.8
>70	15,919	0.0159	10.1
Total	157,206	0.1572	100

Table 6. Grain-size volumes, Zone 1 (Lamoureux Park boat launch), 1993-1998. Source: Rukavina (2000).

Grain-size type	Area (m^2)	Average thickness (m)	Volume (m^3)	Percent of total
mud	27,190	0.63	17,027	69.7
muddy sand	4,816	0.60	2,904	11.9
sand	6,255	0.04	242	1.0
hard/boulders/gravel	118,945	0.04	4,247	17.4
Total	157,206	0.15	24,241	100.0

2.2.3.2 ZONE 2 – Windmill Point to Pilon Island

Zone 2 was the most intensively studied section of the river because it has the highest contaminant concentrations and was the site of a proposed demonstration sediment removal project (Richman 1996; Richman 1999; Rukavina 2000).

Figure 13 shows Zone 2 bathymetry based on all RoxAnn depth data collected between 1993 and 1998. An inshore shelf runs along most of the shoreline and extends about 4 m offshore. Beyond this the bottom slopes steeply out to the mid-channel where the water is 12-15 m deep. The fine-grained sediments are located on this slope between about the 4 and 11 m depth contours. They "form a ribbon-like deposit 2200 m long and 50-200 m wide on the slope between the inshore shelf and the main channel" (Rukavina 2000) (Figures 14 and 16).

Further inshore from the fine-grained deposit is an irregular bottom of mixed sediment types in water 3-9 m deep. Offshore from the deposit there is a rugged erosional channel where water depth varies by up to 2 m. Prominent shoals with an irregular bottom occur just east of Windmill Point near the shore and further offshore just west of Pilon Island (Rukavina 2000).

The pattern of sediment thickness in Zone 2 is highly variable (Figure 17). 44% of the sediment is less than 10 cm thick and only 10% is thicker than 50 cm. The thickest sediments occur "as small discrete areas scattered through the deposit" (Rukavina 2000).

Zone 2 was surveyed by RoxAnn mapping every October from 1993 to 1998. Figure 14 shows the average of RoxAnn bottom types in Zone 2 based on all of the data collected over the six year period. Six individual maps, one for each year of mapping, are shown in Figure 15 (Rukavina 2000).

The same basic pattern was found each year: a hard and weedy bottom by the shore, a ribbon-like deposit of mud and muddy sand at mid-depth, and sand and coarser sediments mid-channel at the offshore boundary of the mapped area. The earlier maps (1993-1995) are more variable than the later ones (1996-1998) which are very similar even in detail. This may be because survey equipment and procedures had stabilized by 1996. The sediment pattern for 1994 is much finer across the entire deposit than for other years. According to Rukavina (2000), "the increase in the area of muddy sediments and sands from 1993 to 1994 may be real. Flow rates peaked in 1993 at more than 10,000 m^3/s and then dropped to a low of less than 6,000 m^3/s in 1994. That should have resulted in flushing of finer sediment in 1993 and its recovery in 1994."

Estimates of area and volume for the various sediment types in Zone 2 are given in Tables 7-9 (Rukavina 2000).

Table 7. Areas of grain-size types, Zone 2 (Windmill Point to Pilon Island), 1993-1998. Source: Rukavina (2000).

Grain-size type	Area (m^2)	Area (km^2)	Percent of total
mud	110,814	0.1108	13.4
muddy sand	249,077	0.2491	30.1
sand	255,723	0.2557	30.9
hard/boulders/gravel	198,411	0.1984	24.0
weeds	12,583	0.0126	1.5
Total	826,606	0.8266	100.0

Table 8. Areal distribution by sediment thickness, Zone 2 (Windmill Point to Pilon Island), 1993-1998. Source: Rukavina (2000).

Thickness (cm)	Area (m^2)	Area (km^2)	Percent of total
0-10	373,930	0.3739	44.2
11-30	220,075	0.2201	26.0
31-50	170,074	0.1701	20.1
51-70	69,022	0.0690	8.2
>70	12,861	0.0129	1.5
Total	845,962	0.8460	100.0

Table 9. Grain-size volumes, Zone 2 (Windmill Point to Pilon Island), 1993-1998. Source: Rukavina (2000).

Grain-size type	Area (m^2)	Average thickness (m)	Volume (m^3)	Percent of total
mud	110,814	0.36	39,736	22.4
muddy sand	249,077	0.32	79,038	44.5
sand	255,723	0.18	46,805	26.4
hard/boulders/gravel	198,411	0.04	8,654	4.9
weeds	12,583	0.27	3,377	1.9
Total	826,606	0.21	177,610	100.0

2.2.3.3 ZONE 3 – Tank Farm/Oil Tank Storage area

Zone 3 (Figures 18, 19, 20, 21) has a "simple bathymetry consisting of a broad inshore shelf extending to 4 m [water depth], a steep slope from 4 to 7 m, and then an area of low relief between 7 and 9 m. Fine-grained sediments occur on the inshore shelf in depths generally less than 6 m. The deposit extends about 700 m along the shoreline and ranges in width from 100-200 m. Sediment thickness ranges from 10 to >70 cm in the north-central part of the area and is less than 10 cm elsewhere" (Rukavina 2000).

Sediment maps for this zone are based on a much lower density of sample points and are not as reliable as for Zones 2 and 4. They were used to obtain the best estimates currently available for areal coverage and volume (Tables 10-12).

When winds are from the east, Zone 3 is a high energy area subject to wave action + ice scouring (Richard David, Mohawk Government of Akwesasne, pers. comm.)

Table 10. Areas of grain-size types, Zone 3 (oil tank storage area embayment), 1993-1998. Source: Rukavina (2000).

Grain-size type	Area (m^2)	Area (km^2)	Percent of total
mud	2,038	0.0020	3.3
muddy sand	16,457	0.0165	26.7
sand	14,953	0.0150	24.2
hard/boulders/gravel	28,277	0.0283	45.8
Total	61,725	0.0617	100.0

Table 11. Areal distribution by sediment thickness, Zone 3 (oil tank storage area embayment), 1993-1998. Source: Rukavina (2000).

Thickness (cm)	Area (m^2)	Area (km^2)	Percent of total
0-10	39,117	0.0391	63.4
11-30	11,111	0.0111	18.0
31-50	7,785	0.0078	12.6
51-70	0	0.0000	0.0
>70	3,711	0.0037	6.0
Total	61,725	0.0617	100.0

Table 12. Grain-size volumes, Zone 3 (oil tank storage area embayment), 1993-1998. Source: Rukavina (2000).

Grain-size type	Area (m^2)	Average thickness (m)	Volume (m^3)	Percent of total
mud	2,038	0.32	650	5.7
muddy sand	16,457	0.25	4,153	36.4
sand	14,953	0.23	3,473	30.5
hard/boulders/gravel	28,277	0.11	3,125	27.4
Total	61,725	0.18	11,402	100.0

2.2.3.4 ZONE 4 – Northeast Cornwall Island

The bathymetry of Zone 4 is complex, with highly irregular depth contours (Figure 22). Rukavina (2000) writes: "the site is bounded on the south by a broad inshore shelf extending to depths of 4-6 m and then a steeper slope to depths of 13-16 m in a small s-shaped basin. A large shoal area with depths as low as 2 m is present at the eastern margin....The fine grained sediment is a basin deposit 1700 m long and 100-350 m wide extending from 50-100 m offshore at its western end to 100-600 m offshore at its eastern end [Figures 23 and 24]...Deposit depths are 7-8 inshore, increase to a maximum of 15-16 m along the basin axis and then decrease to 12-14 m at the offshore limit. Thickness is greatest along the axis of the basin (30->70 cm), least (0-10 cm) along the northeast margin and 10-30 cm thick through the balance of the area" (Figure 25).

Detailed areal and volume estimates for Zone 4 are provided in Tables 13-15.

Table 13. Areas of grain-size types, Zone 4 (northeast Cornwall Island), 1993-1998. Source: Rukavina (2000).

Grain-size type	Area (m^2)	Area (km^2)	Percent of total
mud	363,837	0.3638	53.5
muddy sand	116,516	0.1165	17.1
sand	117,881	0.1179	17.3
hard/boulders/gravel	81,804	0.0818	12.0
Total	680,037	0.6800	100.0

Table 14. Areal distribution by sediment thickness, Zone 4 (northeast Cornwall Island), 1993-1998. Source: Rukavina (2000).

Thickness (cm)	Area (m^2)	Area (km^2)	Percent of total
0-10	219,496	0.2195	32.3
11-30	106,066	0.1061	15.6
31-50	177,679	0.1777	26.1
51-70	158,717	0.1587	23.3
>70	18,079	0.0181	2.7
Total	680,037	0.6800	100.0

Table 15. Grain-size volumes, Zone 4 (northeast Cornwall Island), 1993-1998. Source: Rukavina (2000).

Grain-size type	Area (m^2)	Average thickness (m)	Volume (m^3)	Percent of total
mud	363,837	0.34	124,040	75.1
muddy sand	116,516	0.16	18,348	11.1
sand	117,881	0.14	16,799	10.2
hard/boulders/gravel	81,804	0.07	6,026	3.6
Total	680,037	0.24	165,212	100.0

2.2.3.5 All Zones Combined and Study Area as a Whole

Tables 16 lists area, thickness and volume estimates for all four depositional zones combined. The proportions of mud and muddy sand differ from the acoustic estimates in Table 17 "because of the non-uniform distribution of data points and the presence of gas in the muds which shifts their acoustic labels up to muddy sand" (Rukavina 2000). As mentioned earlier, limitations of RoxAnn with respect to gassy sediments are discussed in Rukavina and Caddell (1997) and Rukavina (1998).

Table 16. Total grain-size areas and volumes, Zones 1-4 combined. Source: Rukavina (2000).

Grain-size type	Area (m^2)	Average thickness (m)	Volume (m^3)	Percent of total
mud	503,880	0.36	181,453	47.9
muddy sand	386,865	0.27	104,443	27.6
sand	394,812	0.17	67,319	17.8
hard/boulders/gravel	427,437	0.05	22,052	5.8
weeds	12,583	0.27	3,377	0.9
Total	1,725,577	0.18	378,644	100.0

Table 17. GIS areas and volumes for acoustic types for entire study area shown in Figure 5. Source: Rukavina (2000).

RoxAnn acoustic bottom type	Area (m^2)	Area (km^2)	% total	Average thickness (m)	Volume (m^3)
mud	300,182	0.3000	6.0	0.50	150,091
muddy sand	850,356	0.8500	16.9	0.31	263,610
other	3,881,137	3.8811	77.1		
Total	5,031,675	5.0320	100.0		413,701

2.2.4 Sediment Stability

Studies of sediment stability in Zone 2 done in parallel with the sediment mapping indicated that the sediment deposit is stable (although there is also evidence of unstable episodes as discussed in Section 2.2.6) (Rukavina 2000).

Acoustic monitoring equipment was designed to measure the within-sediment depth of disturbance by currents and the likelihood of resuspension or erosion of contaminants. Measurements were made using an echo sounder placed on the river bottom (a "fixed transducer") which tracked the changing level of the sediment surface with a precision of a few millimetres. These acoustic bottom sensors were installed in 1993 at 3 sites (all in Zone 2) (Figure 26) with periodic monitoring at one or more of the sites through 1994 and 1995 (Rukavina 2000).

In order to continuously monitor changes in sediment depth, a system of automatic data collection was required for the bottom sensors. Equipment was developed and tested during 1996 and data was logged continuously at sites FT2 and FT3 (Figure 26) from July 1997 to July 1998. This provided a complete record of sediment stability over a full year and included the first record of sediment changes under the ice. Equipment fouling problems prevented the collection of continuous monitoring data at site FT1 (Rukavina 2000).

Rukavina (2000) reported that "in general, the acoustic records showed that bottom changes were limited to within 2 cm of the starting elevation although there were occasional spikes to as much as 6 cm. Diver measurements were generally in agreement with the acoustic data." Figure 27 shows acoustic data on bottom stability at the three fixed transducer sites and Figure 28 compares the bottom stability data with river flow rates (daily averages). Flow rates ranged widely, from 4690-10,240 m^3/s, during the study period (July 15, 1997 to July 15, 1998). However, there was "no obvious correlation between the flow rates and bottom response even during the periods when the [flow] rates were shifting most rapidly" (Rukavina 2000).

The data indicated that the sediment deposit in Zone 2 is stable (but see also the discussion in Section 2.2.6). According to Rukavina (2000), "the small-scale variations which characterize the acoustic data appeared from diver observations to be caused mainly by transport of a veneer of light-weight organic material. There was no evidence that the underlying sediment was disturbed. The deposit appears to be stable because of its location inshore of the main-channel currents in a zone where water movement consists of back eddies too slow for sediment erosion or transport."

For additional information on methods and results of the sediment stability study please refer to Rukavina (2000).

2.2.5 Sedimentation Rates

Sedimentation rates were determined by ^{210}Pb (radioisotope) dating of sediment core samples from several sites in the study area. Results are mapped in Figure 29. Sedimentation rates were highest in Zone 1 at 2.6 cm/y and lowest east of Pilon Island (0.4 cm/y). The rates were 0.8-1.1 cm/y in Zones 2 and 4 respectively. No measurements were taken in Zone 3 (Rukavina 2000).

Cores from the TCTI and Pilon 2 sites (Figure 29) contained the oldest sediment. ^{210}Pb dating indicated that sediments were over 110 years old in the bottom 10 and 23 cm of the TCTI and Pilon 2 cores respectively. However, this dating may be unreliable because both of the cores had "irregular activity profiles suggesting non-uniform sedimentation" (Rukavina 2000).

Rukavina (2000) reported an "independent size marker" in Core 109 (from Zone 2) in the form of a horizon of fibrous material in the 44-52 cm portion. This portion of Core 109 fell within the ^{210}Pb date range of 1898-1915. It had "the texture and appearance of cotton batting and appear[ed] to be some form of textile debris" (Rukavina 2000).

2.2.6 Grain Size Profiles

Information about the depositional environment was also obtained from grain size profiles within sediment cores. If depositional conditions were uniform as sediment accumulated, then the relative amounts of different grain sizes would be fairly constant vertically through the core (i.e., little variation in percentages of gravel, sand, silt and clay).

Detailed grain-size profiles were obtained for cores collected in 1996 and 1997 at 14 locations in the study area (Figure 30). Each profile shows the proportions of gravel, sand, silt and clay in 2 cm slices throughout the length of the core. Core numbers correspond to the station locations shown. In most of the cores, there were only "minor to moderate" vertical fluctuations in proportions of the various particle size. This suggests depositional conditions were relatively uniform as the sediment accumulated (Rukavina 2000).

However, three sites had large variations in sand content through the profile, indicating unstable depositional conditions (stations 109 and 128 in Zone 2 and station 'Pilon 2') (Figure 30). The vertical fluctuations in sand content would be caused by either large changes in transport rate or in sediment load. Rukavina (2000) reported that "because sites 128 and 109 occur in the midst of sites with no similar changes, it is likely that they were affected by localized discharge of sediment."

Within Zone 2 (Windmill Point to Pilon Island), size increased upwards in cores 126, 131, 132 and 156; downwards in core 117; and fluctuated in cores 109, 128 and TCTI. East of Pilon Island there were limited changes in one core (172) and very large vertical variations in the other (Pilon 2). In Zone 4 (northeast Cornwall Island) the cores were relatively uniform but sand content increased upward in core 179 and downward in core 182. One of the Zone 3 (boat launch) cores (166) was uniform throughout its length, whereas the other core (168) showed a "pronounced coarsening upwards" (Rukavina 2000).

Rukavina (pers. comm., October 1999) notes that "in spite of the acoustic evidence of bottom stability in Zone 2 [Section 2.2.4], cores from that zone show several sand lenses at various levels which suggest episodes of higher erosion, and sand content in surface samples can be as high as 20%. Furthermore, sediment captured in a sedimentation trap mounted 2 m above the bottom near the eastern acoustic-monitor site (TCTI) had a sand content of up to 5%. This raises the question of how sand transport occurs in a zone where both the current-strength and wave-disturbance data suggest very low bottom energy."

2.2.7 "Exotic" Materials in Sediment

Many of the sediment samples contained "exotic" material such as shells, wood fragments, oil and grease and fibrous material. The distribution of oil and grease and wood chips shown in Figures 31, 32 and 33 is notable because of the possible relationship of these materials with sediment contaminants.

Oils and greases (Figure 31) and wood chips (Figure 32) were found at only one site along the northeast shore of Cornwall Island (Zone 4) but were present along the entire north shore of the study area from the Lamoureux Park boat launch to east of Pilon Island (area included Zones 1, 2, 3). The frequency of occurrence along the north shore increased in the eastward (downstream) direction (Rukavina 2000).

Unidentified fibrous material (Figure 33) was found in about half of the samples collected from Zone 2 but only a very small proportion of samples from Zones 1, 3 and 4.

2.3 Bottom Sediment Chemistry

Many studies of bottom sediment chemistry along the Cornwall waterfront have been done since 1970. For a detailed review of the 1970-1994 studies of Cornwall waterfront sediment see Richman and Dreier (in prep.). Table 18 lists all surveys of Cornwall waterfront sediment done between 1970 and 1999 and the chemical parameters analyzed in each. Mercury, zinc, copper and lead have been consistently found in sediment along the Cornwall waterfront at concentrations high enough to be of concern.

Table 18. Year of sampling and chemical parameters analyzed in bottom sediment surveys at Cornwall, Ontario waterfront from 1970 to 1999. Summary maps corresponding to Figure numbers shown are provided in this review.

Year sampled	Summary map	Chemical parameters analyzed (abbreviations are defined below)	Reference
1970	Fig. 34	Hg *	MOE (1979)
1975	Fig. 35	Hg	MOE (1979)
1979	Figs. 36-39	Hg, Zn, Cu, Pb, Cd, Cr, Fe, Al, TP, TKN, TOC, oils & greases, PCBs, organochlorine pesticides	Kauss et.al. (1988)
1985	Figs. 40-43	Hg, Zn, Cu, Pb, As, Cd, Cr, Fe, Ni, TP, TKN, oils & greases, PCBs, organochlorine pesticides	Anderson (1990)
1991	Figs. 44-47	Hg, methyl Hg, Zn, Cu, Pb, Al, As, Cd, Cr, Fe, Mn, Ni, Se, TP, TKN, TOC, S	Richman (1994)
1991	n/a	Hg, Zn, Cu, Pb, Cd, Cr, Cu, Fe_2O_3, MnO, Ni, TP, TN, TOC, PAHs, PCBs, dioxins & furans	P. Mudroch, Envt Canada, unpublished data
1992	Figs. 73A & B	Hg, Zn, Cu, Pb, As, Cd, Cr, Ni, TP, TKN, TOC, oils & greases, PAHs, PCBs	Metcalfe-Smith et al. (1995)
1993	Figs. 74A & B	Hg, Zn, Cu, Pb, Ni, oils & greases, PCBs	Metcalfe-Smith et al. (1995)
1994	Figs. 48-59	Hg, Zn, Cu, Pb, Al, As, Cd, Cr, Fe, Mn, Ni, Sb, Se, TP, TKN, TOC, oils & greases, PAHs	Richman (1996)
1997	Figs. 60-72	Hg, Zn, Cu, Pb, Al, Cd, Cr, Fe, Mn, Ni, TP, TOC *PAHs, PCBs, organochlorines*	Richman (1999) *data for parameters in italics not available in a report*

* Hg=total Hg unless otherwise noted

Abbreviations:
Al-aluminum; As-arsenic; Cd-cadmium; Cu-copper; Fe-iron; Hg-mercury; Mn-manganese; Ni-nickel; O-oxygen, PAH-polycyclic aromatic hydrocarbon; PCB-polychlorinated biphenyl; Pb-lead; S-sulfur; Sb-antimony; Se-selenium; TKN-total Kjeldahl nitrogen (ammonia and organic nitrogen); TOC-total organic carbon; TP-total phosphorus; Zn-zinc.

2.3.1 Provincial Sediment Quality Guidelines

Contaminant concentrations in bottom sediment can be compared to three "effect levels" defined in the Ontario Ministry of Environment *Provincial Sediment Quality Guidelines* (Persaud et al. 1992). These levels relate concentrations of specific contaminants to their potential effect on benthic (sediment-dwelling) organisms, as follows:

No Effect Level	The contaminant concentration which is expected to have no effect on benthic organisms.
Lowest Effect Level (LEL)	The contaminant concentration that can be tolerated by the majority of benthic organisms. If the concentration is above this level, the benthic community may be impaired.
Severe Effect Level (SEL)	The contaminant concentration that is expected to be detrimental to the majority of the benthic species.

If a sediment contaminant exceeds the SEL, this means that the contamination could be having an effect on benthos. Biological impacts of contamination depend not just on contaminant concentration, but on additional factors that affect the availability of the contaminant to the benthic organisms. Therefore, a biological investigation is required to determine whether the organisms living in any given sediment are actually affected by contaminants exceeding the SEL.

Provincial Sediment Quality Guidelines for mercury, zinc, copper and lead are provided in Table 19.

Table 19. Provincial Sediment Quality Guidelines lowest and severe effect levels for mercury, zinc, copper and lead (Persaud et al. 1992).

Effect Level	Contaminant Concentration in Bottom Sediment (µg/g dry weight)			
	Mercury	Zinc	Copper	Lead
LEL	0.2	120	16	31
SEL	2.0	820	110	250

2.3.2 Cornwall Waterfront Sediment Chemistry

2.3.2.1 Overview

For the purposes of this discussion, the Cornwall waterfront is defined as *the north channel of the St. Lawrence River from the west (upstream) end of Cornwall Island, downstream to the west side of Pilon Island.* Four zones of fine sediment deposition have been identified in the area between the international bridge and Pilon Island as shown in Figure 6. These areas are of particular interest because contaminants tend to be associated with fine-grained sediment as discussed in Section 2.2.

The contaminants of concern along the Cornwall waterfront are mercury, zinc, copper and lead. In sediment surveys since 1970 these were the only contaminants consistently above the SEL and consistently enriched relative to upstream reference stations (Richman and Dreier, in prep.). Most of the following discussion therefore focuses on these four metals.

Information about other contaminants can be summarized as follows (Richman and Dreier, in prep.):

- Arsenic, iron and manganese were below the LEL in most samples from all surveys since 1979. Chromium, cadmium and nickel were above the LEL throughout the study area in several surveys but were only slightly elevated relative to upstream reference sites, with the possible exception of chromium.

- In surveys since 1970 concentrations of total phosphorus, total Kjeldahl nitrogen (i.e., ammonia and organic nitrogen) and total organic carbon along the Cornwall waterfront exceeded the LEL at most sites and the SEL at a few, but since results for these parameters were mostly similar to upstream reference sites local enrichment is not indicated.

- Sediment samples collected in 1994 between Windmill Point and Pilon Island were analyzed for PAHs as well as metals. There was evidence of local source(s) of PAHs, but concentrations were no higher than other urbanized areas of the Great Lakes. These results were confirmed in a 1997 survey that encompassed a greater area of the Cornwall waterfront.

- Total PCB concentrations have not been above the SEL in any Cornwall waterfront sediment collections. PCBs were most recently analyzed in 1997 sediment samples. The maximum concentration of PCBs observed in any survey was 2.67 μg/g in 1979 at a station near Courtaulds' shore-based outfalls (SEL=16.5 μg/g, TOC=3.1%).

Mercury, zinc, copper and lead results from each sediment survey since 1970 are summarized graphically in Figures 34-74. This series of maps shows metal concentrations relative to the LEL and SEL and maximum concentrations observed in each survey. Concentrations uncorrected for particle size were used for all maps. The sediment contaminants data are shown in relation to depositional zones of fine-grained sediment (mud and muddy sand) as determined by Rukavina (2000) (see Section 2.2). Please note that only approximate station locations were available for the 1970, 1975, 1985 and 1991 surveys. The lack of sophisticated georeferencing tools at the time of those surveys makes it impossible to accurately pinpoint station locations from the study reports.

The sediment surveys done since 1970 repeatedly established that sediment in parts of the north channel was contaminated with mercury and zinc and, to a lesser extent, lead and copper. Mercury enrichment (i.e., concentrations in sediment higher than at upstream reference locations) has been documented as far downstream as the eastern basin of Lake St. Francis as described in Section 2.3.3 (Sloterdijk 1991; Lorrain et al. 1993).

2.3.2.2 Zones of severe mercury contamination

The following discussion of mercury contamination refers to total rather than methyl mercury, unless otherwise noted. Although methyl mercury is the biologically available form of this metal, data for methyl mercury in Cornwall sediments are limited to analysis of sediment samples collected in a 1991 survey (Richman 1994). At that time, MOE and EC laboratories did not analyze for methyl mercury and very few labs in Canada did. The 1991 samples were shipped to Seattle, WA for analysis (L. Richman, MOE, pers. comm.) The high cost of methyl mercury analysis prohibited including it as a routine parameter so almost all of the Cornwall data discussed here is for total mercury, as are the Provincial Sediment Quality Guidelines.

The most severe mercury contamination in sediment along the Cornwall waterfront occurs at the following two zones, shown in Figure 6:

ZONE 1 — Downstream of the canal discharge at the present day Lamoureux Park boat launch, about 1.4 km downstream of the Domtar/ICI diffuser. (maximum observed mercury concentration = 18.0 µg/g, 1975 (MOE 1979))

ZONE 2 — Downstream of Windmill Point, immediately adjacent to the former Courtaulds shore based outfalls and extending about 1 km downstream. (maximum observed mercury concentration = 44.0 µg/g, 1975 (MOE 1979))

The recent data—from 1991, 1994 and 1997 (Richman 1994; Richman 1996; Richman 1999)—indicate that Zone 2 sediment has higher concentrations of mercury than Zone 1. Zone 2 sediment is also contaminated with zinc, copper and lead. The concentrations of mercury, zinc, copper and lead decrease with increasing distance downstream of Cornwall point sources.

The most recent survey (1997) is discussed in more detail in Section 2.3.2.3 below. Data from the 1970-1994 surveys (reviewed by Richman and Dreier, in prep.) suggest that concentrations of mercury in surface sediment have been decreasing with time. This conclusion is based on maximum concentrations detected in each survey and a general review of contaminant patterns at Zones 1 and 2.

This temporal decrease is especially evident when comparing: (i) 1991 and 1994 data with 1970 and 1975 data (Figures 44-59 and 34-35 respectively), and (ii) mercury concentration in the top 10 cm versus the bottom (>15 cm) of core samples collected in 1994 (Figures 48 and 52 respectively). In the 1994 study, mercury concentrations in core bottoms were over 70% higher than in core tops at most of the 47 stations for which cores greater than 15 cm in length were obtained (Richman 1996).

A number of factors suggest that sediment contamination at the Cornwall waterfront was locally generated. Although upstream sources such as Lake Ontario certainly contribute contaminants to the area (see Section 2.6), a review of the sediment surveys done since 1970 indicated that the primary sources of mercury and zinc in the sediment along the Cornwall waterfront were local (Richman and Dreier, in prep.). This conclusion was based on the following observations:

- Concentrations of mercury and zinc (lead and copper to a lesser extent) were consistently high in sediment downstream of known local point sources of these metals as compared to upstream reference stations.

- There was a pattern of decreasing sediment contaminant concentrations with increasing distance from local point sources.

- The pattern was similar for methyl mercury (analyzed in 1991) (Richman 1994), for which the highest concentrations were observed in sediment located downstream of local point sources.

- Surveys from 1979 and 1985, which compared mercury and zinc concentrations in sediment collected from the north versus the south channels of the St. Lawrence River at Cornwall and Massena, consistently found the highest concentrations of mercury and zinc along the north shore (Kauss et al. 1988; Anderson 1990). One would expect metal concentrations in sediment to be similar in both channels if sources upstream of the study area were the major contributors to the contamination observed along the Cornwall waterfront.

- Sediment mercury contamination extends along the north shore throughout Lake St. Francis but not along the south shore of the lake (see Section 2.3.3). Flow patterns in Lake St. Francis separate sediment contamination along the north and south shores. There is a strong current along the central axis of the lake in the St. Lawrence Seaway shipping channel and slow-moving water north and south of the channel. As a result, contaminants in south shore sediment reflect Massena point sources whereas contaminants from north shore sources tend to accumulate in north shore sediment (Lorrain et al. 1993; Carignan et al. 1994). Corrections of sediment data for particle size and organic carbon did not affect this pattern of mercury contamination in Lake St. Francis sediment (Sloterdijk 1991).

2.3.2.3 1997 Cornwall Waterfront Sediment Survey

In 1997 MOE and EC jointly collected sediment samples from a total of 24 sites in three depositional zones along the Cornwall waterfront (Zones 1, 2 and 4 in Figure 6). Samples were collected near the Lamoureux Park boat launch about 1.4 km downstream of the Domtar/ICI diffuser (Zone 1); along the north shore adjacent to and downstream of Courtaulds (Zone 2); and on the south side of the north channel off the northeast end of Cornwall Island (Zone 4) at the locations shown in Figure 60 (Richman 1999).

Please note that for discussion purposes, in this section of the report "north shore" and "south shore" refer only to the north channel of the St. Lawrence River at Cornwall (i.e., Zones 1 and 2 on the north shore and Zone 4 on the south shore).

At each 1997 station, surface samples (top 3 cm) and cores (top 10 cm) were collected from a mini-box corer sample. At six stations, two full length cores were collected from the box corer (maximum depth: 46 cm). Sediment samples were analyzed for the range of contaminants and physical parameters listed in Table 18.

Sediment chemistry data were analyzed to identify stations with contaminant enrichment, account for major sources of variation in sediment quality along the Cornwall waterfront and assess temporal changes in contaminant concentrations.

Contaminant concentrations and particle size data are provided in Tables 20 and 21. Mercury, zinc, copper and lead results are mapped in Figures 61-72. The following points summarize the results of the 1997 survey (Richman 1999):

- All sediment samples collected along the north shore of the Cornwall waterfront (Zones 1 and 2) exceeded the LEL for mercury and 48% of the samples exceeded the SEL of 2.0 μg/g (data uncorrected for particle size). In contrast, the south side of the north channel (Zone 4) had only two samples with mercury concentrations above the LEL and the SEL was not exceeded in any samples.

- In 10 cm core samples mercury concentrations were highest at stations CS131 (19.5 μg/g) and CS128 (11.2 μg/g). In the top 3 cm surface samples mercury concentrations were highest at CS131 (14.7 μg/g) and CS109 (5.79 μg/g). These stations (Figure 60) were all in Zone 2.

- Mercury concentrations were above the LEL but below the SEL in sediment from Zone 1, upstream of Courtaulds and downstream of ICI/Domtar. At the three stations sampled (CS166, CS167 and CS168) mercury concentrations in 10 cm core samples were 0.79, 1.19 and 1.71 μg/g respectively.

- Comparing median concentrations in top 10 cm sediment samples (uncorrected for particle size):
  - cadmium, nickel and TOC at north shore sites were similar to median values for south shore (Zone 4) sites;
  - median copper and lead concentrations were also similar on both sides of the north channel, but the range in concentrations was higher on the north shore than the south, suggesting some enrichment on the north shore;
  - zinc was higher on the north shore than the south shore indicating north shore enrichment;
  - aluminum, iron, manganese, total phosphorus and chromium were higher on the south shore than the north but when data were corrected for particle size this difference was eliminated.

  Medians and ranges of the uncorrected data are provided in Table 22.

- Particle size corrected data showed that all stations along the north shore (i.e., Zones 1 and 2) were enriched with mercury compared with an upstream reference area (Maitland). Sediment in Zone 2 (adjacent to and downstream of Courtaulds) was also enriched with zinc at all stations and with lead and copper at some stations relative to the upstream reference area. These results were confirmed by two different methods of particle size correction.

- Principle components analysis (PCA) showed that mercury, zinc, copper and lead were positively correlated with one another. Stations with low sand content and high concentrations of mercury, zinc, copper and lead were grouped together on the PCA plots (for actual plots and further explanation please see Richman (1999)). All stations in this group were located along the north shore in the vicinity of the former Courtaulds site (CS105, CS109, CS115, CS132 and CS164). This group did not include stations CS167 and CS168, in Zone 1 upstream of Courtaulds and downstream of ICI/Domtar, because although they were high in mercury and low in sand they were not high in zinc, copper or lead.

The PCA grouped station CS166 (also in Zone 1) with the south shore stations, which were low in mercury and sand.

- Metal concentrations in top 3 cm samples were compared with concentrations in top 10 cm samples by analysis of covariance (ANCOVA). Apart from weak differences for cadmium and chromium, metal concentrations were not significantly different in the top 3 cm *vs.* the top 10 cm of sediment collected in 1997.

  Assuming that the top 3 cm contains the most recently deposited sediment, this comparison could be used to assess whether contaminant concentrations have changed over time. However, it may be impossible to measure temporal changes in sediment quality within the top 10 cm of sediment since this is the zone of bioturbation (mixing due to the activities of sediment-dwelling and bottom feeding organisms). Sediment mixing due to physical processes within the river could also occur in all or part of this zone. Rukavina (2000) found fluctuations of up to 2 cm in sediment depth during a one year period at sites along the Cornwall waterfront (see Section 2.2.4).

- At six selected stations in Zones 1 and 4 sediment chemistry in core bottoms was compared with 10 cm core tops and top 3 cm surface samples. The approximate length of cores obtained at each station was: 29, 20, 23, 24, 20 and 24 cm at stations CS166, CS167, CS168, CS175, CS179 and CS181 respectively.
  - sediment chemistry in core tops was similar to core bottoms at the south shore (Zone 4) stations (CS175, CS179 and CS181) and at one station in Zone 1 (CS166).
  - at the other two stations in Zone 1 (CS167 and CS168):
    - sediment concentrations for many parameters were higher in core bottom samples than in top 3 and 10 cm samples. However, the percentage of clay and silt was greater in the bottom of the cores and when particle size differences were taken into account by normalization to aluminum content there were no differences between core tops and bottoms for any parameter except mercury.
    - for mercury, normalization to aluminum generated ratios of mercury to aluminum that were three and five times higher in the bottom samples than the top samples for stations CS167 and CS168 respectively. This confirms historical contamination with mercury at these two sites.

- Comparable sediment chemistry data from 1994 and 1997 was examined for six sites (CS115, CS126, CS128, CS131, CS132 and CS156 on Figure 60). Concentrations of lead, phosphorus and total organic carbon were significantly lower in 1997 than 1994. Mercury concentration was lower in 1997 at CS126 and had not changed at CS156. It may not be possible to draw conclusions from this limited data set, for the following reasons:
  - It is unknown whether changes in sediment contamination can be measured over a three year period in the St. Lawrence River. It is probably not possible, given that Rukavina (2000) observed fluctuations of ± 2 cm in sediment depth over a one year period (see Section 2.2.4).
  - The comparison of 1994 and 1997 data was done with samples collected at slightly different sites (offsets of about 5 m) and with slightly different sampling procedures. Core samples were taken by gravity core tube in 1994 and mini-box corer in 1997. The short-range spatial variability of contaminant levels is not fully understood. Diver and underwater television observations have shown that there

are large changes in the texture of Cornwall waterfront sediment which could affect contaminant levels over distances of one to a few meters (N. Rukavina, Environment Canada, National Water Research Institute, pers. comm., October 1999). However, the 1994 and 1997 data comparison used particle-size corrected data in an attempt to account for this spatial variability.

- The 1997 sediment samples were also analyzed for individual and total PAHs and PCBs. Total PAHs did not exceed the SEL in any samples nor did any individual PAHs. The range of total PAH concentrations was between 2 and 135 times lower than the SEL. Likewise, the SEL was not exceeded in any sample with respect to total PCBs or for any PCB congener. Total PCB concentrations were 36-250 times lower than the SEL.

The 1997 survey results are consistent with the results of earlier surveys discussed in Section 2.3.2.2 . Zone 2 sediment had higher concentrations of mercury than Zone 1 sediment. Zone 2 sediment was also contaminated with zinc, copper and lead. Fine-grained sediment on the south side of the north channel of the St. Lawrence River at Cornwall was not enriched with any of these metals relative to an upstream reference site. Summary maps showing 1997 concentrations of mercury, zinc, copper and lead relative to the LEL and SEL are provided in Figures 61-72.

Table 20. Metal, total phosphorus and TOC concentrations (µg/g dry weight) in sediment samples (top 10 cm core sample) collected from the St. Lawrence River, 1997. (n=1). B - bottom 10 cm of the core. S - surface sample (top 3 cm). (Source: Richman 1999).

ENV.CAN STN #	AL ug/g	CD ug/g	CR ug/g	CU ug/g	FE ug/g	HG ug/g	MN ug/g	NI ug/g	PB ug/g	ZN ug/g	Total P ug/g	TOC %
CS105	19600	0.952	46.8	45.1	19300	1.67	295	26.8	63.3	600	761	2.926
CS109	16500	1.170	44.0	55.4	16600	**4.83 ****	252	26.2	136.0	759	638	2.872
CS109-S	19000	0.929	49.1	55.6	18600	**5.79 ****	317	26.1	156.0	673	1020	2.753
CS115	15400	1.450	41.8	57.9	16400	**3.66 ****	281	26.6	99.8	603	678	2.446
CS115-S	19400	1.080	43.7	57.6	19400	1.63	339	28.0	60.5	335	977	2.266
CS117	11500	0.608	31.4	34.9	13300	**2.00 ****	245	19.9	83.5	279	549	2.353
CS117-S	18700	0.869	40.6	42.8	18600	1.35	333	25.8	60.8	270	965	2.049
CS126	9630	0.720	23.8	26.6	12700	**5.20 ****	201	19.1	25.2	125	457	3.076
CS126-S	12200	0.424	28.5	21.8	18800	1.20	357	16.0	19.7	82	852	2.980
CS127	13500	0.473	27.8	29.4	13600	**4.32 ****	280	13.8	53.6	152	548	1.681
CS128	16800	0.637	32.7	33.6	17900	**11.20 ****	345	18.0	37.3	177	681	2.850
CS128-S	10300	0.359	25.1	21.1	15200	**3.44 ****	293	14.5	23.6	69	765	1.901
CS131	17000	0.645	33.3	42.5	18300	**19.50 ****	347	17.8	40.6	186	659	4.037
CS131-S	14900	0.758	32.0	37.6	14700	**14.70 ****	288	24.9	33.7	251	853	2.737
CS132	21800	0.949	43.8	60.9	17300	**6.78 ****	329	24.6	63.3	474	818	2.516
CS135	19000	0.984	40.3	45.5	18500	**3.75 ****	331	22.9	58.8	234	660	2.108
CS135-S	12300	0.624	27.2	19.3	14300	1.22	288	21.1	27.2	117	758	1.753
CS156	11400	0.490	26.2	16.9	14400	0.80	272	14.6	26.7	134	745	1.740
CS164	16000	0.888	48.7	59.9	17700	**3.09 ****	292	24.9	136.0	669	847	2.454
CS164-S	14800	0.795	38.4	47.3	15900	1.98	291	25.2	100.0	395	841	2.822
CS166	24400	1.220	47.5	43.2	23000	0.79	360	27.5	36.0	132	1150	3.236
CS166-B	27000	1.290	53.1	46.1	23500	0.12	328	32.8	44.2	142	1140	3.263
CS166-S	23400	1.120	45.3	43.4	22200	1.06	335	30.2	33.7	133	986	3.491
CS167	16800	0.910	37.0	39.7	16300	1.19	265	29.9	36.6	107	867	2.970
CS167-B	24700	1.420	44.0	64.4	15000	**4.63 ****	247	38.8	58.3	158	1120	6.745
CS167-S	12200	0.687	26.7	23.6	13700	1.72	239	19.5	21.9	76.9	653	2.370
CS168	17000	0.877	37.5	31.7	17000	1.71	290	24.0	38.0	102	947	1.813
CS168-B	25600	1.590	51.5	75.1	17000	**13.70 ****	286	35.9	73.6	187	1090	5.818
CS168-S	11800	0.754	26.8	21.2	13100	0.70	222	20.8	24.7	80.7	724	2.178
Lowest Effect Level		0.6	26	16	2%	0.2	460	16	31	120	600	1
Severe Effect Level **		10	110	110	4%	2	1100	75	250	820	2000	10

Table 20 continued:

ENV.CAN STN #	AL ug/g	CD ug/g	CR ug/g	CU ug/g	FE ug/g	HG ug/g	MN ug/g	NI ug/g	PB ug/g	ZN ug/g	Total P ug/g	TOC %
CS171	18600	0.690	36.5	31.3	19100	0.44	359	20.6	29.6	118	918	2.016
CS172	31200	1.120	58.8	46.1	26600	0.62	439	34.1	44.0	350	1120	3.340
CS173	32700	1.210	58.4	46.1	28900	0.13	489	34.8	39.3	155	1150	3.130
CS175	29200	1.020	52.4	39.7	26500	0.08	464	30.4	37.5	136	1080	2.532
CS175-B	23000	1.380	48.5	36.4	21600	0.24	359	30.8	46.8	131	931	2.246
CS176	33300	1.150	59.1	46.3	30000	0.15	592	35.4	39.0	156	1180	3.099
CS176-S	27500	1.100	48.7	41.5	26200	0.15	665	38.4	29.5	136	1110	3.229
CS177	28600	0.976	49.5	38.1	26000	0.12	495	30.8	32.8	132	1020	2.599
CS179	28200	1.020	50.6	38.9	26500	0.14	488	31.5	35.2	136	1090	2.227
CS-179-B	20400	1.070	41.5	30.0	20300	0.16	344	26.9	36.2	108	909	1.945
CS-179-S	23900	1.040	42.9	34.0	22900	0.12	445	33.7	26.5	120	956	3.079
CS181	25500	1.010	47.9	35.6	24300	0.13	435	29.9	33.2	126	996	2.030
CS-181-B	22900	1.290	47.7	35.9	22300	0.20	366	31.2	43.3	124	952	2.141
CS-181-S	21700	0.916	39.7	30.6	20800	0.13	386	28.4	27.0	110	698	2.528
CS182	24300	0.901	43.8	31.8	23500	0.12	492	28.4	27.2	117	1090	2.507
CS-182-S	23500	0.939	42.4	33.4	23000	0.12	513	32.3	26.3	116	986	2.827
Lowest Effect Level		0.6	26	16	2%	0.2	460	16	31	120	600	1
Severe Effect Level **		10	110	110	4%	2	1100	75	250	820	2000	10

Table 21. Particle size data for sediment samples (top 10 cm core sample) collected from the St. Lawrence River, 1997. (n=1). B - bottom 10 cm of the core. S - surface sample (top 3 cm). Source: Richman (1999).

Station Number	Percent Sand	Percent Silt	Percent Clay		Station Number	Percent Sand	Percent Silt	Percent Clay
CS105	15.7	52.4	31.9		CS167-S	36.7	45.4	17.9
CS109	14.2	55.9	29.9		CS167	27.9	51.6	20.5
CS109-S	9	68.6	22.4		CS167-B	8.7	60.6	30.6
CS115	17.6	60.9	21.5		CS168-S	45	38.7	16.2
CS115-S	33	42.8	24.1		CS168	42.1	42.1	15.9
CS117-S	42.2	35.2	22.5		CS168-B	3.7	62.4	34
CS117	51.2	33.3	15.5		CS171	35.2	43.6	21.1
CS126-S	55.6	28.4	16		CS172	9.9	59.4	30.6
CS126	58.3	30.5	10.5		CS173	5.4	66	28.6
CS127	68.6	18.2	13.2		CS175	12.9	59.1	28
CS128-S	65.8	18.8	15.4		CS175-B	13.7	64.8	21.5
CS128	39.8	42.3	16.8		CS176-S	3.6	61.8	34.6
CS131-S	25.9	52.6	21.5		CS176	6.2	65	28.8
CS131	35.7	43.1	21.2		CS177	13.5	58.4	28.2
CS132	39	38	23		CS179-S	11.4	59.1	29.5
CS135-S	38.6	39.5	21.8		CS179	14.2	57.7	28.1
CS135	33.2	44.9	21.9		CS179-B	22.1	55	22.9
CS156	48.6	31.7	19.7		CS181-S	11.8	60.1	28.1
CS164-S	16.5	60	23.5		CS181	11.5	62.2	26.3
CS164	29	53.6	17.4		CS181-B	16.2	59.6	24.2
CS166-S	9.3	66.4	24.3		CS182	14	56.1	29.9
CS166	7.6	66.2	26.2		CS182-S	9.8	57.1	33.1
CS166-B	6.1	65.4	28.5					

Table 22. Median, minimum and maximum (all stations combined) concentrations of metals, total phosphorous and TOC (μg/g dry wt) in sediment (top 3 cm grab and top 10 cm core samples) collected from the St. Lawrence River in 1997. Source: Richman (1999).

Parameter	Median, minimum and maximum concentration (µg/g dry wt) in sediment (all stations combined)											
	North Shore (Zones 1 & 2)						South Shore (Zone 4)					
	Surface (top 3 cm)			Core (top 10 cm)			Surface (top 3 cm)			Core (top 10 cm)		
	median	min	max	median	min	max	median	min	max	median	min	max
Aluminum	148000	10300	23400	16800	9630	31200	23700	21700	27500	28600	24300	33300
Cadmium	0.758	0.359	1.12	0.888	0.473	1.45	0.9895	0.916	1.1	1.02	0.901	1.21
Chromium	32	25.1	49.1	37.5	23.8	58.8	42.65	39.7	48.7	50.6	43.8	59.1
Copper	37.6	19.3	57.6	42.5	16.9	60.9	33.7	30.6	41.5	38.9	31.8	46.3
Iron	15900	13100	22200	17300	12700	26600	22950	20800	26200	26500	23500	30000
Mercury	1.63	0.696	14.7	3.06	0.436	19.5	0.129	0.124	0.151	0.131	0.079	0.153
Manganese	293	222	357	292	201	439	479	386	685	489	435	592
Nickel	24.9	14.5	30.2	24	13.8	34.1	33	28.4	38.4	30.8	28.4	35.4
Total Phosphorus	852	653	1020	745	457	1150	971	698	1110	1090	996	1180
Lead	33.7	19.7	156	44	25.2	136	26.75	26.3	29.5	35.2	27.2	39.3
Zinc	133	69	673	186	102	759	118	110	136	136	117	156
TOC (%)	2.37	1.753	3.491	2.516	1.681	4.037	2.953	2.528	3.229	2.532	2.03	3.13

2.3.2.4 Oil Tank Storage Area/Tank Farm Site

Contaminated sediment has also been found in a depositional zone (Zone 3 in Figure 6) located in a small embayment upstream of Windmill Point about 500 m downstream from Cornwall Harbour and the above ground oil storage tanks at Universal Terminals. This depositional area, referred to as the oil tank storage area or tank farm site, was considered for a demonstration sediment clean-up project on the basis of a preliminary sediment assessment in 1991 (P. Mudroch, Environment Canada, unpublished data). The site was investigated in more detail in 1992 (MOE) and 1993 (EC). Chemical characterization and biological assessment of sediment collected in 1992 and 1993 indicated that sediment clean-up was not required at the oil tank storage area site (Metcalfe-Smith et al. 1995).

In 1991, three replicate surface sediment samples were collected from one station only in the oil tank storage area depositional zone. Concentrations of PCBs, PAHs and mercury were consistently above the LEL. Mercury concentrations in the three replicate samples were 0.7, 1.1 and 4.2 µg/g, exceeding the SEL (2.0 µg/g) in one replicate. However, intensive sampling in 1992 and 1993 showed that the degree of sediment contamination at the oil tank storage area was not severe and the area affected was very localized (Metcalfe-Smith et al. 1995).

In 1992, severe effect levels were exceeded at only one station (89) and only for chromium and copper (Figures 73A and 73B). In 1993 the SEL was not exceeded for any parameter (Figures 74A and 74B). Oils and greases were higher than MOE Open Water Disposal Guidelines (Persaud et al. 1992) at almost all sites in both surveys as shown in Figures 73A and 74A. Maximum concentrations of total PAHs were 30-50% of the SEL at three stations (# 89, 91, 94) in 1992; PAHs were not measured in 1993 (Metcalfe-Smith et al. 1995).

In spite of the high concentrations of oils and greases at most sites, bioassessment of the 1993 samples did not indicate that sediments from the tank farm study area were particularly toxic or that the benthic community was severely degraded. Toxic effects were identified for at least one bioassay endpoint at five of twelve sites tested (# 5, 7, 11, 15, 17). But at each of these five sites, only one or two endpoints (tests) out of ten showed statistically significant toxicity, and the decreases in growth, survival or reproduction were not extreme. Only one station (# 7) had a benthic community (abundance of organisms, number of taxa, percent distribution of taxa) that was degraded relative to comparable Great Lakes reference sites (Metcalfe-Smith et al. 1995).

Based on the combined chemical and biological findings from the 1992 and 1993 in-depth studies, it was concluded that sediment clean-up (removal and disposal) was not required at the tank farm site (Metcalfe-Smith et al. 1995).

2.3.2.5 Northeast Cornwall Island Depositional Zone

Another zone of fine sediment deposition is located on the south side of the north channel, off the northeast shore of Cornwall Island (Zone 4 in Figure 6). Historical and recent data show that contaminant concentrations in Zone 4 sediment are similar to those at upstream reference sites (MOE 1979; Kauss et al. 1988; Anderson 1990; Richman 1994; Richman 1999). Figures 61-72 show 1997 mercury, zinc, copper and lead concentrations relative to the LEL and SEL in this depositional zone.

2.3.3 Lake St. Francis Sediment Chemistry

Lepage (1999) summarized the results of surveys of sediment quality (chemical and physical parameters) conducted in Lake St. Francis since 1975 as follows:

> All these studies [cited in Lepage 1999] found that the northern part of Lake Saint-François (region of Cornwall, Lancaster Basin, Saint-Zotique basin) [Figure 75] was particularly affected by high concentrations of mercury, whereas the southern part (regions of Massena and Christatie Island, Cèdres Basin and Grenadier Basin) was contaminated to a large extent by PCBs. It was suggested that in both cases the contamination came from upstream sources in the Cornwall-Massena sector...The data collected in 1989 indicated that the lake sediments were much less contaminated than they had been 20 years earlier. The highest mercury levels appear to have been recorded in the mid-1970s, with a maximum value of 3.2 µg/g recorded in 1975..., subsequently dropping to 1.47 µg/g in the late 1970s...With regard to total PCBs and excluding the high concentrations encountered near the Massena industrial plants, the maximum value in Lake Saint-François was 1.9 µg/g in the late 1970s and 0.27 µg/g in 1989.

The 1989 data referred to above was collected by Lorrain et al. (1993), who delineated PCB and mercury contamination in Lake St. Francis bottom sediment and described changes in contamination patterns and levels that had occurred since a similar survey in 1979-81 (Sloterdijk 1985). The comparison between two studies done ten years apart was made possible by a geostatistical data processing method called universal kriging.

Northern vs. southern sectors

Sediment in Lake St. Francis is enriched with mercury in the northern sector and with PCBs in the southern sector. According to Lorrain et al. (1993) "this spatial distribution suggests preferential transport of PCBs from the Massena region to the south shore, and mercury from the Cornwall region to the north shore". There was a significant difference in mean mercury and PCB concentrations between the north and south sectors in both 1979 and 1989, confirming that the water masses along the north and south sides of the lake do not mix with each other. This segregation is thought to be due to high current velocities in the seaway shipping channel which runs lengthwise down the centre of the otherwise slow-moving lake (Sloterdijk 1985; Lorrain et al. 1993).

In 1989 sediment was analyzed from 60 stations throughout Lake St. Francis. The highest mercury concentrations were observed north of the St. Lawrence Seaway near Pilon Island (0.64 µg/g), Hamilton Island (0.61 µg/g) and in the Lancaster basin (0.67 µg/g). These values are all above the LEL (0.2 µg/g) but below the SEL (2.0 µg/g) for mercury. Throughout the southern sector of Lake St. Francis, mercury concentrations were below 0.39 µg/g (Lorrain et al. 1993).

In both 1979 and 1989 there was a significant difference ($p<0.0001$) between the northern and southern sectors of the lake with respect to mean mercury concentration. Mean mercury concentrations in the northern sector were 71% and 60% higher than in the southern sector in 1979 and 1989 respectively (Lorrain et al. 1993).

1979 vs. 1989

Over the ten year period from 1979 to 1989, there was a significant (34%) reduction in mean mercury concentrations in Lake St. Francis overall (including north and south sectors). Mean kriged whole lake mercury concentrations were 0.182 ±0.114 µg/g in 1989 and 0.275 ±0.241 µg/g in 1979. In comparison, the mean whole lake PCB concentration showed a significant decrease of 89% over the same period.

In the St. Zotique and Lancaster basins and upstream from Hamilton Island (i.e., the northern sector of the lake) (Figure 75) the reduction over the ten year period was greater than for the lake as a whole. In the northern sector there was a 45% reduction in mean kriged mercury concentrations from 1979 to 1989, whereas in the southern sector the mean mercury concentration was comparable in both 1979 and 1989 (Lorrain et al. 1993).

2.4 Suspended Solids

A long-term (5 year) study of the resuspension, transport and contamination of suspended matter in Lake St. Francis began in 1994 (Lepage 1999). The study had three main objectives:

- to evaluate the effectiveness of remedial operations undertaken in the Cornwall-Massena sector of the St. Lawrence River based on the quality of the suspended solids that drift towards Lake St. Francis;
- to link contamination of suspended solids in Lake St. Francis to corresponding contaminant sources; and
- to assess the mechanisms of suspended solids transport in the Cornwall-Massena and Lake St. Francis stretch of the St. Lawrence River.

An additional goal was to investigate the influence of atmospheric (storm winds) and hydrologic (variations in discharge rates and water levels) events on the resuspension, transport and deposition of contaminated sediment in the upstream part of Lake St. Francis.

Results obtained during the first three years of the study (September 1994 to September 1997) are documented in Lepage (1999) and summarized here. Sampling results up to May 1999 are summarized in Tables 23 and 24.

Samples of suspended matter were collected from the following six long term sensing sites (LTSS) shown in Figure 76:

PILON	just east of Pilon Island
TCTI	just west of Pilon Island downstream from the former Courtaulds site, offshore from NavCanada, formerly Transport Canada Training Institute (TCTI)
SFS	St-Francis South, near shore area on south side of wide central portion of lake
SFC	St-Francis Centre, shipping channel in wide central portion of lake
SFN	St-Francis North, near shore area on north side of wide central portion of lake
LSL	an upstream reference site in Lake St. Lawrence, initially located half way between Moulinette and Macdonnell Islands (1994-1995) but moved to just upstream of the Moses-Saunders Power Dam (1996-1997) due to low sedimentation rates at the original site.

At each site, samples of suspended matter were continuously collected in sediment traps suspended one and two metres above the river bottom. Samples were periodically retrieved by divers, and approximately every four months frozen samples were sent for physical and chemical analysis including analysis for mercury and PCBs. Current speed and direction, light transmission, water conductivity and temperature were measured at all stations except the two sites near Pilon Island (Lepage 1999).

Two additional sites were sampled at the Cornwall waterfront between November 1997 and May 1999. These were the "RAP Reef site" just downstream from the Lamoureux Park boat launch and the "Government Dock" site in the oil tank storage study area (embayment immediately downstream from the oil tank farm). Results for these sites are discussed separately in Section 2.4.4.

2.4.1 Quantities of Suspended Matter

Sedimentation in the Great Lakes occurs in the deep basins upstream of Kingston, so St. Lawrence River water carries only a small amount of suspended matter compared with other large rivers of the world (Lepage 1999). The mean concentration of suspended matter at

Cornwall has been estimated at 1.0 ±0.6 mg/L based on measurements between 1989 and 1993 (Rondeau et al. (in press) cited in Lepage 1999). This is equivalent to an annual load of 199,000 ±10,000 metric tons, but only a small proportion is deposited in the sedimentation basins in Lake St. Francis. An estimated 79,000 metric tons of fine sediment are deposited annually in Lake St. Francis, which is 17% of the total annual load to the lake (Carignan and Lorrain (in press) cited in Lepage 1999).

The daily average quantity of suspended matter collected between July 1996 and September 1997 by Lepage (1999) varied spatially and temporally. The amount of suspended matter was consistently highest at SFS (1.10 ± 0.34 g/day wet wt) and lowest at LSL (0.31 ± 0.15 g/day). Daily accumulation rates were highest in the period from May-July 1997 (mean 1.05 ± 0.46 g/d) and lowest from November 1996 to February 1997 (mean 0.48 ± 0.27 g/d) (Lepage 1999).

2.4.2 Contaminants in Suspended Matter

The highest mercury concentrations occurred in the area of Pilon Island. On the western side of the island (TCTI), at the Cornwall waterfront, mean mercury concentration was 1.15 ± 0.38 µg/g. On the eastern side (PILON) it was 0.46 ± 0.28 µg/g. Immediately downstream, in Lake St. Francis, mean mercury concentration was highest on the north side of the river (0.33 ± 0.04 µg/g at SFN). This was two times higher than the value observed on the south side at SFS (0.15 ± 0.02 µg/g). dry weight

dry weight

PCB contamination showed the opposite pattern, with the highest mean concentration seen on the south side of Lake St. Francis (0.30 ± 0.20 µg/g at SFS). This was five times higher than the mean concentration observed on the north side at SFN (0.06 ± 0.01 µg/g) and ten time higher than the value for the upstream reference site in Lake St. Lawrence (0.03 ± 0.02 µg/g at LSL) (Lepage 1999). dry weight

dry weight

The patterns of mercury and PCB contamination in Lake St. Francis reported by Lepage (1999) are consistent with earlier reports by Sloterdijk (1985) and Lorrain et al. (1993).

Table 23 lists mean mercury concentrations at each of the five long term sensing sites and Figure 77A, reproduced from Lepage (1999), shows variations in total mercury concentration in suspended matter at each of the five sites sampled. Detailed data for nutrient and total metal concentrations in suspended solids at the LTSS sites are provided in Table 24.

west ? dry weight

Mean mercury concentrations were highest in the area of Pilon Island, with a maximum at the TCTI station to the east of Pilon Island (1.15 ± 0.38 µg/g). Concentrations at TCTI were 8-9 times higher than at SFS and 5-7 times higher than at LSL. The lowest mean mercury concentration was at the SFS station, whereas the lowest mean PCB concentration was at the LSL station (Lepage 1999).

In 1988, total mercury in suspended solids ranged from 0.11-0.25 µg/g dry wt at 5 sites in the St. Lawrence River at Cornwall and Massena (Anderson and Biberhofer 1991). In this study, water was pumped from the river at a depth of 1 m and centrifuged to collect suspended particulate matter. Two of the five 1988 sampling sites were located in the north channel of the river: one between the Moses-Saunders Power Dam and the west end of Cornwall Island and the other about 4 km downstream of Pilon Island.

Lepage (1999) reported that PCB concentrations were highest at the SFS station: ten times higher than LSL and five times higher than SFN values. The composition of PCB isomers in suspended solids indicated that PCBs at the north shore of Lake St. Francis (SFN) are mainly from incoming Great Lakes waters, whereas local sources of PCB contamination mainly influence the south shore (SFS) (Lepage 1999). Anderson and Biberhofer (1991) found trace concentrations of PCBs (<130 ng/g) in one sample only, from a site at the mouth of the Raquette River on the south shore. PCBs were undetected in all other 1988 samples.

Table 23. Mean total mercury concentration in suspended matter collected at long term sensing sites in Lake St. Francis between July 1996 and September 1997 (Lepage 1999).

Station Name	Station Location	Mean total Hg in suspended matter (µg/g wet wt)
TCTI	west ~~east~~ of Pilon Island, opposite TCTI	1.15 ± 0.38
PILON	east ~~West~~ of Pilon Island	0.46 ± 0.28
SFN	north side of Lake St. Francis	0.33 ± 0.04
SFS	south side of Lake St. Francis	0.15 ± 0.02
LSL	Lake St. Lawrence	0.17 ± 0.05
SFC	[illegible]	

Provincial Sediment Quality Guidelines for total mercury (Persaud et al. 1992):
Severe Effect Level=2.0 µg/g   Lowest Effect Level=0.2 µg/g

Table 24. Nutrient and total metal concentrations in suspended solids collected at long term sediment sampling (LTSS) sites. (Source: Serge Lepage, Environment Canada, Quebec Region.) BOTTOM=sample trap located 1.0 m above sediment surface; BOTTOM=sample trap located 1.5-2.0 m above sediment surface. Station locations are shown in Figure 76.

Station	Trap	Date Sampled	Cu (µg/g)	Pb (µg/g)	Zn (µg/g)	Hg (µg/g)	C org total (% weight)	C inorg total (% weight)	N org total (% weight)
LSL (1994-95)	?	06-Jun-95	--	--	--	--	--	--	--
	?	05-Jul-95	44	30	138	0.125 R	3.19	0.84	0.39
	?	10-Aug-95	insf. mat.	insf. mat.	insf. mat.	0.151	4.81	0.45	0.51
	?	06-Sep-95	53	35	156	0.135	4.84	0.65	0.56
	?	06-Sep-95	51	33	150	0.134	4.50	0.71	0.52
	?	04-Oct-95	53	32	155	0.150	5.19	0.32	0.59
	?	04-Oct-95	52	33	161	0.319	4.51	0.86	0.53
	?	08-Nov-95	59	34	155	0.147	5.44	0.61	0.60
LSL (1996-97)	?	05-Dec-95	--	--	--	--	--	--	--
	?	15-Mar-96	67	38	170	0.142	insf. mat.	insf. mat.	insf. mat.
	Mixed T+ B	30-Jul-96	86 R	35 R	182 R	0.162	5.63	0.75	0.55
	Mixed T+ B	10-Sep-96	726	39	296	0.234	6.15	0.12	0.50
	Mixed T+ B	06-Nov-96	91	42	191	0.194	6.12	0.15	0.64
	Mixed T+ B	26-Feb-97	323	40	177	0.179	6.45	0.43	0.73
	Mixed T+ B	15-May-97	675	21	169	0.165	6.35	0.42	0.70
	Mixed T+ B	07-Jul-97	3380	31	176	0.147	5.79	0.32	0.61
	TOP	18-Sep-97	93	20	149	0.192	5.87	0.35	0.69
	TOP	06-Nov-97	1470	39	167	0.166	6.04	0.35	0.70
	TOP	20-Feb-98	215	42	172	0.056	6.24	<0.01	0.63
	TOP	21-May-98	2400 R	48	149	0.045	5.81	0.50	0.61
	TOP	21-Jul-98	76	42	173	0.217	5.88	0.40	0.68
	TOP	18-Nov-98	115	42	169	0.654	6.24	0.27	0.65
TCTI	?	06-Jun-95	--	--	--	--	--	--	--

Station	Trap	Date Sampled	Cu (µg/g)	Pb (µg/g)	Zn (µg/g)	Hg (µg/g)	C org total (% weight)	C inorg total (% weight)	N org total (% weight)
	?	05-Jul-95	--	--	--	--	--	--	--
	?	10-Aug-95	--	--	--	--	--	--	--
	?	06-Sep-95	--	--	--	--	--	--	--
	?	04-Oct-95	--	--	--	--	--	--	--
	?	07-Nov-95	--	--	--	--	--	--	--
	?	05-Dec-95	insf. mat.	insf. mat.	insf. mat.	1.440 R	5.27	0.32	0.50
	?	16-Mar-96	89	47	271	1.350 R	6.54	0.21	0.60
	Mixed T+B	30-Jul-96	71	37	208	0.540	5.26	1.28	0.63
	Mixed T+B	10-Sep-96	361	40	263	0.886	6.23	0.01	0.50
	Mixed T+B	06-Nov-96	255	39	256	0.818	6.88	0.01	0.65
	Mixed T+B	01-Mar-97	114	34	293	1.680	5.67	0.84	0.57
	Mixed T+B	01-Mar-97	70	42	275	1.120	5.65	1.33	0.60
	Mixed T+B	15-May-97	254	47	215	1.350	4.98	0.86	0.52
	TOP	07-Jul-97	180	34	196	0.408	5.41	0.93	0.61
	TOP	18-Sep-97	480	27	195	0.658	5.59	0.52	0.63
	TOP	07-Nov-97	125	33	203	0.480	6.06	1.04	0.71
	TOP	22-May-98	137	52	191	0.783 R	4.15	2.49	0.49
	TOP	22-May-98	380	38	182	4.180	4.81	1.66	0.57
	TOP	22-Jul-98	217 R	49 R	185 R	4.070	5.73	0.53	0.59
	TOP	17-Nov-98	118	32	173	0.893	5.89	0.27	0.59
	TOP	17-Nov-98	173	37	202	1.020	6.19	0.38	0.67

Station	Trap	Date Sampled	Cu (µg/g)	Pb (µg/g)	Zn (µg/g)	Hg (µg/g)	C org total (% weight)	C inorg total (% weight)	N org total (% weight)
PILON	?	06-Jun-95	--	--	--	--	--	--	--
	?	05-Jul-95	--	--	--	--	--	--	--
	?	10-Aug-95	--	--	--	--	--	--	--
	?	06-Sep-95	--	--	--	--	--	--	--
	?	04-Oct-95	--	--	--	--	--	--	--
	?	07-Nov-95	--	--	--	--	--	--	--
	?	05-Dec-95	insf. mat.	insf. mat.	insf. mat.	0.261	5.68	0.11	0.56
	?	16-Mar-96	46	42	157	1.100	insf. mat.	insf. mat.	insf. mat.
	TOP	30-Jul-96	50	33	194	0.319 R	4.31	1.71	0.50
	TOP	10-Sep-96	74	34	205	0.250	5.51	0.32	0.37
	TOP	10-Sep-96	82	35	206	0.724	5.61	0.23	0.40
	TOP	06-Nov-96	75	35	195	0.327	6.25	0.08	0.61
	TOP	27-Feb-97	68	28	149	0.372	4.42	0.59	0.45
	TOP	15-May-97	183	42	161	0.281	3.92	0.63	0.37
	TOP	15-May-97	121	52	145	0.295	3.43	1.22	0.43
	TOP	07-Jul-97	61 R	28 R	172 R	0.274 R	4.92	0.70	0.52
	TOP	18-Sep-97	63	32	174	0.281	4.38	1.32	0.53
	TOP	07-Nov-97	79	33	412	0.266	5.39	1.29	0.62
	TOP	22-May-98	66	37	117	0.590	4.62	0.89	0.48
	TOP	22-May-98	74	46	147	1.840	4.07	1.42	0.47
	TOP	22-Jul-98	66	42	167	0.706	4.93	0.87	0.50
	TOP	17-Nov-98	60	33	159	0.398	6.25	0.27	0.64
	TOP	17-Nov-98	88	35	191	0.656	5.35	1.29	0.67
Gov Dock	BOTTOM	07-Nov-97	38	25	134	0.305	4.93	0.18	0.46
	TOP	22-May-98	43	39	116	1.250	3.92	1.76	0.51
	TOP	22-Jul-98	40	43	144	0.976	5.13	0.57	0.51
	BOTTOM	22-Jul-98	43	43	144	4.140	3.88	2.54	0.54
	TOP	17-Nov-98	48	34	145	1.750	5.92	< 0.01	0.50
	BOTTOM	17-Nov-98	44	32	135	1.360	9.51	< 0.01	0.42

Station	Trap	Date Sampled	Cu (µg/g)	Pb (µg/g)	Zn (µg/g)	Hg (µg/g)	C org total (% weight)	C inorg total (% weight)	N org total (% weight)
RAP Reef	BOTTOM	07-Nov-97	41	29	143	0.423	3.76	1.21	0.42
	TOP	22-May-98	42	32	120	4.970	5.89	1.71	0.57
	TOP	22-Jul-98	39	41	136	1.470	5.83	0.97	0.59
	BOTTOM	22-Jul-98	37	39	133	1.120	6.20	0.68	0.54
	TOP	17-Nov-98	40	31	133	1.730	5.72	0.41	0.58
	BOTTOM	17-Nov-98	46	37	147	2.050	6.59	< 0.01	0.58

Notes:

-- indicates that no sampling was done

insf. mat. indicates that the analysis was not performed because of insufficient material

R represents the average value of two replicas done by NLET

2.4.3 Sediment Resuspension

Overall, the winter period (November to March) appeared to be associated with higher contaminant levels in suspended solids. This includes the time of ice cover, when no wind or wave action affects the sediments. Lepage (1999) proposed that the higher contaminant levels could be linked to increased resuspension of contaminated surficial sediments during periods of more dynamic hydrometeorological conditions. To test this hypothesis, wind, wave and current regimes likely to cause resuspension were analyzed (Lepage 1999), excluding the period of ice cover (January 31-March 25) for each year.

Sediment resuspension in aquatic ecosystems is generally greatest with a fetch (unobstructed stretch of surface water) of a few kilometres and wind velocities of at least 20-30 km/h. Lepage (1999) calculated that winds of 28 km/h would generate waves likely to affect the water bottom to a depth of about 1.0 m in the Cornwall/Massena channels and about 1.3 m in the area of Thompson and Christatie Islands further downstream (Figure 75).

Hourly weather station data collected at Saint-Anicet (mid-south shore, Quebec) were used to calculate the number of hours during sediment trap deployment in which wind speed exceeded 28 km/h out of the SW, S or E (azimuths 240-110°) and the NE, N or NW (azimuths 40-300°). To account for the effect of ice cover, wind information for January 31 to March 25 of each year was excluded. The wind has no effect on the water surface during this period and cannot cause sediment resuspension.

Winds exceeding 28 km/h blew 6% and 4% of the time for azimuths 240-110° and 40-300° respectively. Measured concentrations of mercury and PCBs for samples collected at each long term sensing site were compared with the number of hours of wind >28 km/h in the period during which the samples were collected (Lepage 1999).

Only the SFS station (south shore) showed significant ($p<0.002$) correlations between the wind and the mercury and PCB concentrations measured in suspended solids. Mercury concentration was only correlated with winds in the 240-110° range (SW, S or SE winds). High winds from the south would cause the greatest wave action along the north shore of Lake St. Francis. This could result in resuspension of sediment contaminated with mercury. Unlike mercury, PCBs were correlated with winds from both the 240-110° range (SW, S or SE winds) and the 40-300° range (NE, N or NW winds) (Lepage 1999).

Please note that although there was a correlation between SFS mercury and wind data, the actual concentrations of mercury in suspended solids at the SFS site were at or below concentrations observed at the upstream reference site (LSL).

Lepage (1999) concluded that "these results provide additional support for the hypothesis that resuspension of contaminated sediment in the upstream part of Lake St. Francis contributes to contamination of the fluvial sector where the SFS station was set up."

In addition to wind, fluctuations in river flow (discharge) could also resuspend sediment and may explain some of the increased contaminant concentrations observed in winter. South shore tributary discharges fluctuate regularly during periods of flooding and in winter; these tributaries are known to contribute to the suspended solids load moving through Lake St. Francis. As well, although the discharge of the St. Lawrence River itself is regulated, "variations of over 1000 m^3/s are noted occasionally over eight- to ten-day intervals, as occurred in January and February 1995. Such variations could cause the resuspension of unconsolidated sediment in the Cornwall-Massena channels, where the river cross-section is less than 10,000 m^2" (Lepage 1999).

2.4.4 Supplementary Cornwall Waterfront LTSS Stations

Introduction

The LTSS network was supplemented in the summer of 1997 with 2 additional stations located at the Cornwall waterfront. Figure 76 shows the locations of the RAPReef and GovDock stations. RAPReef was located near a submerged artificial reef constructed as a RAP project in 1995. GovDock was in the oil tank storage area/tank farm site embayment (depositional Zone 3), immediately upstream of Windmill Point and downstream of the former government dock at Cornwall Harbour. Water depth was 11 m at both stations and both were positioned at the boundary between the high flow of the main channel and the more quiescent near shore zone.

Each station was equipped with 2 sets of 2 passive cylindrical sediment traps. One set of traps ("bottom") was positioned 1 m above the river bottom; the second set ("top") was 5 m above the river bottom. There were 5 sampling events with the deployment time for the events ranging from 61-194 days (Table 25). The corresponding sampling schedule at the upstream reference station (LSL) is shown in Table 26. Suspended particulate matter (SPM) recovered from the sediment traps was analyzed for a number of parameters including mercury, zinc, copper and inorganic and organic carbon.

Table 25. Sampling schedule for RAPReef and GovDock LTSS sites, 1997-1999. N=1 unless otherwise shown. Source: H. Biberhofer and S. Lepage, Environment Canada.

Sampling Interval	Trap Deployment Date	Sample Recovery Date	Samples Recovered	Deployment Time (days)
1	July 8, 1997	Nov 7, 1997	RAPReef bottom GovDock bottom (N=2)	122
2	Nov 7, 1997	May 22, 1998	**RAPReef top** GovDock top	194
3	May 22, 1998	July 22, 1998	RAPReef top, bottom **GovDock** top, **bottom**	61
4	July 22, 1998	Nov 17, 1998	RAPReef top, bottom GovDock top, bottom	117
5	Nov 17, 1998	May 18, 1999	RAPReef top, bottom GovDock top, bottom	181

Table 26. Sampling schedule for upstream reference LTSS site (LSL), 1997-1999. Source: S. Lepage, Environment Canada.

Trap Deployment Date	Sample Recovery Date	Samples Recovered	Deployment Time (days)
July 7, 1997	Sept 18, 1997	LSL top	73
Sept 18, 1997	Nov 6, 1997	LSL top	49
Nov 6, 1997	Feb 20, 1998	LSL top	75
Feb 20, 1998	May 21, 1998	LSL top	90
May 21, 1998	July 21, 1998	LSL top	61
July 21, 1998	Nov 18, 1998	LSL top	89
Nov 18, 1998	May 20, 1999	LSL top	183

Results and Discussion

Seventeen samples in total were collected from the RAPReef and GovDock sites (including 2 replicates from GovDock bottom trap November 7, 1997) as shown in Table 25. The data range for mercury in the two GovDock replicate samples was small (0.26-0.31 µg/g). The results are shown in Figure 77B (upstream reference site LSL), Figures 77C and 77D (RAP Reef) and Figures 77E and 77F (GovDock). Please note that values shown as 0.01 on these figures were actually <0.01.

In approximately half (9/17) of the GovDock and RAPReef samples, mercury concentration was an order of magnitude higher than in the corresponding LSL sample (upstream reference site). Two samples exceeded the severe effect level for mercury (2.0 µg/g). On May 22, 1998 RAPReef (top) contained 4.97 µg/g mercury compared with 0.05 µg/g at LSL (top) over the corresponding period. On July 21, 1998 GovDock (bottom) contained 4.14 µg/g mercury compared with 0.22 µg/g at LSL (top) over the corresponding period.

Climate data was reviewed to determine whether precipitation events could have resulted in the flushing of land-based material that might have been recovered from the traps in the May and July 1998 samples (Environment Canada, Climate and Water Products Division). Mean rainfall was 2.7 mm (SD=5.9, N=813) over the entire period of sample trap deployment (July 1997-May 1999) at the GovDock and RAPReef sites (H. Biberhofer, Environment Canada, pers. comm.).

89.5 mm of rain fell over a 2-day period in sampling interval 3 (May 22, 1998 to July 22, 1998). Surface runoff associated with this event may have contributed to the mercury enrichment measured in the bottom trap at GovDock on July 22, 1998. Increased inorganic carbon was also observed in the sample. Samples collected over the same period from the top trap at GovDock did not show similar increases for mercury or inorganic carbon (H. Biberhofer, Environment Canada, pers. comm.). These results could reflect the presence of a submerged plume of stormwater runoff at this station and sampling period (H. Biberhofer, Environment Canada, pers. comm.). A storm sewer discharges into the embayment in which the GovDock site was located (Figure 76).

Mercury concentration also exceeded the severe effect level at the RAPReef site (top trap, May 22, 1998). There was no specific precipitation event during sampling interval 2 (November 7, 1997 to May 22, 1998) that might explain this. The interval does include spring thaw and the site is located 600 m downstream of the snow dump and the Brookdale Avenue combined sewer overflow. However, no corresponding peak in mercury concentrations was observed at this site in the spring 1999 samples.

The mercury enrichment observed at RAPReef in May 1998 could be related to surface runoff associated with construction of the EcoPark, which is situated at the waterfront immediately upstream. Heavy construction work involving extensive regrading for the Rotary Eco Garden Park occurred in January-February 1997. Additional grading was done in the summer of 1997 and the stream system discharging to the St. Lawrence River was opened in the spring of 1998. Although measures were taken to prevent soil erosion into the river during construction (hay bales and stream closure) there could have been soil releases into the stream or river during any of the three time periods Jan/Feb 1997, summer 1997 and spring 1998 (N. Levac, City of Cornwall, pers. comm. to K. Columbus, MOE, Eastern Region).

2.5 Contaminant Dispersion Modelling

2.5.1 Nettleton (1999)

In April 1999 Peter Nettleton, MOE completed a modelling assessment of effluent plume dispersion and related sediment impacts from point source discharges in Cornwall (Nettleton 1999). The purpose was to delineate the contaminant plumes associated with past discharges from major Cornwall point sources and their likely contributions to long-term concentrations of mercury and zinc in riverbed sediment along the Cornwall waterfront. Only the north channel of the St. Lawrence from approximately the upstream end of Cornwall Island to about the downstream end of Ile Saint Regis and Ile Jaune was modelled during this exercise.

An updated version of MOE's KETOX model (Version 5.2) was used which can graphically present modelled impacts on water, sediment and aquatic biota using geographic information system (GIS) mapping software. Details on the modelling procedures are provided in Nettleton (1999) and much of the basic model application and calibration is discussed in Nettleton's (1996) Cornwall Municipal-Industrial Strategy for Abatement (MISA) Pilot Site modelling component report.

The parameters used for point source loading inputs were average flow and average concentrations of mercury and zinc based on the 1989-90 twelve month MISA monitoring for Domtar, ICI Canada, ICI Conpak, Cornwall Chemicals and Courtaulds. Similar parameters were used for the Cornwall Water Pollution Control Plant (WPCP) based on 1987 MISA data. Point source values used in the modelling exercise are provided in Table 27. Please note that Domtar, ICI Canada, ICI Conpak and Cornwall Chemicals all discharged through a common diffuser referred to here as the Domtar/ICI diffuser. As well, the Courtaulds shore based and diffuser discharges referred to throughout this section were active prior to 1992.

Nettleton (1999) first delineated the water column plumes from all sources. Three basic loading scenarios were then evaluated, each using the same river background concentrations and long-term average total river flow and stage conditions (Nettleton 1996). The loading scenarios were:

(1) Use of average loads discharging from <u>all</u> point sources simultaneously. This represents the best estimate of actual combined impacts within the river (with respect to river background conditions), from all key point sources;

(2) Use of <u>only</u> the Domtar, ICI Canada, ICI Conpak and Cornwall Chemicals average loads (i.e., with zero loading from Courtaulds and the Cornwall WPCP). This scenario estimated the contributions of the impact within the river associated with the Domtar/ICI diffuser discharge only;

(3) Use of <u>only</u> the Courtaulds average loads (i.e., with zero loading from Domtar, ICI Canada, ICI Conpak and Cornwall Chemicals diffuser and Cornwall WPCP). This scenario estimated the contributions of the impact within the river associated with the discharges from Courtaulds only.

No scenario based solely on loadings from the Cornwall WPCP was run because any influence from this source was dominated by the industrial discharges. Furthermore, the WPCP diffuser is downstream from the areas of highest sediment contamination.

The above three loading scenarios were run for the following types of bottom sediment:

average ("fraction of fines"=0.54), coarse ("fraction of fines"= 0.24), fine ("fraction of fines"=0.85) and spatially-variable bulk sediment. "Fraction of fines" refers to the percentage of fine-grained sediment, defined as sediment particles <63 μm in diameter (i.e., silt or clay). "Spatially-variable" bulk sediment was defined using a combination of data from the KETOX model of bottom sediment type and actual field data on sediment type. Spatially variable sediment most closely resembles the actual distribution of sediment type on the river bottom at Cornwall. The derivation of each of the bulk sediment types is provided in Nettleton (1999).

For brevity, this review includes only the following modelling results from Nettleton (1999):

I. delineation of water column contaminant plumes from all point sources simultaneously;
II. impacts on "average" bed sediment from all point sources simultaneously;
III. impacts on "average" bed sediment from individual point sources; and
IV. impacts on "spatially-variable" bed sediment from all point sources simultaneously.

For a complete analysis of all scenarios for water column plume delineations and impacts on all sediment types please refer to Nettleton (1999).

Table 27. Measured flow / concentration data for individual effluents & calculated combined source parameters. Source: Nettleton (1999).

Sewer	Flow (m3/d)	Flow (cfs)	Chemical concentration (ug/L) Hg	Zn	Cu	Pb	Chemical loading rate (kg/day) Hg	Zn	Cu	Pb
Domtar diffuser:										
Domtar	129,070	52.75	0.02	44.9	8.33	n/a	0.0026	5.7964	1.0751	0.0000
ICI Canada	4,010	1.64	16.8	29.2	20.1	47.2	0.0674	0.1171	0.0806	0.1893
ICI Conpak	21	0.01	3.15	187.0	679	728	0.0001	0.0039	0.0143	0.0153
Cornwall Chemicals	655	0.27	3.06	117.8	16.8	n/a	0.0020	0.0772	0.0110	0.0000
Combined diffuser	133,756	54.67	0.54	44.8	8.8	1.5	0.0720	5.9946	1.1810	0.2046
Courtaulds:										
Acid sewer (diffuser)	5,426	2.22	10.26	52,384.9	61.2	238.0	0.0557	284.2329	0.3321	1.2914
Alk. sewer (diffuser)	1,524	0.62	4.73	2,459.9	26.2	91.9	0.0072	3.7488	0.0399	0.1401
Storm	10,077	4.12	0.27	560.4	32.3	11.8	0.0027	5.6470	0.3255	0.1189
Acid recovery	31,604	12.92	0.18	1,242.6	24.1	12.0	0.0057	39.2701	0.7616	0.3792
CS2 sewer	6,447	2.64	0.18	1,489.6	8.6	12.0	0.0012	9.6032	0.0554	0.0774
Caravelle sewer	11,167	4.56	0.14	1,410.5	7.0	13.5	0.0016	15.7506	0.0782	0.1508
all 6 Court. sewers	66,245	27.08	1.12	5,408.1	24.0	32.6	0.0740	358.2527	1.5927	2.1577
4 Court. shore sew.	59,295	24.24	0.19	1,185.1	20.6	12.2	0.0111	70.2709	1.2207	0.7263
2 Court. diffusers	6,950	2.84	9.05	41,437.3	53.5	206.0	0.0629	287.9817	0.3720	1.4314
A.R.+CS2+Caravelle	49,218	20.12	0.17	1,313.0	18.2	12.3	0.0084	64.6239	0.8952	0.6074
CS2 + Caravelle	17,614	7.20	0.15	1,439.5	7.6	13.0	0.0027	25.3538	0.1336	0.2281
Cornwall WPCP:	48,712	19.91	0.04	30.0	10	20	0.0019	1.4613	0.4871	0.9742
All Point Sources:	*248,713*	*101.65*					*0.1480*	*365.7085*	*3.2608*	*3.3364*

Notes:
1. *The Domtar diffuser and Courtaulds' individual effluent data are based upon average 1989-90 MISA monitoring results.*
2. *The Cornwall WPCP data are based upon average 1987 MISA measurements.*
3. *The underlined sources / parameters are those used in this modelling assessment.*

I. Delineation of water column contaminant plumes from all sources

Figures 78 and 79 show expected mercury concentrations in the water column plumes with all point sources discharging simultaneously, based on the MISA monitoring data. Mercury concentrations range from an estimated background value of 10 ng/L to about 43 ng/L in the vicinity of Courtaulds' near shore outfalls.

Figures 80 and 81 show similar concentration contour plots for zinc. Zinc concentrations range from an estimated background of 2.0 µg/l to about 251 µg/l in the vicinity of Courtaulds' near shore outfalls.

When interpreting the water plume delineation results in Figures 78-81 it is important to understand that for KETOX modelling purposes the effluents are assumed to be "instantly mixed" at the point of discharge and, for this reason, the model tends to under predict water plume concentrations. The KETOX model is not designed to simulate recirculation zones or back eddies, as occur along a portion of the Courtaulds near shore area (see Section 2.1). The implication of this limitation is that the KETOX model will predict a plume that is narrower than the actual plume from the combined near shore discharges of Courtaulds (Nettleton 1999).

II. Impact upon "average" bed sediment from all sources

Nettleton predicted "potential" impacts upon bed sediment assuming a uniform "average" river bottom sediment type over the entire waterfront area. Four plots depicting the areas of greatest impact with a high degree of resolution are included here (Figures 82 and 83 for mercury and Figures 84 and 85 for zinc).

The predicted mercury concentrations in "average" bed sediment ranged from an estimated background concentration of 0.13 mg/kg to about 0.53 mg/kg in the vicinity of Courtaulds' near shore outfalls. Zinc concentrations ranged from an estimated background value of 52 mg/kg to nearly 6500 mg/kg in the vicinity of Courtaulds' near shore outfalls (Nettleton 1999).

III. Impact upon "average" bed sediment from individual sources

The area between Windmill Point and Pilon Island is the zone of greatest mercury and zinc impact on bed sediment. To estimate the impact associated with each of the sources, separate model simulations were made using the following scenarios:

(1) only the average 1989-90 MISA monitoring loads through the Domtar/ICI diffuser (i.e., the loads from Domtar, ICI Canada, ICI Conpak and Cornwall Chemicals) exist;

(2) only the average 1989-90 MISA monitoring loads from all Courtaulds sources exist.

For both of these loading scenarios, the same general river background concentrations of mercury and zinc as used previously were assumed to exist (10 ng/L and 2.0 ng/L respectively). Figures 86 and 87 (mercury) and Figures 88 and 89 (zinc) show the results of the simulations. The location of point source discharges (near shore outfalls and diffusers) is shown on each map as a small open black circle. Sediment impacts of mercury and zinc from individual point sources as predicted by the model (Nettleton 1999) are summarized below.

Mercury results:

- The contribution of mercury impacts along the north shore in the vicinity of Courtaulds and downstream for about 1.5 km would be dominated by Courtaulds' near shore outfalls (compare Figures 86 and 87). In the region adjacent to the shore within about 200 m of the near shore outfalls, the approximate contributions of mercury to the "average" bulk bed sediment would be: 0.12 mg/kg from the general river background, 0.02 mg/kg from the Domtar/ICI diffuser and 0.16-0.35 mg/kg from Courtaulds' near shore outfalls.

- Further offshore, in the region under the plumes from the two offshore Courtaulds diffusers and up to about 200 m downstream from the diffusers, there is much more river mixing and the point source contributions are not large compared with the river background contribution. Approximate contributions of mercury to the "average" bed sediment matrix in this region would be: 0.12 mg/kg from the general river background, less than 0.01 mg/kg from Domtar/ICI diffuser and about 0.03 mg/kg from the Courtaulds diffusers.

Zinc results:

- Contributions from Courtaulds sources dominate the impact of zinc on bed sediment from Windmill Point to Pilon Island (compare Figures 88 and 89). Looking at an area within about 750 m of Courtaulds, the predicted contributions of zinc to the "average" bed sediment are: 52 mg/kg from the general river background, 3 mg/kg from the Domtar/ICI diffuser and 765-6445 mg/kg from Courtaulds' near shore outfalls.

- Further offshore, under the plumes from the two offshore Courtaulds diffusers and up to about 200 m downstream from the diffusers, the approximate zinc concentrations expected in "average" bulk bed sediment would be contributed as follows: 52 mg/kg from the general river background, about 1 mg/kg from the Domtar/ICI diffuser and about 177-273 mg/kg from Courtaulds' diffusers.

IV. Impact upon "spatially-variable" bed sediment from all sources

To assess the impacts of point sources upon bottom sediments that have variable grain size, Nettleton (1999) mapped spatially variable bulk sediments using the KETOX model and existing field data on sediment characteristics (Kauss et al. 1988; Anderson 1990; Rukavina 1997). Modelling results for spatially variable sediment (Nettleton 1999) are shown in Figures 90 and 91 (mercury) and Figures 92 and 93 (zinc) and are summarized below.

Mercury results:

- In the yellow and orange zone shown in Figure 90, mercury concentration in sediment is elevated above that expected, due mainly to background exposure with some contribution from Domtar/ICI. These zones represent sediment mercury concentrations of 0.20-0.30 mg/kg (yellow) and 0.30-0.40 mg/kg (orange).

- The orange and red areas within the large zone between Windmill Point and Pilon Island and in the smaller zone to the east of Pilon Island (Figure 91) are due to point source contributions from both the Domtar/ICI diffuser and Courtaulds' shore based outfalls. The orange and red zones immediately adjacent to the Courtaulds site shoreline would be largely caused by Courtaulds' discharges.

- The orange area within the zone south of Pilon Island reflects more the very "fine" bed sediment in this region than the point source exposure levels (Nettleton 1999).

Zinc results:

- The maximum expected concentration of zinc in bed sediment associated with the river background exposure is about 70 mg/kg.

- The zone in Figure 92 approximately 1 km downstream of the Domtar/ICI diffuser has zinc elevated only slightly above that expected, due only to the river background exposure. (The estimated zinc levels in this area are in the 65-70 mg/kg range, slightly above the value of 60 which would be expected for the bed sediment type).

- All zones in Figure 93 colored green, yellow, red, brown or black represent predicted sediment concentrations of zinc greater than 120 mg/kg. As such they are clearly associated with Courtaulds' discharges and above the LEL for zinc (120 mg/kg).

- The entire near shore portion of the zone between Windmill Point and Pilon Island (over 2 km in length as shown in Figure 93) is significantly impacted by Courtaulds' shore based discharges. Within this zone, there is a smaller area (coloured red, brown and black) about 700 m long in which the SEL for zinc (820 mg/kg) would be exceeded.

- The far field zone east of Pilon Island (coloured green on Figure 93) is largely above the LEL of 120 mg/kg, due to both the near shore and offshore (diffuser) plumes from Courtaulds.

- The zone to the south of Pilon Island reflects impacts from Courtaulds' offshore diffusers in its northern portion and higher levels from the interaction of river background exposure and very "fine" sediment in its southern portion (Figure 93).

Overall Conclusions

Nettleton drew the following conclusions from the various modelling simulations. Direct quotes are from Nettleton (1999):

- Delineation of plumes within the water column of the St. Lawrence River showed that "most of the chemical mass flux from Courtaulds' shore based discharges travels north of Pilon Island and the Colquhoun Islands. Most of the mass flux associated with Courtaulds' diffusers and the Cornwall WPCP diffuser travels south of Pilon Island but north of the Colquhoun Islands. The chemical mass flux from the Domtar/ICI diffuser would be approximately split in its flow past Pilon Island, but would largely (around 95%) pass to the north of the Colquhoun Islands."

- "Mercury concentrations in the water column were likely less than 5 ng/L above the river background levels, except along a limited near shore area of the Courtaulds waterfront where they would have been more pronounced. However, the PWQO would not have been exceeded" in any of the plumes.

- "The concentrations of zinc in the plumes from Courtaulds' discharges (forming both a combined near shore plume and a combined offshore plume) would have been significantly above the river background concentration. There was likely a significant zone within the combined near shore plume from Courtaulds, of about 2 km in length, where the interim PWQO (of 20 µg/L) would have been exceeded." (Background zinc concentration estimated at 2.0 µg/L; maximum predicted zinc concentration of 251 µg/L.)

- The modelling results of long-term impact associated with steady effluent loading rates (similar to the 1989-90 twelve month MISA monitoring values) on "average", "fine" and "coarse" bulk sediments "indicate that mercury seems to be more sensitive than zinc to variation in the bed sediment type. This may be due in part, to greater distribution of zinc among the differing grain sizes."

- "The relative contribution of the long-term impacts associated with the Domtar[/ICI] diffuser and Courtaulds' discharges, upon "average" bulk bed sediment, were delineated in the Windmill Point to Pilon Island portion of the river. The Courtaulds shore based outfalls would have contributed the most significant portion (80% or more) of the mercury impacts along its waterfront, particularly within about 700 m of the upstream end of the Courtaulds facility. Virtually all of the discernible impacts of zinc (well over 95%) are due to the various discharges from Courtaulds."

- Without the modelling analysis on spatially variable sediment, "looking solely at the field measured data...it would be difficult to accurately assess and quantify the impact associated with point source discharges. This is particularly so when the chemical's accumulation within bed sediment is highly sensitive to the "fraction of fines" (e.g., such as mercury), and the exposure levels from the discharge plumes are not excessively greater than the river background concentration. Under this circumstance, it would be easy to associate impacts with point source discharges, when in reality, they may be more reflective of sediment pockets possessing larger concentrations of "fines". Conversely, areas of higher impact concentrations further downstream from a key point source, may be arbitrarily explained as being solely due to "finer" bed sediment, when in reality, this may only partially account for the higher levels."

- "Having an intensive, bed sediment characterization measurement grid is highly desirable for quantifying the actual long-term impacts of point source discharges upon bed sediment. Without it, "potential" impact results must be used, which are derived for an assumed, uniformly spatially-distributed bed sediment type.

- "Potential" bed sediment impact results may be beneficial in setting water-quality based effluent loading limits based on avoidance of unacceptable impacts upon downstream bed sediment.

2.5.2 Kauss et al. (1988)

Kauss et al. (1988) reported the results of mixing zone modelling of point sources along the Cornwall waterfront. Modelling indicated that discharges from Domtar/CIL[later ICI]/Cornwall Chemicals and Courtaulds would be confined to the north shore, whereas the Cornwall WPCP discharge would mainly flow south of Pilon Island and mix with US discharges further downstream. The north shore plume dispersion pattern was attributed to two factors: (1) the strong lateral river velocity, and (2) the fact that 40% of the outfall diffuser ports were located close to the shore in shallow water (≤0.8 m), preventing efficient mixing with river water.

Using phenolics loading data from a 1979 survey, the area with concentrations >1 µg/L was predicted to extend about 10 km down river from the Domtar/CIL/Cornwall Chemicals diffuser, with a maximum lateral width of 190 m. The model predictions coincided well with 1980 field survey results (Kauss et al. 1988).

2.6 Upstream Loading of Mercury

A portion of the contaminants in the Cornwall section of the St. Lawrence River originates from the Great Lakes basin and upper St. Lawrence River. Upstream inputs of mercury are discussed here.

In order to calculate meaningful upstream loadings of mercury to the river at Cornwall, it is necessary to use total mercury concentrations in river water (dissolved plus particulate) that were determined using a sufficiently low method detection limit. Low level mercury techniques were used to analyze samples collected from the St. Lawrence River immediately upstream of Cornwall in 1988 (unpublished EC/MOE data) and in a 1995-1996 mass balance study of the St. Lawrence River by Cossa et al. (1998). Table 28 compares upstream loadings of mercury to the Cornwall waterfront area calculated from these data.

Table 28. Calculated average daily upstream mercury loading to the north channel of the St. Lawrence River at Cornwall, 1988 and 1995-1996.

Year sampled	1988[1]	1995-1996[2]
Total flow of St. Lawrence River (annual mean)[3]	6850 m^3/s	7543 m^3/s
Portion flowing through north channel	34%	34%
Mean mercury concentration	0.6 ng/L	dissolved 0.27 ng/L particulate 218 ~~µg/g~~[4] ng/g
Analytical detection limit	0.1 ng/L	dissolved 0.01-0.11 ng/L particulate 2.1-21.1 ng/g
Number of samples Sampling time	2 replicate samples summer 1988	39 samples May 1995 to end Sept 1996
Sample location	downstream of Moses-Saunders Dam but upstream of Cornwall point sources	approximately 1 km upstream of Moses-Saunders Dam in the main channel
Annual mercury load (dissolved and particulate) at Moses-Saunders Power Dam	n/a	1~~12 kg/day~~ 116 kg/year
Calculated upstream mercury loading to north channel at Cornwall	121 g/d	~~104.3 g/d~~ 108.2 g/day

1 unpublished EC/MOE data
2 Cossa et al. (1998)
3 measured at Moses-Saunders Power Dam (St. Lawrence River Committee on Gauging 1985-1995)
4 mean total suspended solids=1.0 mg/L (+/- 0.6) in 1995-1996 (Cossa et al. 1998)

Rain R. would require an annual ave. mean Hg conc. of 0.25 ug/l

These figures are rough estimates only because of the variability in results for mercury in water samples. Low level mercury analysis of St. Lawrence River at Cornwall water samples by MOE Dorset (method of Mierle 1990) showed mercury concentrations ranging from 0.3-0.9 ng/L in 1988 samples and 2-3.5 ng/L in 1991 samples. Although the reason for the difference is unknown, it may have been the result of seasonal differences in sample collections. In 1988 samples were collected in the summer, whereas 1991 samples were collected in March (Richman 1994).

2.7 Air Deposition of Mercury

ICI Forest Products (ICI) operated a chlor-alkali plant at Cornwall from 1935-1995, releasing mercury directly to the atmosphere. MOE monitored the impact of air emissions of mercury from ICI on the local environment (Dixon and Emerson 1994) and assessed the associated human health risk (Fleming et al. 1995). MOE also used data from Environment Canada audit monitoring of the ICI facility to model the potential impact of mercury emissions on the local environment (Ladouceur 1994).

The principal source of mercury from ICI was the cell room, which accounted for over 99% of the mercury emitted to the atmosphere (Ladouceur 1994). Within the cell room, mercury evaporated from contaminated equipment and surfaces, from spills and the mercury-cells. The vaporized mercury dispersed throughout the room and eventually exhausted to the atmosphere through roof monitors (exhaust vents). As part of their compliance testing program, Environment Canada sampled mercury at ICI's roof monitors in 1993. During the compliance monitoring period, mercury emissions averaged 2.85 g/day/1000 kg Cl_2 produced and ranged from 1.54-4.62 g/day/1000 kg Cl_2 produced. The Federal standard for mercury emissions is 5 g/day/1000 kg of Cl_2 production (Ladouceur 1994).

Based on ICI (Cornwall) production figures and Environment Canada inspection monitoring data, the average daily loading to the atmosphere was calculated as 434 g mercury (range 235-704 g/day) in 1993. In contrast, ICI estimated that mercury loadings to air from all sources at the Cornwall plant from 1982-1991 averaged 220 g/day (Ladouceur 1994). Losses of mercury from other on-site sources besides the cell room roof vents would add to the amount of mercury lost to the atmosphere. However, if the cell room accounted for more than 99% of the mercury emitted to air, the amount of mercury from the additional on-site sources at ICI, Cornwall would be small (Ladouceur 1994).

Between 1987 and 1991, MOE sampled surface soil (0-5 cm depth), tree foliage and moss bags (Fleming et al. 1995). Surface soil was collected in 1991 at 22 locations in Cornwall. Five sites near (within 100-360 m of) the ICI Forest Products Ltd. plant had mercury concentrations above the Upper Limit of Normal (ULN) for urban soils (0.5 µg/g). Mercury also exceeded the ULN for urban soils at two other stations located 720 and 1880 m away from the plant. Surface soil sampling results are summarized in Table 29.

Table 29. Mercury concentrations (µg/g) in surface soil (0-5 cm depth) at 22 locations in Cornwall, Ontario, 1991 (summarized in Fleming et al. 1995). Note: these sites were <u>not</u> located in backyard gardens.

Sampling date	Soil mercury concentration (µg/g) dry wt	
	Mean	Range
May 1991	0.45	0.06-1.75
August 1991	0.5	0.04-1.95

In 1993, MOE sampled garden soil and vegetables at ten backyard garden locations. Seven sites were located in the residential area immediately east of ICI; one was located on Edythe Avenue, just west of ICI; and two distant backyard gardens west of ICI were used as controls. Six of the 7 sites east of ICI had elevated mercury concentrations in soil (0.19-2.37

μg/g) relative to control sites (0.06-0.08 μg/g). Mercury concentrations exceeded the ULN for urban soils at 4 of the 7 sites east of ICI (Dixon and Emerson 1994). Results of backyard soil sampling are summarized in Table 30.

Table 30. Mercury concentrations (μg/g) in Cornwall, Ontario backyard garden soils, 1993 (summarized in Fleming et al. 1995).

Sampling date	Sites in vicinity of ICI (N= 8)		Control sites (N=2)	
	Mean	Range	Mean	Range
May 1991	0.84	0.05-2.30	0.07	0.06-0.07
August 1991	0.91	0.07-2.37	0.07	0.06-0.08

Backyard vegetables were also collected from the backyard soil sampling sites, washed and dried prior to mercury analysis. Mercury concentrations in leaf lettuce and beet tops were higher at sites near ICI than at the control sites. Mercury concentrations in air samples were generally higher at locations downwind of ICI than at the upwind locations. For further details of the backyard vegetables study please see Fleming et al. (1995).

MOE's soils, garden vegetables and air data showed that mercury emitted from ICI was deposited locally (Fleming et al. 1995). Modelling of ICI mercury emissions by MOE supported this conclusion (Ladouceur 1994). Mercury emitted to the atmosphere from this facility was deposited on roads, homes and yards and, when the wind was from a northerly direction, onto the St. Lawrence River.

It is not possible to calculate the loading of mercury to the St. Lawrence River by direct deposition from the atmosphere because there is high variability in a number of factors (*e.g.*, wind direction and intensity, daily temperature and humidity, chlorine production) and sampling frequency and location have been inconsistent. Mercury deposited onto the ground can be carried in surface runoff into the St. Lawrence River. However, the actual amount of mercury loading from surface runoff is also difficult to predict since portions of this mercury may be incorporated into the soil matrices or taken up by vegetation. Monitoring of stormwater discharges may provide some insight into the degree to which local mercury emissions to air affect stormwater quality and are transported to the St. Lawrence River in surface runoff. For this information please refer to the Historical Discharges Report (Columbus 2000).

3. BIOTA

Studies of sediment contaminants and biota in the Cornwall area are reviewed in the following sections on sport fish, young-of-the-year spottail shiners and benthic invertebrates. It is important to note that mercury, the main contaminant of concern, may have little or no demonstrable effect on short term survival or growth of organisms but can biomagnify as it is passed up the food chain. Therefore, although this review separately discusses each type of organism studied, the biological findings should be considered together as a whole. This will assist in interpreting the biological data with respect to implications for sediment remedial requirements.

3.1 Mercury in Sport Fish

Please note that the sport fish section is still in draft and a revised version may be provided later as an addendūm.

This section reviews existing interpretations of sport fish contaminant (mercury) trends in the St. Lawrence River from the Thousand Islands area to Lake St. Francis. With respect to fish, which are highly mobile, "Lake St. Francis" should be taken to mean the entire area from the Moses-Saunders Power Dam to the Beauharnois Dam at Valleyfield, Quebec. The Moses-Saunders Power Dam completely spans the north channel of the river just upstream of Cornwall. In the same area the St. Lawrence Seaway Snell Lock blocks the south channel. This physically separates Lake St. Francis fish from the upstream sampling sites, making upstream vs. downstream comparisons possible for the Cornwall area.

Fish are the only biota for which a long-term data set on mercury concentrations is available for the St. Lawrence River at Cornwall. The MOE and MNR Sport Fish Contaminants Program has generated a data set for contaminants in northern pike, walleye, yellow perch and white sucker from the early 1970s to the present. Fish are collected annually or biannually from several regions of the St. Lawrence River (Thousand Islands, Brockville, Lake St. Lawrence and Lake St. Francis) as shown in Figure 94. Years of sampling for each species and region are listed in Table 33B.

Please note that all sport fish tables are placed together at the end of Section 3.1.6, except for Table 31 which is incorporated into the text.

3.1.1 Introduction

Two different approaches have been used to analyze the MOE/MNR sport fish contaminants database for the St. Lawrence River to determine temporal trends and spatial/regional differences in mercury concentrations. Dreier et al. (1997) compared mercury concentrations in standard length fish as described below and Lalonde (1999) used various statistical methods to examine regional and temporal differences in mercury concentrations. Their conclusions are discussed and compared in Sections 3.1.2 and 3.1.3. The implications with respect to RAP delisting criteria for sport fish consumption are discussed in Section 3.1.5.

The sport fish contaminants sampling program was not designed to compare mercury concentrations in fish over time or in different parts of the river. The data were collected specifically to compile the *Guide to Eating Ontario Sport Fish*. Thus length but not age data are available for all fish analyzed.

Fish age is an important factor for a bioaccumulating contaminant such as mercury which gradually increases in concentration with an increase in duration of exposure (i.e., increase in age). This complicates attempts to detect trends in mercury contamination using the MOE/MNR sport fish contaminants database. Temporal and spatial comparisons of mercury trends in fish are most meaningful if made on fish of the same age class. Fish length may be used as an approximator of fish age.

In the absence of age data, Dreier et al. (1997) used fish length as a surrogate parameter. Since length is related to age, mercury data for each species of fish were standardized to a specific length in order to compare tissue concentrations between years and between sampling sites. Dreier et al. (1977) used the following standard lengths: 50 cm walleye, 55 cm northern pike and 25 cm yellow perch.

Length standardization assumes that for all sampled populations of a single species, mercury concentration increases with length in the same proportions. But this is not necessarily so. Several factors affect the relationship between mercury and length, any or all of which can vary between populations of the same species:

- rate of growth;
- amount of mercury exposure, which depends on:
    - degree of mercury contamination in immediate environment,
    - how high on the food chain the fish population is feeding (higher=more mercury exposure);
- genetic predisposition to accumulate mercury at a higher or lower rate.

With respect to the first factor above, the approximate age of a "standard length" fish of a particular species may not be the same in all three St. Lawrence River sites. For instance, whereas a 55 cm northern pike in Lake St. Lawrence would be 1-2 years old, the same length pike in the Thousand Islands or Lake St. Francis would be 2-3 years old (M. Eckersley, MNR, pers. comm. cited in Dreier et al. 1997). Table 31 shows geographical differences in age-length relationships for yellow perch, walleye and northern pike of the St. Lawrence River, Ontario.

Table 31. Age-length relationships in "standard size" fish in the St. Lawrence River (M. Eckersley, MNR, pers. comm. cited in Dreier et al. 1997).

Standard length and species	Age of standard length fish (years)		
	Thousand Islands	Lake St. Lawrence	Lake St. Francis
25 cm yellow perch	rarely reach 25 cm, even at age 8 or 9	7-10 (mostly 7 and 8)	6-12 (mostly 6, 7 and 8)
50 cm walleye	probably same as Lake St. Francis but data too sporadic to be sure	63	3-5 (mostly 4)
55 cm northern pike	2-3	1-2	2-3

The Dreier et al. (1997) and Lalonde (1999) analyses of the existing MOE/MNR Sport Fish Contaminants database lead to some overall conclusions:

- More samples are required, with age data included, in order to fully understand patterns of mercury contamination in sport fish of the St. Lawrence River.
- It is not possible, at the present time, to draw any definite conclusions about the relationship between changes in local point source emissions of mercury and mercury contamination of Lake St. Francis sport fish.
- The currently available data can be used to do the following:
  - Compare current mercury concentrations with consumption restriction guidelines (see Section 3.1.5), and
  - Compare concentrations in fish downstream of local point sources (Lake St. Francis) with upstream concentrations (Thousand Islands, Brockville or Lake St. Lawrence) for each species (see Section 3.1.2). This will indicate whether the short-term delisting criterion for the RAP Area of Concern has been met for the species in question (see Section 3.1.5).

Spatial comparisons should take into account between-region differences in the relationship between mercury and length. Direct comparisons of pooled data (comparing mean mercury concentration for all lengths collected in a given year) can only be made between regions for which the mercury:length relationship is not significantly different. However, a significant regional difference in mercury:length relationship is in itself an important observation because it could indicate a difference in exposure conditions (ambient concentrations of mercury).

3.1.2 Regional Differences in Mercury Concentration

Sport fish contaminants data exist for four regions of the St. Lawrence River in Ontario: Thousand Islands, Brockville, Lake St. Lawrence and Lake St. Francis as shown in Figure 94. As discussed earlier, the Moses-Saunders Power Dam separates fish in the Cornwall area (Lake St. Francis) from the three upstream collection sites in the river.

Lalonde (1999) statistically analyzed the 1991-1995 sport fish data to examine regional differences in mercury concentrations. For detailed results of the statistical analysis please refer to Table 2 in Lalonde (1999). The main conclusions are outlined below and compared with the spatial trends identified by Dreier et al. (1997).

Please note that the statistical significance of observed differences cannot be ascertained when comparing mercury concentrations standardized to a specific length fish. Therefore the conclusions of Dreier et al. (1997) are purposely described here using terms such as "similar to" and "appeared to be different/higher/lower than."

Figures 95A, 95B and 95C show length-standardized mercury concentrations by region and year (mid-1970s to mid-1990s) for walleye, northern pike and yellow perch (A. Hayton, MOE, Environmental Monitoring & Reporting Branch, data files). Error bars are not available, and therefore not shown, because the values graphed were obtained from the regression of concentration vs. length. Table 32 provides data associated with the regressions including R^2 values and number of fish sampled.

The conclusions of Lalonde (1999) and Dreier et al. (1997) regarding regional differences in mercury concentration for each species are compared below:

NORTHERN PIKE

- The relationship between mercury concentration and length was not significantly different in any of the four regions studied (Lalonde 1999). It was therefore considered valid to compare pooled data (i.e, all fish collected in a given year regardless of length) among all four regions.
- Lalonde's (1999) analysis showed that mercury concentrations in the Thousand Islands, Brockville and Lake St. Francis were not significantly different from each other. However, the lack of difference between Thousand Islands and Brockville may have been due to the low power of the test (too few samples to detect a significant difference if one existed) (Lalonde 1999).
- Lalonde (1999) concluded that mercury concentrations were significantly lower in Lake St. Lawrence pike than any other region. Likewise, Dreier et al. (1997) reported that Lake St. Lawrence concentrations appeared to be lower than Thousand Islands and Lake St. Francis, which were both similar to each other.
- The reason for the lower mercury concentration in Lake St. Lawrence is unknown but it could be explained by a faster growth rate for pike in this region which is indicated in Table 31 above. In rapidly growing fish, the mass of mercury accumulated is diluted by the greater mass of tissue ("biomass dilution") so that mercury concentration will be higher in a rapidly growing population compared with a population that grows more slowly (M. Whittle, Fisheries and Oceans Canada, pers. comm.).

WALLEYE

- Lalonde (1999) found that walleye mercury concentration increased with length significantly more in Lake St. Francis than in Thousand Islands and Lake St. Lawrence. Lake St. Francis walleye data was therefore not included in spatial comparisons of mercury concentration (all lengths combined) by year.
- The significantly different mercury:length relationship in Lake St. Francis is an important observation although the reason for it is unknown at present. It could be due to greater mercury exposure in Lake St. Francis or a different growth rate in Lake St. Francis fish or a combination of both. Mercury enrichment in north shore Lake St. Francis sediments has been demonstrated (Lorrain et al. 1993; Lepage 1999) as described in Section 2.3.3.
- An examination of median total fish length and the range in total lengths for each population (Thousand Islands, Lake St. Lawrence, Lake St. Francis) shows that differences between populations are modest compared to the large difference in tissue mercury concentration (two to three times higher for Lake St. Francis). This is especially evident when the range in mercury concentration and total length are compared on an annual basis (L. Richman, MOE, Environmental Monitoring & Reporting Branch, pers. comm.).
- Dreier et al. (1997) reported that mercury concentrations appeared to be highest in Lake St. Francis walleye but that statistical analysis was required to confirm this. Lalonde (1999) also stated that concentrations were highest in Lake St. Francis and above consumption guidelines.
- Mercury concentrations in Thousand Islands and Lake St. Lawrence walleye were not significantly different but this may have been due to the low power of the test (too few samples) (Lalonde 1999).
- No data were available for walleye in the Brockville region.

The significantly higher increase in mercury concentration with length in Lake St. Francis compared with upstream walleye is noteworthy because it may be due to higher mercury exposure in Lake St. Francis. This would be consistent with the mercury enrichment of sediment in the northern half of Lake St. Francis discussed in Section 2.3.3.

It cannot be determined from the Sport Fish Contaminants Program data set how mercury concentrations in Lake St. Francis walleye statistically compare with walleye upstream of the Cornwall area. This comparison requires more samples, with age data included for each fish.

YELLOW PERCH

- The mercury:length relationship was not significantly different for all regions except Brockville (Lalonde 1999). Brockville yellow perch were therefore not included in Lalonde's (1999) spatial comparisons by year (all lengths combined).
- Both Lalonde (1999) and Dreier et al. (1997) reported that mercury concentrations were similar in Lake St. Francis, Lake St. Lawrence and Thousand Islands yellow perch in the 1990s. The lack of a significant difference between Thousand Islands and Lake St. Lawrence reported by Lalonde (1999) may have been due to the low power of the statistical test (too few samples to see a difference if there was one).

WHITE SUCKER

- Lake St. Lawrence was not be included in Lalonde's (1999) comparison because the relationship between mercury and length was significantly different from the other three regions. In fact, mercury concentrations did not vary with length in white suckers from Lake St. Lawrence, but this may have been because sample size and length of fish were smaller than in other regions.
- For the other three regions, mercury concentrations were significantly higher in Brockville, followed by Lake St. Francis then Thousand Islands (Lalonde 1999).
- Dreier et al. (1997) did not include white sucker in their analysis.

3.1.3 Temporal Trends in Mercury Concentration

The main conclusions of Lalonde (1999) and Dreier et al. (1997) on temporal trends are discussed separately below.

3.1.3.1 Temporal Trends (Lalonde 1999: MSc Thesis)

Lalonde (1999) analyzed temporal trends in all species and regions for which there was a long term data set for the 1970s, 80s and 90s. Statistically significant between-year differences in mercury concentration are shown for each species and region in Table 33A and the results summarized below. The years for which sport fish contaminants data were available for each species and region are listed in Table 33B.

LAKE ST. FRANCIS

In *all species*, mercury concentrations appear to have decreased from the late 1970s to the early 1980s and remained fairly stable with minor fluctuations from then on.

LAKE ST. LAWRENCE

Northern pike concentrations have been stable all years sampled (1981-1995).
For *walleye*, Lalonde (1999) concluded that there were insufficient temporal trend data for a valid analysis.
Yellow perch concentrations decreased each sampling year from 1977-1993 (later years

were not analyzed).
White sucker followed the same pattern seen in Lake St. Francis, with mercury concentrations decreasing from the late 1970s to early 1980s, afterwards remaining fairly stable with minor fluctuations.

THOUSAND ISLANDS

In *northern pike*, mercury concentrations declined from a maximum in 1976 to a level that remained stable from the mid-1980s through 1995, the last year included in the analysis. Lalonde (1999) did not analyze data for *walleye, yellow perch* or *white sucker* in the Thousand Islands region because the data did not date back to the 1970s.

BROCKVILLE

The pattern for *northern pike* was different than in other regions. Mercury concentrations in the 1970s and 1980s were not higher than the 1990s. The 1992 concentration was significantly higher than both 1986 and 1994.
Walleye data were unavailable for the Brockville area (no samples collected).
For *yellow perch*, 1976 was significantly higher than all later years. 1985, 1986 and 1990 were higher than 1992, and 1992 was higher than 1994. In the Brockville area, mercury concentration appears to have been continually decreasing in yellow perch over the period from 1976 to 1994.
In *white suckers* concentrations were higher in 1976 than from 1983 onwards. 1985, 1986 and 1988 values were significantly higher than 1994.

3.1.3.2 Temporal Trends (Dreier et al. 1997: St. Lawrence RAP Stage 2 Report)

The length-standardized data used by Dreier et al. (1997) did not allow for comparisons of statistically significant between-year differences. Conclusions regarding temporal changes in mercury concentration were therefore tentative and requirements for further statistical analysis or additional sampling were identified. The discussion of temporal trends focused on the area downstream of local point sources of mercury (i.e., Lake St. Francis).

LAKE ST. FRANCIS

In *northern pike*, mercury concentrations appeared to decrease slightly from 1977 and 1978 to 1984. Values then remained stable until 1990, with the exception of a peak in 1989. Concentrations may have decreased in 1992-1994 relative to the mid-1980s but additional data and statistical analysis for the next few years is required to confirm this trend.
For *walleye*, mercury concentrations appear to have decreased from 1970 to 1982, remaining fairly stable thereafter except for a peak in 1992.
Yellow perch have shown only small decreases in mercury concentration since 1978. Concentrations were highest in 1977, decreased in 1978, then remained stable until 1990, with the exception of a possible increase in 1986 (statistical analysis required to determine significance) and a peak in 1989. A 1989 peak was also observed for northern pike. The 1994 concentrations appear lower than 1990 but additional sampling is required to confirm this trend.
White suckers were not included in the study.

THOUSAND ISLANDS and LAKE ST. LAWRENCE

Walleye mercury concentrations at Thousand Islands (1989-1993) and Lake St. Lawrence (1981-1993) have shown little change. In these regions, sampling did not date back as far as in Lake St. Francis.

3.1.4 1998 Data

There has been additional MOE/MNR Sport Fish Contaminants Program sampling since Dreier et al. (1997) and Lalonde (1999) reported on the data set. Preliminary data shows that for walleye in Lake St. Francis, mercury concentration in 1998 was in the range of concentrations observed throughout the 1980s and 1990s. The 1998 concentration may be higher than 1994 and 1991 values, although statistical analysis would be required to confirm this.

3.1.5 Status with respect to RAP Delisting Criteria

The stated long term goal of the St. Lawrence (Cornwall) Remedial Action Plan is to have *no consumption restrictions on any fish species in the AOC* (Dreier et al. 1997). Since this depends upon reduction or virtual elimination of mercury from the Great Lakes-St. Lawrence Basin as a whole, a short term delisting criterion was set as well.

In the short term, restrictions on sport fish consumption will be delisted as an impairment of beneficial use when: *contaminant levels in fish from the AOC are the same as or less than those in fish from the St. Lawrence River upstream of the Iroquois Dam* (Dreier et al. 1997). The Iroquois Dam is located upstream of Lake St. Lawrence and downstream of Brockville, so this comparison does not include Lake St. Lawrence fish.

Body burdens of mercury have shown a gradual reduction in some sport fish species in the RAP Area of Concern (Lake St. Francis) since 1970, likely in response to decreases in local emissions. However, there are still restrictions on consumption of some species (walleye, northern pike, smallmouth bass) due to mercury contamination. Total mercury concentrations in larger, older walleye (≥65 cm) and channel catfish (≥65 cm) still exceed Health Canada guidelines for restrictions on the consumption of fish such that no fish of this size and type should be consumed.

Based on the short term RAP delisting criterion which compares upstream and downstream mercury concentrations, it may be possible to delist for yellow perch in the area of concern (Lake St. Francis). Both Lalonde (1999) and Dreier et al. (1997) reported that upstream and downstream mercury concentrations are similar for this species. Based on the results of the two analyses, it may also be possible to delist northern pike, but additional samples should be collected to confirm this.

Fish collections that include age data are required in order to compare upstream and downstream mercury concentrations in walleye. Valid comparisons cannot be made using the current data set.

Using the existing walleye data, mercury uptake through time was compared for young vs. old walleye of Lake St. Francis (A. Hayton, MOE data). If concentrations in younger fish have decreased over time relative to older fish this would indicate a decrease in the ongoing mercury contamination of walleye. Figure 95D compares mercury concentration in a standard length walleye for each sampling year by three different length categories: (1) all fish collected that were <45 cm in length, (2) only the samples >45 cm in length, and (3) all of the samples collected (all lengths combined). Figure 95D shows no obvious difference in mercury concentration over time among the three categories compared.

3.1.6 Ion et al. (1997)

Ion et al. (1997) analyzed mercury and PCBs in St. Lawrence River yellow perch collected from ten sites in a 150 km stretch of the St. Lawrence River from the western border of Quebec to Trois Rivieres. Five geographical sectors of the river were sampled: Lake St-Francis, Lake St-Louis, Laprairie, Montreal East, Lake St-Pierre. Two sites were sampled within each sector, one on the north side of the river and one on the south. The objective of the study was to examine spatial variations in concentrations of mercury and PCBs within and between sectors and assess the usefulness of yellow perch as a biomonitoring tool.

Although no significant differences in mercury concentrations were detected among sectors or within any sector (i.e., north vs. south) this may well be because the sample size was very low. Only 5 fish were collected per site. According to Ion et al. (1997), power analysis showed that 80 fish per sector would be required to detect a 33% difference in mercury levels in yellow perch with 90% confidence that a significant difference had not been missed (Ion et al. 1997).

Table 32. Data associated with regressions of mercury concentration vs. fish length used to calculate mercury concentration in standardized length sport fish (see Figures 95A, 95B and 95C). Source: A. Hayton, MOE, Environmental Monitoring & Reporting Branch, data files, 2000.

R^2 = coefficient of variation of regression; N = number of fish analyzed; Standard value = mercury concentration (mg/kg) in fish of a standard length. Standard lengths used were: 60 cm northern pike, 45 cm walleye and 20 cm yellow perch.

Species	Location	Sample Date	R^2	N	Total Mercury (mg/kg)			Total Length (cm)			Standard value (mg/kg)
					Mean	Max	Min	Mean	Max	Min	
Northern Pike	Thousand Islands	27-May-1976	0.67	16	0.73	1.60	0.11	62.0	75.60	47.00	0.66
		01-Jan-1983	0.87	18	0.47	1.20	0.09	54.9	76.20	40.00	0.54
		01-Jan-1984	0.85	20	0.64	1.30	0.29	63.0	82.20	44.70	0.55
		15-Jun-1984	0.85	20	0.64	1.30	0.29	63.0	82.20	44.70	0.55
		01-Aug-1984	0.71	20	0.50	1.10	0.16	60.3	76.50	44.60	0.48
		04-Sep-1985	0.43	6	0.56	0.90	0.23	62.2	68.10	55.40	0.53
		06-Sep-1985	0.23	10	0.58	0.78	0.26	57.7	71.10	49.80	0.59
		22-Sep-1986	0.85	15	0.52	1.10	0.16	58.5	72.10	40.10	0.52
		26-Nov-1987	0.87	20	0.53	1.70	0.15	55.1	87.50	40.60	0.63
		15-Aug-1988	0.84	14	0.41	0.84	0.13	53.6	71.10	42.30	0.53
		22-Aug-1988	0.29	21	0.57	1.10	0.23	59.0	74.30	49.00	0.58
		01-Sep-1989	0.76	20	0.52	1.00	0.13	60.4	83.00	44.10	0.50
		01-Sep-1991	0.70	29	0.51	1.20	0.18	60.8	84.60	48.00	0.49
		30-Jul-1992	0.47	19	0.60	1.10	0.14	55.8	79.00	27.50	0.65
		01-Nov-1993	0.45	23	0.50	1.10	0.18	58.5	76.00	46.70	0.52
		03-Oct-1995	0.47	20	0.49	0.93	0.20	59.6	83.80	19.20	0.48
Northern Pike	Middle Corridor	01-Jan-1975	0.61	24	0.62	2.00	0.17	60.7	87.00	37.00	0.53
		01-Jun-1978	0.12	23	0.53	1.50	0.20	61.2	77.50	38.00	0.53
		01-Jan-1983	0.61	20	0.49	1.10	0.14	58.2	69.50	43.00	0.52
		01-Jan-1983	0.93	6	0.85	1.20	0.31	61.1	75.00	40.00	0.81
		20-Jun-1984	0.71	20	0.56	1.20	0.25	61.2	78.20	39.30	0.52
		21-Jul-1984	0.82	20	0.42	0.91	0.24	52.9	73.10	39.80	0.51
		09-Sep-1985	0.80	9	0.47	0.99	0.21	58.0	80.90	47.90	0.49
		24-Sep-1986	0.72	19	0.46	1.40	0.13	62.2	81.80	40.20	0.38
		25-Sep-1986	0.47	16	0.44	1.00	0.23	54.6	69.60	43.10	0.52
		18-Aug-1988	0.39	18	0.50	0.81	0.19	53.0	68.30	45.80	0.58
		23-Aug-1988	0.65	20	0.53	1.30	0.23	62.2	78.50	47.20	0.47
		07-Sep-1990	0.86	20	0.60	1.20	0.17	62.3	79.00	47.50	0.51
		30-Jul-1992	0.21	8	0.70	0.93	0.60	64.1	72.80	55.50	0.69
		27-Sep-1994	0.53	19	0.50	1.68	0.11	58.2	72.40	42.70	0.52
Northern Pike	Lake St. Lawrence	03-Oct-1977	0.71	8	0.45	0.82	0.17	63.7	77.10	36.30	0.39
		01-Jan-1981	0.74	21	0.26	1.30	0.11	50.6	84.50	32.00	0.35
		08-Sep-1983	0.16	20	0.33	0.76	0.15	60.5	77.80	49.00	0.33
		24-Sep-1985	0.83	20	0.39	1.10	0.14	66.1	85.50	47.50	0.27
		10-Sep-1987	0.80	19	0.31	0.65	0.09	63.2	77.60	51.50	0.26

Species	Location	Sample Date	R^2	N	Total Mercury (mg/kg)			Total Length (cm)			Standard value (mg/kg)
					Mean	Max	Min	Mean	Max	Min	
		01-Sep-1989	0.64	20	0.31	1.00	0.13	60.7	86.00	40.50	0.30
		15-Nov-1991	0.72	20	0.42	0.88	0.21	65.1	80.50	50.40	0.34
		01-Nov-1993	0.79	14	0.41	0.93	0.15	65.6	87.10	54.70	0.31
		03-Oct-1995	0.67	20	0.44	1.00	0.16	64.8	78.50	54.60	0.34
Northern Pike	Lake St. Francis	01-Apr-1977	0.76	40	0.69	1.20	0.26	53.5	71.00	37.00	0.82
		01-Aug-1978	0.76	7	0.93	1.50	0.31	51.4	67.00	36.50	1.17
		01-Jan-1981	0.90	21	0.74	1.50	0.25	57.8	77.20	33.00	0.74
		11-Sep-1984	0.75	23	0.65	1.20	0.27	56.4	80.20	31.40	0.70
		27-Sep-1984	0.74	25	0.42	0.84	0.18	52.3	69.50	40.40	0.54
		30-Sep-1986	0.88	17	0.64	1.60	0.28	57.8	80.70	38.00	0.64
		25-Oct-1988	0.15	20	0.59	1.60	0.22	50.4	59.40	41.60	0.69
		15-Dec-1989	0.88	15	0.58	1.40	0.23	48.4	70.00	36.00	0.91
		15-Dec-1989	0.08	23	0.53	1.10	0.16	49.4	67.00	32.50	0.55
		01-May-1990	0.75	20	0.64	2.60	0.32	59.4	92.00	42.00	0.54
		02-Nov-1992	0.83	25	0.60	1.30	0.20	63.3	89.20	47.60	0.49
		07-Oct-1994	0.86	30	0.47	1.20	0.20	57.8	82.50	40.20	0.48
Walleye	Thousand Islands	01-Sep-1989	0.49	13	0.37	1.10	0.05	52.9	71.40	25.50	0.29
		01-Sep-1991	0.79	8	0.34	0.82	0.09	54.4	70.00	32.50	0.16
		01-Nov-1993	0.86	10	0.33	0.87	0.06	49.8	71.00	29.20	0.20
		03-Oct-1995	0.94	5	0.16	0.32	0.08	37.2	50.30	26.20	0.24
Walleye	Middle Corridor	07-Sep-1990		1	0.32	0.32	0.32	50.2	50.20	50.20	0.32
		27-Sep-1994	0.14	6	0.29	0.40	0.21	47.4	55.00	42.00	0.28
Walleye	Lake St. Lawrence	01-Jan-1981	0.89	20	0.19	0.34	0.09	43.5	53.80	31.50	0.20
		08-Sep-1983	0.93	17	0.27	0.72	0.08	41.3	62.20	28.20	0.28
		24-Sep-1985	0.92	20	0.28	0.71	0.11	46.7	63.00	38.50	0.21
		15-Apr-1986	0.82	30	0.47	0.97	0.06	51.8	60.00	41.50	0.31
		10-Sep-1987	0.92	20	0.19	0.45	0.08	46.7	59.60	33.40	0.16
		25-Oct-1988	0.82	11	0.62	1.40	0.14	57.4	74.60	33.40	0.26
		01-Sep-1989	0.74	21	0.28	1.40	0.12	45.5	64.50	32.50	0.19
		15-Nov-1991	0.80	20	0.28	0.92	0.08	46.2	62.00	19.00	0.20
		01-Nov-1993	0.88	29	0.15	0.48	0.07	39.5	61.00	26.80	0.19
		03-Oct-1995	0.88	20	0.21	0.44	0.09	41.2	58.90	26.50	0.23

Species	Location	Sample Date	R^2	N	Total Mercury (mg/kg)			Total Length (cm)			Standard value (mg/kg)
					Mean	Max	Min	Mean	Max	Min	
Walleye	Lake St. Francis	04-Apr-1976	0.91	19	1.32	3.40	0.60	54.7	75.00	44.00	0.69
		01-Apr-1977	0.97	4	1.19	1.60	0.76	52.8	59.00	47.00	0.85
		01-Apr-1977	0.73	13	1.57	2.80	0.69	52.8	60.50	47.00	1.04
		01-Jan-1981	0.80	21	1.04	3.40	0.30	52.4	73.80	31.50	0.52
		01-Jan-1982	-0.11	20	0.74	2.10	0.39	53.2	72.60	37.60	0.74
		27-Sep-1984	0.85	10	0.95	2.10	0.58	56.1	67.20	51.70	0.50
		31-Aug-1985	0.85	20	0.65	1.30	0.24	50.5	70.00	38.50	0.52
		30-Sep-1986	0.94	20	0.73	1.90	0.24	52.2	76.50	38.20	0.47
		05-Sep-1987	-0.15	9	0.61	1.20	0.30	53.4	69.40	45.40	0.61
		25-Oct-1988	-0.09	20	0.80	1.80	0.27	46.1	57.90	38.00	0.80
		15-Dec-1989	0.74	5	0.45	0.55	0.30	42.4	52.50	31.00	0.48
		15-Dec-1989	0.59	12	0.44	0.93	0.24	41.5	55.00	31.00	0.47
		15-Apr-1990	0.18	20	0.92	1.50	0.33	60.8	69.90	42.80	0.78
		01-May-1990	0.87	8	0.66	1.20	0.34	53.0	61.40	48.10	0.41
		19-Dec-1991	0.78	19	0.48	1.10	0.06	47.8	68.40	33.50	0.41
		20-Dec-1991	0.70	20	1.03	1.90	0.33	59.3	74.50	48.00	0.57
		02-Nov-1992	0.19	7	1.01	2.00	0.48	62.4	75.00	52.00	0.80
		07-Oct-1994	0.80	11	0.48	1.10	0.27	47.3	63.50	36.50	0.42
Walleye	Raisin River	01-Jan-1978	0.81	31	0.85	2.10	0.30	48.0	60.10	37.90	0.66
		01-May-1991	0.69	16	0.79	1.60	0.33	55.8	75.20	40.00	0.51
		11-May-1992	0.91	19	0.64	1.50	0.20	53.2	66.50	40.20	0.34
		03-May-1993	0.89	20	0.84	2.00	0.34	59.8	75.70	44.60	0.33
Yellow Perch	Thousand Islands	27-May-1976	0.61	6	0.33	0.49	0.20	20.5	25.00	18.90	0.32
		01-Jan-1983	0.77	12	0.19	0.45	0.10	19.0	24.50	16.20	0.21
		01-Jan-1984	0.43	23	0.40	0.64	0.19	23.6	26.60	21.10	0.28
		15-Jun-1984	0.43	23	0.40	0.64	0.19	23.6	26.60	21.10	0.28
		22-Sep-1986	0.92	11	0.24	0.78	0.07	19.6	25.30	17.30	0.23
		15-Aug-1988	0.61	28	0.17	0.64	0.01	18.5	23.50	15.40	0.21
		22-Aug-1988	0.31	17	0.08	0.29	0.02	18.5	22.60	15.00	0.09
		01-Sep-1989	0.27	20	0.27	0.38	0.15	22.4	27.10	19.30	0.24
		01-Sep-1991	-0.13	14	0.20	0.32	0.09	20.8	23.40	19.00	0.20
		30-Jul-1992	0.82	22	0.18	0.52	0.06	18.0	25.40	15.50	0.24
		01-Nov-1993	0.52	27	0.15	0.37	0.05	20.6	24.80	12.20	0.14
		03-Oct-1995	0.02	20	0.19	0.33	0.08	21.5	26.70	14.60	0.19

Species	Location	Sample Date	R^2	N	Total Mercury (mg/kg)			Total Length (cm)			Standard value (mg/kg)
					Mean	Max	Min	Mean	Max	Min	
Yellow Perch	Middle Corridor	01-Jan-1975	0.77	27	0.40	1.11	0.19	21.4	29.50	17.60	0.31
		01-Jun-1978	0.68	7	0.35	0.73	0.21	18.0	20.00	15.00	0.51
		01-Jan-1983	0.89	17	0.33	0.76	0.12	20.1	27.70	14.20	0.30
		01-Jan-1983	0.84	20	0.30	0.62	0.14	24.2	29.50	21.20	0.15
		20-Jun-1984	0.48	19	0.47	0.71	0.31	25.5	29.30	23.30	0.30
		21-Jul-1984	0.82	30	0.34	0.67	0.12	24.8	30.10	21.80	0.15
		09-Sep-1985	-0.31	10	0.30	0.51	0.17	23.8	27.00	22.40	0.30
		13-Sep-1985	0.64	8	0.30	0.38	0.21	24.8	31.00	22.80	0.23
		24-Sep-1986	0.83	20	0.22	0.45	0.11	20.6	26.10	16.50	0.20
		25-Sep-1986	0.64	20	0.29	0.50	0.12	23.5	28.50	18.60	0.19
		18-Aug-1988	0.50	24	0.12	0.60	0.01	22.2	29.60	16.70	0.05
		23-Aug-1988	0.68	21	0.30	0.70	0.18	21.4	27.50	17.00	0.24
		07-Sep-1990	0.02	20	0.25	0.39	0.10	23.7	28.00	16.00	0.25
		30-Jul-1992	0.71	42	0.27	0.56	0.11	19.4	30.10	15.00	0.28
		27-Sep-1994	0.42	28	0.25	0.77	0.13	21.8	26.60	18.70	0.21
Yellow Perch	Lake St. Lawrence	03-Oct-1977	0.80	25	0.24	0.72	0.10	18.8	25.50	13.60	0.25
		01-Jan-1981	0.48	19	0.30	0.67	0.08	22.8	26.80	16.50	0.23
		08-Sep-1983	0.63	60	0.19	0.42	0.06	19.2	37.20	14.60	0.20
		24-Sep-1985	0.68	20	0.22	0.47	0.10	21.0	27.50	15.20	0.20
		10-Sep-1987	0.81	20	0.15	0.30	0.05	20.6	26.10	15.10	0.13
		25-Oct-1988	0.49	10	0.17	0.24	0.14	23.0	27.40	18.30	0.15
		01-Sep-1989	0.35	20	0.23	0.46	0.14	20.3	28.70	15.30	0.23
		15-Nov-1991	0.52	20	0.18	0.38	0.05	21.0	25.20	16.30	0.16
		01-Nov-1993	0.86	26	0.12	0.38	0.04	20.0	30.00	14.20	0.12
		03-Oct-1995	0.21	20	0.19	0.33	0.09	23.9	27.00	20.30	0.16
Yellow Perch	Lake St. Francis	01-Apr-1977	0.63	32	0.46	1.30	0.17	21.6	29.00	16.00	0.34
		01-Aug-1978	-0.42	4	0.29	0.35	0.22	21.2	22.00	20.00	0.28
		01-Jan-1981	0.82	20	0.30	0.46	0.22	23.0	27.00	21.30	0.20
		01-Jan-1983	0.48	28	0.36	0.62	0.18	25.2	27.50	22.40	0.22
		11-Sep-1984	0.78	32	0.29	0.72	0.11	24.0	29.30	14.60	0.15
		27-Sep-1984	0.75	29	0.29	0.61	0.14	21.8	27.50	14.20	0.24
		30-Sep-1986	0.64	20	0.24	0.49	0.10	24.4	28.30	18.60	0.13
		25-Oct-1988	0.75	20	0.26	0.55	0.12	23.9	29.20	14.70	0.17
		15-Dec-1989	0.60	18	0.32	0.54	0.17	22.9	26.00	20.30	0.23
		15-Dec-1989	0.27	20	0.29	0.43	0.15	20.9	23.00	18.30	0.28
		01-May-1990	0.67	40	0.20	0.54	0.08	22.2	29.00	18.10	0.15
		02-Nov-1992	0.27	31	0.19	0.32	0.09	23.9	31.50	18.20	0.16
		07-Oct-1994	0.03	20	0.15	0.21	0.11	21.3	25.50	18.10	0.15

Table 33A. Temporal variability in mercury concentration within species and regions from Lalonde's (1999) analysis of MOE/MNR Sport Fish Contaminants database.

Region	Species	Significant between-year differences in mercury concentration
Thousand Is.	northern pike	• 1976 higher than 1985 onwards • stable from 1985-1995
	walleye	*long term data set from 1970s onwards not available*
	yellow perch	*long term data set from 1970s onwards not available*
	white sucker	*long term data set from 1970s onwards not available*
Brockville	northern pike	• 1992 higher than 1986 and 1994 • 1970s and 80s not higher than 1990s
	walleye	• no samples collected
	yellow perch	• 1976 higher than 1983 onwards • 1985, 1986 and 1990 higher than 1992 • 1992 higher than 1994
	white sucker	• 1976 higher than 1983 onwards • 1985, 1986 and 1988 higher than 1994
Lake St. Lawrence	northern pike	• stable all years sampled (1981-1995)
	walleye	*long term data set from 1970s onwards not available*
	yellow perch	• decreased each sampling year from 1977-1993
	white sucker	• 1975 and 1977 higher than 1985 onwards • stable from 1985-1993
Lake St. Francis	northern pike	• 1977 higher than 1984 and 1990 • 1981, 1988 and 1989 higher than 1992 and 1994 • apparent decrease over time especially from 1980s to 1990s
	walleye	• 1976 higher than 1982 onwards • 1977 and 1978 higher than 1981 onwards • stable 1989-1994 (i.e., 1989, 1990, 1991, 1992, 1993, 1994)
	yellow perch	• decreased each sampling year from 1977 to 1990 • stable 1990, 1992 and 1994
	white sucker	• 1976, 1977, 1978, 1981 and 1982 higher than later years (1988, 1990 and 1994) • stable during 1988, 1990 and 1994

Table 33B. Years of MOE/MNR Sport Fish Contaminants Program sampling for each species and region reported in Lalonde (1999).
Note: data for additional years may be available in some cases.

Region	Species	Years Sampled		
		1970s	1980s	1990s
Thousand Is.	northern pike	76	84,88,89	91,93,95
	walleye		89	91,93,95
	yellow perch		83,84,88,89	91,93
	white sucker		83,84,85,86,87,88	93
Brockville	northern pike	75,78	83,84,85,86,87,88	90,92,94
	walleye	no samples collected		
	yellow perch	75	83,84,85,86	90,92,94
	white sucker	75	83,84,85,86,87,88	92,94
L. St. Lawrence	northern pike		81,85,87,89	91,93,95
	walleye		81,85,86,87,88,89	91,93,95
	yellow perch	77	81,85,87,88,89	92,93,95
	white sucker	75,77	85,87,89	93
L. St. Francis	northern pike	77	81,84,86,88,89	90,92,94
	walleye	76,77,78	81,82,84,85,86,88,89	90,91,92,93,94
	yellow perch	77	81,83,84,86,88,89	90,92,94
	white sucker	76,77,78	81,82,88	90,94

3.2 Spottail Shiners

The spottail shiner (*Notropis hudsonius*) is a forage fish that accumulates substances such as PCBs and mercury by ingestion of contaminated sediment, uptake from water passing over the gills and feeding on contaminated organisms in the food chain. Young-of-the-year spottails have a very limited home range, foraging in a stretch of the near shore zone with a range <1 km. It can therefore be inferred that contaminants found in juvenile spottail tissues were present in the aquatic ecosystem during the year of fish collection and within the confines of their home range. Spottail shiners are therefore useful bioindicators of localized sources of bioaccumulative contaminants and have been used extensively as such throughout the Great Lakes basin.

MOE has collected young-of-the-year spottails from over 26 stations in the north and south channel of the St. Lawrence River at Cornwall and Massena since 1979 (Figure 96). All stations were not sampled every year, but for at least six stations there are over five years of data and at least 11 stations have data from three or more years.

The total PCBs data for spottails are more useful than the mercury data in the Cornwall AOC. The correlation between tissue and sediment concentrations is much stronger for PCBs than it is for mercury. This may be because PCB sediment concentrations in south shore sediments were much higher than mercury concentrations in north shore sediment, making it easier to detect bioaccumulation of PCBs in spottails. Moreover, PCBs may accumulate more readily than mercury in young-of-the-year spottail shiners. PCB results are discussed in further detail in Dreier et al. (1997).

Mercury concentrations in spottails collected from the upstream reference site in Lake St. Lawrence (Macdonnell Island) ranged from a low of 11±8 ng/g wet wt in 1989 to a high of 63±8 ng/g wet wt in 1991 as shown in Table 34 and Figure 96A. In all years of sampling, mercury concentrations were similar in fish collected from the south channel of the river at Massena, NY and fish from the upstream reference site.

As shown in Figure 96A, spottails collected from the north channel of the river at Cornwall generally also had mercury concentrations similar to those at the upstream reference site except in 1988, 1989 and 1990. In these three years, total mercury concentrations in spottails collected at the Cornwall marina (Cornwall Civic Complex) were significantly higher than in fish from both upstream (reference site) and downstream (at Pilon Island) of Cornwall. Concentrations downstream at the Pilon Island site were similar to upstream reference site values (Dreier et al. 1997).

Except for the 1988-1990 data, these results do not reflect the presence of local discharges of mercury along the Cornwall waterfront known to have occurred from 1979 to the 1990s until the closure of Courtaulds in 1992 and the ICI chlor-alkali plant in 1995. Possible reasons for the unclear mercury data for spottails collected along the Cornwall waterfront are listed below:

- Spottails may take up mercury more from the water column (transferred into blood stream from water passing over gills) than from the sediment (ingestion of benthic organisms and contaminated sediment). If so, mercury contamination would only be seen in fish collected close enough to mercury point sources (effluent) to be affected via the water column. Spottail sample sites were limited to places where fish could be found. The Cornwall marina site was about 1.5 km downstream of Domtar/ICI discharge, so the home range of spottails collected there probably didn't include the point of discharge. Likewise, the station closest to Courtaulds was about 1.5 km downstream of the Courtaulds discharges.

- There could be a delay between discharge of inorganic mercury and the appearance of mercury in spottails, as inorganic mercury is converted to methyl mercury which is then taken up through the food chain.

- Spottails are forage fish that feed on plankton and benthic invertebrates. There may not be enough biomagnification at this point in the food chain to see a clear effect of mercury accumulation as can be seen in sport fish that are at the top of the food chain.

The Cornwall area spottail shiners data are discussed further in Dreier et al. (1997) and Crabbe et al. (1997). Mercury data from 1987-1997 are summarized in Table 34 below.

Table 34. Total mercury concentration (ng/g wet wt) in spottail shiners from the St. Lawrence River near Cornwall, Ontario, 1987-1997. Source: L. Richman, MOE, Environmental Monitoring & Reporting Branch data files. sd=standard deviation.

Year Sampled	MacDonnell Island		Cornwall Marina		Pilon Island		Thompson Island		Point Mouillee	
	mean	sd	mean	sd	mean	sd	mean	sd	mean	sd
1987	38	4	46	5	40	0	40	0	37	12
1988	26	15	101	27	47	17				
1989	11	8	61	31	25	11				
1990	42	9	64	9	29	7	40	0		
1991	63	8	77	10	60	6				
1993	48	5	54	5						
1996					26	5	28	4	28	4
1997	16	5	36	11	28	8	22	5		

3.3 Biological Sediment Assessments

The effects of sediment contamination on aquatic organisms must be considered when determining the need for remedial actions. Examples of biological effects include:

- toxicity to benthic (sediment-dwelling) invertebrates
- alterations in the types and numbers (community structure) of benthic invertebrates
- availability of sediment contaminants to aquatic organisms.

Unfortunately, the effects of mercury contamination on aquatic organisms are not readily measured. Although some contaminants have demonstrable short-term effects on aquatic organisms, bioaccumulative substances such as mercury act over a longer time period. As a result, standard bioassessment techniques such as short term contaminant uptake studies and acute toxicity tests are not likely to show significant effects due to mercury. Neither is mercury contamination expected to cause significant alterations in the structure of communities of organisms low on the food chain such as benthic invertebrates.

The biological impact of mercury contamination occurs through the process of biomagnification. Mercury tends to be retained once ingested and is therefore passed by ingestion up through the food chain, resulting in higher and higher concentrations at each trophic (feeding) level. Top predators such as sport fish can accumulate mercury to concentrations that exceed human consumption guidelines.

The sediment bioassessment results discussed in the following sections should be interpreted in this light and in combination with the sport fish mercury data (Section 3.1). Standard laboratory bioassessments of Cornwall sediment samples are discussed below, but since mercury is one of the key contaminants of concern other types of studies would be required to fully assess the biological effects of contamination in these sediments.

Sediment collected from the Cornwall waterfront in October 1997 was assessed for acute toxicity, short term bioaccumulation potential and benthic community structure by MOE (Bedard 1999) and Environment Canada (Reynoldson, in prep.). These and earlier biological studies of Cornwall sediment (Richman 1994; Jaagumagi 1987) are reviewed below.

Please note that there were differences in sediment sampling and analysis for Bedard (1999) and Reynoldson (in prep.) accounting for slight differences sediment chemistry reported by the two authors. Two sediment collections were made on the same date from opposite sides of the boat. Bedard (1999) sediment was collected by Ponar dredge. Chemical analysis of these samples was done by MOE on sieved sediment. Reynoldson (in prep.) sediment samples were collected by mini-box corer and analyzed unsieved by Environment Canada (Sepratech) (T. Reynoldson, Environment Canada and L. Richman, MOE, pers. comm).

3.3.1 Toxicity and Bioaccumulation, 1997 (Bedard 1999)

Bedard (1999) assessed the spatial pattern of sediment toxicity and chemical bioaccumulation using static, laboratory lethal and sub-lethal toxicity tests and a short-term bioaccumulation assay. The testing was done on surficial sediment collected by Ponar dredge in October 1997 from seven sites along the Cornwall waterfront between the Lamoureux Park boat launch and Pilon Island (Richman 1999) as shown in Figure 97.

Three independent tests were performed on bulk sediment samples: (1) mortality, growth and avoidance behaviour of the burrowing mayfly, *Hexagenia limbata*, were measured in a

21-day test; (2) mortality and growth of the chironomid, *Chironomus tentans*, were measured in a 10-day test; and (3) mortality and inorganic chemical uptake by the juvenile fathead minnow, *Pimephales promelas*, were examined using a standard 21-day test. Mayflies and chironomids are sediment dwelling (benthic) invertebrates and the fathead minnow is a bottom forager.

Sediments were analyzed for physical parameters, nutrients, trace metals (including total mercury), chlorinated organic compounds and pesticides, PCBs, polycyclic aromatic hydrocarbons (PAHs) and total petroleum hydrocarbons (TPH). Sediments used for the uptake assay were stored frozen for 3 months and reanalyzed at the time the assay was done.

Bioassay results for the test sediments were compared with results for sediment from two locations:

(a) a 'reference' site (station CS-175) near the south shore of the north channel, known to be less influenced by local contaminant sources than sites near the north shore; and

(b) a 'negative control' site at Honey Harbour, Georgian Bay that provided a sample of relatively uncontaminated sediment.

3.3.1.1 Toxicity (survival and growth) (Bedard 1999)

The bioassay test results are reported in Table 35, reproduced from Bedard (1999). Few statistically significant toxicological effects were found. Organism survival (mortality) of all three test species was not affected by sediment from any of the test sites. Growth was significantly reduced at two sites for one test animal, the mayfly (*Hexagenia limbata*). Mayfly nymphs were 47% and 28% smaller by weight than the reference animals on exposure to sediment from stations CS-131 and CS-128 respectively (Bedard 1999).

Apart from a weak positive correlation between mayfly growth and sediment TOC, no significant correlations were found between the biological (organism survival and growth) and sediment quality data (physical and chemical parameters). This may have been due to the narrow range in biological responses in conjunction with a relatively small sample size (Bedard 1999).

Chemical and physical data for test and reference sediments are provided in Tables 36, 37 and 38, reproduced from Bedard (1999). Total mercury concentration in sediment exceeded the severe effect level of 2.0 µg/g at three of the seven test stations: CS-109 (3.6 µg/g), CS-131 (5.4 µg/g) and CS-128 (6.1 µg/g). Sediment mercury concentration was not correlated with a significant biological response. Most of the remaining trace metals were present at concentrations near the lowest effect level (Bedard 1999).

It should be noted, however, that no biological testing was done on the most mercury contaminated sediment collected in 1997. Richman (1999) recorded mercury concentrations of 19.5 and 14.7 µg/g in the top 10 and 3 cm respectively of sediment collected at station 131 in 1997. Concentrations were not as high in any of the sediment collected for bioassessment (Bedard 1999).

Bedard (1999) reported that "most of the test sediments had a characteristic petroleum-like odour, oily sheen and viscous blobs that stained laboratory equipment". The nature of this material was unclear, however, since it did not occur as high concentrations of PAHs, TPH or PCBs. Total PAH concentrations were near the lowest effect level of 4.0 µg/g. Total petroleum hydrocarbons (which include PAHs) were below the detection limit of 100 µg/g. Total PCBs were above the lowest effect level of 70 ng/g at all test sites but at least 70 times lower than the severe effect level (corrected for organic carbon content).

In the laboratory test chambers, average un-ionized ammonia concentrations in the water overlying the sediments exceeded the Provincial Water Quality Objective of 0.02 mg/L for most test sites. Un-ionized ammonia concentrations were higher for test sites than for reference and control sites. Mean un-ionized ammonia concentrations ranged from 0.03 mg/L for the mayfly test to 0.06 mg/L for the minnow test, with an intermediate concentration of 0.05 mg/L for the midge test. These are well below literature reports of acute and chronic un-ionized ammonia concentrations for fathead minnows and midge larvae (Bedard 1999). Literature values for mayflies were unavailable, but even at concentrations ten times higher than in this study mayfly growth is unlikely to be compromised (D. Bedard, pers. comm.).

Toxicity tests were conducted in 1991 at several similar sites in the St. Lawrence River (Bedard and Petro 1992; Richman 1994) as discussed in Section 3.3.5. In these earlier tests, mayfly and midge growth was reduced by 79% and 89% respectively on exposure to sediment collected near station CS-105 (Figure 97). The 1991 sediment had a total mercury concentration of over 7.5 µg/g and concentrations of copper, lead and zinc were also above the SEL (Richman 1994).

In contrast, the 1997 sediment from CS-105 contained 1.5 µg/g mercury and was not associated with a reduction in mayfly or midge growth. Significant reductions in mayfly growth were observed on exposure to sediment from two other 1997 stations: CS-131 (5.4 µg/g mercury; 47% weight reduction) and CS-128 (6.1 µg/g mercury; 27% weight reduction) (Bedard 1999). The greater biological effects observed in the 1991 study may have been due to higher concentrations of multiple metals, including mercury, and the possible presence of other organic contaminants (Bedard 1999).

[illegible] the [illegible] suggests [illegible] growth effects at CS-131 + CS-128 [illegible] [illegible]

Table 35. Summary of biological results of mayfly, midge and minnow sediment bioassays for control and St. Lawrence River 1997 sediments. Source: Bedard (1998).
Mean (std deviation) shown. n=4 for mayfly and midge tests; n=3 for minnow tests.

Test Organism	*Hexagenia limbata* (Mayfly)		*Chironomus tentans* (Midge)		*Pimephales promelas* (Fathead Minnow)
Station	*% Mortality*	*Ave. Individual Body Weight* (mg wet wt.)	*% Mortality*	*Ave. Individual Body Weight* (mg wet wt.)	*% Mortality*
Honey Harbour Control	A 0 (0)	B 6.21 (0.6)	AB 6.6 (5)	BC 8.49 (0.8)	A 0 (0)
Stn CS - 175 Reference	A 0 (0)	A 11.05 (1.1)	A 0 (0)	A 10.53 (1.3)	A 0 (0)
St. Lawrence R. Stn CS - 105	A 0 (0)	A 10.66 (1.1)	AB 1.6 (3)	BC 7.69 (0.7)	A 0 (0)
Stn CS - 109	A 2.5 (5)	A 11.39 (1.0)	AB 8.3 (10)	BC 7.72 (0.9)	A 0 (0)
Stn CS - 128	A 5.0 (10)	B 8.05 (0.8)	AB 6.6 (5)	ABC 8.92 (0.6)	A 0 (0)
Stn CS - 131	A 10.0 (14)	B 5.82 (0.2)	AB 1.6 (3)	C 7.18 (1.3)	A 0 (0)
Stn CS - 156	A 0 (0)	A 10.95 (0.4)	B * 26.6 (28)	AB 9.34 (2.4)	A 0 (0)
Stn CS - 167	A 2.5 (5)	A 13.17 (2.9)	AB 1.6 (3)	BC 8.59 (0.7)	A 0 (0)
% MSD % C.V. D.P.	11.6 171.4 2.0	- 10.5 6.8	19.5 112.1 3.5	- 13.0 2.9	- - -

Zone 4

Zone 2

Zn 1

sig. diff. from ref but not control

→ toxicity to Chironomus tentans; mayfly would see toxicity to ... - need to review

* %Mortality value is significantly different than the reference sediment (Dunnett's 1-tailed t-test;p<0.05).

Note:

Means that share a common letter within a column are not significantly different; Tukey's HSD test for % Mortality (p<0.05) and planned comparisons using LSMEANS for comparing Body Weight (p<0.01).

MSD - Minimum Significant Difference; C.V. - Coefficient of Variation; D.P. - Discriminatory Power.

→ ... minnows allowed 25% mortality in controls (validity ...)

Table 36. Sediment physical and nutrient characteristics in control and St. Lawrence River 1997 sediment used in sediment bioassays. Source: (Bedard 1998).

Station	*% Sand* (2mm-62um)	*% Silt* (62-3.7um)	*% Clay* (3.7-0.1um)	*% LOI*	*TOC* mg/g	*TP* mg/g	*TKN* mg/g
Honey Harbour Control	33.0	44.0	22.3	8.6	40	1.0	3.1
Station CS - 175 Reference	41.0	36.9	22.2	7.3	36	0.5	1.1
St. Lawrence River							
Station CS - 105	53.0	31.1	16.1	6.3	30	0.9	2.5
Station CS - 109	12.0	59.5	29.4	12.0	41	1.1	3.5
Station CS - 128	50.0	35.4	14.2	14.0	24	0.8	0.7
Station CS - 131	31.0	44.8	23.8	7.3	36	1.0	1.9
Station CS - 156	51.0	33.2	15.1	5.4	24	0.9	2.4
Station CS - 167	45.0	40.2	14.7	7.7	42	0.9	2.8
PSQG SEL Conc (mg/g dry weight)					100	2.0	4.8

Table 37. Bulk concentrations of trace metals in control and St. Lawrence River 1997 sediment (µg/g dry weight) used in sediment bioassays (benthic invertebrates). Source: Bedard (1998).

Station	*Al %*	*As*	*Cd*	*Cr*	*Cu*	*Fe %*	*Hg*	*Mn*	*Ni*	*Pb*	*Zn*
Honey Harbour Control	2.2	5.0	1.4	44	24	3.7	0.09	980	35	54	140
Station CS - 175 Reference	1.6	3.3	1.5	40	41	2.3	0.1	400	28	30	140
St. Lawrence River Station CS - 105	1.2	4.0	1.5	41	55	1.8	1.5	270	25	70	750
Station CS - 109	1.3	4.8	1.7	55	85	2.0	***3.6***	290	32	170	***1200***
Station CS - 128	0.6	2.6	0.7 <T	18	26	1.2	***6.1***	260	13	21	100
Station CS - 131	1.0	4.3	1.3	34	51	1.6	***5.4***	280	23	49	280
Station CS - 156	0.8	3.0	0.8 <T	26	28	1.4	0.7	250	19	31	180
Station CS - 167	1.0	3.6	1.5	33	40	1.6	1.0	220	25	39	120
PSQG SEL Conc.	NA	33	10	110	110	4.0	2.0	1100	75	250	820
PSQG LEL Conc.	NA	6.0	0.6	26	16	2.0	0.2	460	16	31	120

<T - Trace Amount; NA - Not Available.

Bold italics indicates sediment trace metal concentrations that exceed the severe effect level (SEL).

Underlining indicates sediment trace metal concentrations that exceed the lowest effect level (LEL).

Table 38. Bulk concentrations (dry wt) of polycyclic aromatic hydrocarbons (ng/g), total polychlorinated bi (ng/g) and total petroleum hydrocarbons (µg/g) in St. Lawrence River 1997 sediment.

Parameter	Reference CS - 175	Cornwall CS - 105	Cornwall CS - 109	Cornwall CS - 128	Cornwall CS - 131	Cornwall CS - 156	Cornwall CS - 167
Acenaphthene	20 <W	120 <T	76 <T	38 <T	44 <T	68 <T	42 <T
Acenaphythylene	22 <T	110 <T	38 <T	20 <W	23 <T	44 <T	31 <T
Anthracene	20 <W	260	270	61 <T	96 <T	170 <T	140 <T
Benzo[a]anthracene	160 <T	610	940	160 <T	370	720	490
Benzo[b]fluoranthene	180 <T	1000	1600	180 <T	450	740	700
Benzo[k]fluoranthene	45 <T	310	440	70 <T	160 <T	150 <T	180 <T
Benzo[ghi]perylene	91 <T	580	780	86 <T	210	390	360
Benzo[a]pyrene	140 <T	710	1100	140 <T	360	610	530
Chrysene	100 <T	750	1000	140 <T	480	830	480
Dibenzo[ah]anthracene	40 <W	170 <T	150 <T	40 <W	60 <T	110 <T	74 <T
Fluoranthene	140 <T	1200	2100	320	630	850	820
Fluorene	20 <W	160 <T	100 <T	42 <T	60 <T	89 <T	62 <T
Indeno[123-cd]pyrene	120 <T	710	1000	100 <T	250 <T	410	420
Naphthalene	20 <W	120 <T	55 <T	20 <W	25 <T	36 <T	37 <T
Phenanthrene	56 <T	620	970	280	450	480	430
Pyrene	150 <T	1000	1600	340	690	990	730
Total PAHs	1204 <T	8430	12219	1957 <T	4358	6687	5526
Total PCBs	60	200	360	100	200	160	160
TPHs	100 <W	100 <W	100 <W	100 <W	100 <W	100 <W	100 <W

<W - Not Detected; <T - Trace Amount; Underlining indicate sediment PAH and PCB concentrations that exceed PSQG-LELs; PSQG's not available for acenapthene, acenaphthylene, benzo[b]flouranthene and naphthalene.

3.3.1.2 Metal Uptake in Juvenile Fathead Minnows (Bedard 1999)

In an aquatic ecosystem, fish can acquire sediment contaminants in a number of ways depending on their feeding habits. Contaminants can be taken in directly from sediment particles in the water column or while bottom feeding; from ingesting benthic invertebrates; from ingesting planktonic (free-floating) algae and invertebrates; or from ingesting prey fish.

Bedard (1999) measured metal uptake in juvenile fathead minnows (*Pimephales promelas*) exposed to sieved test sediments for 21 days under laboratory conditions. Whole-body tissue concentrations of cadmium, copper, mercury, manganese, nickel, lead and zinc were analyzed in the exposed fish. Metal concentrations in fish and sediment were compared to assess the availability of sediment metals to minnows by means of direct contact with the sediment and uptake from overlying water. Mean metal concentrations in assayed minnows and sediment samples are shown in Tables 39 and 40 (Bedard 1999).

Metal availability was measured as the biota-sediment accumulation factor (BSAF), calculated for each metal as the ratio of metal concentration in minnows vs. metal concentration in sediment to which they are exposed. The BSAF is greater than 1.0 when metal concentration in exposed minnows exceeds the concentration in test sediment. A BSAF>1.0 indicates uptake of the metal from the sediment into the fish (bioaccumulation).

The fathead minnow short-term bioaccumulation test showed a low availability of total mercury and other metals under controlled laboratory conditions using static exposures. Tissue mercury concentration was highest (0.17 µg/g wet weight) in minnows exposed to sediment from station CS-128 (station 128 on Figure 97). However, this value was only double the concentration in fish exposed to sediment from the other test sites and the reference site. It was also three times lower than the IJC Aquatic Life Guideline and the Ontario sportfish consumption guideline (both set at 0.50 µg/g). CS-128 had a total sediment mercury concentration of 5.1 µg/g and was one of three stations at which the SEL was exceeded for mercury. However, the BSAF at CS-128 was only 0.21 indicating that mercury availability was low (Bedard 1999).

BSAFs greater than 1.0 were reported for mercury and zinc at some of the less contaminated sites. But rather than indicating high metal availability, these results reflected the unexplainably high pre-exposure mercury concentrations in the lab cultured minnows used for the test, which ranged from 0.41-0.59 µg/g dry wt (Bedard 1999). With a high initial metal concentration in the minnows, the lower the sediment metal concentration the higher the apparent BSAF ratio.

Mercury availability and uptake was marginal for the Cornwall sediments and may be accounted for by the process of sediment ingestion. There was a slight correlation between total mercury concentrations measured in the sediment and those measured in the fish. The direct transfer of mercury contaminated sediment into bottom-feeding fish could be ecologically relevant given the capacity of mercury to biomagnify in the aquatic food chain (Bedard 1999).

This type of short-term bioaccumulation assay rarely shows significant uptake of metals and is more effective at measuring uptake of organic compounds such as PCBs (D. Bedard, pers. comm.). Some other types of studies (possibly *in situ* uptake) would therefore be required in order to fully evaluate bioaccumulation of mercury or other metals from Cornwall sediments, especially in those areas where high mercury concentrations were encountered.

Table 39. Mean metal concentrations (μg/g wet weight) in fathead minnows exposed to control and St. Lawrence River 1997 sediments in the laboratory and associated biota-sediment accumulatio Standard deviation in parentheses. n=3. Source: Bedard (1998).

Station	Cd	Cu	Hg	Mn	Ni	Pb	Zn	
Pre-exposure	0.04 (.01)	0.89 (.03)	0.09 (.01)	1.7 (.07)	0.02 <W (.00)	0.02 <W (.00)	39.5 (0.7)	
Georgian Bay Control	A 0.10 (.04)	A 0.96 (.11)	A 0.08 (.01)	C 7.3 (.94)	AB 0.17 (.13)	AB 0.15 (.18)	AB 43.3 (2.5)	
Reference Station CS-175	AB 0.03 (.03)	BC 1.30 (.10)	A 0.06 (.01)	B 4.8 (.30)	BC 0.47 (.08)	AB 0.15 (.11)	A 42.3 (1.5)	4
BSAF	0.14	0.21	4.30	0.08	0.10	0.03	2.14	
Station CS-105	AB 0.06 (.02)	BC 1.20 (.05)	A 0.06 (.01)	A 2.7 (.20)	A 0.12 (.13)	B 0.55 (.24)	AB 46.0 (3.0)	
BSAF	0.29	0.15	0.27	0.06	0.03	0.05	0.40	
Station CS-109	B 0.01 (.01)	D 1.66 (.05)	A 0.08 (.02)	A 3.3 (.10)	ABC 0.36 (.10)	C 1.40 (.20)	B 50.0 (2.6)	
BSAF	0.03	0.12	0.13	0.07	0.08	0.05	0.27	
Station CS-128	AB 0.04 (.03)	A 0.97 (.03)	B 0.17 (.03)	A 2.8 (.11)	AB 0.12 (.11)	A 0.07 (.08)	A 41.0 (1.7)	2
BSAF	0.36	0.26	0.21	0.07	0.06	0.02	2.45	
Station CS-131	AB 0.02 (.03)	BC 1.20 (.11)	AB 0.12 (.04)	A 2.8 (.30)	A 0.06 (.05)	AB 0.23 (.22)	A 42.3 (2.0)	
BSAF	0.12	0.15	0.09	0.06	0.02	0.03	0.99	
Station CS-156	AB 0.05 (.02)	AB 1.13 (.11)	A 0.08 (.01)	A 3.1 (.49)	AB 0.14 (.13)	AB 0.42 (.15)	AB 46.3 (1.5)	
BSAF	0.31	0.26	1.30	0.08	0.05	0.08	1.69	
Station CS-167	AB 0.09 (.03)	CD 1.46 (.05)	A 0.08 (.00)	A 3.0 (.28)	C 0.55 (.17)	AB 0.51 (.17)	AB 44.6 (3.0)	1
BSAF	0.39	0.24	0.43	0.09	0.15	0.08	2.45	

Note: Means that share a common letter within a column are not significantly different using Tukey's HSD test (p<0.05).

Table 40. Bulk concentrations of trace metals and TOC in reference and St. Lawrence River sediment stored for 3 months then used for fathead minnow metal uptake assay and biota-sediment accumulation factor (BSAF) calculations (see Table 39). Source: Bedard (1998).

Station	TOC mg/g	*Cd* µg/g	*Cu* µg/g	*Hg* µg/g	*Mn* µg/g	*Ni* µg/g	*Pb* µg/g	*Zn* µg/g
Station CS - 175 Reference	37	1.5	39	0.1	370	28	32	130
St. Lawrence River								
Station CS - 105	34	1.4	55	1.5	270	25	71	750
Station CS - 109	37	2.0	88	***3.8***	280	29	180	***1200***
Station CS - 128	27	0.8	24	***5.1***	260	13	23	110
Station CS - 131	36	1.4	53	***8.6***	280	23	54	280
Station CS - 156	29	1.1	28	0.4	250	18	32	180
Station CS - 167	41	1.5	40	1.2	210	24	39	120
PSQG SEL Conc.	100	10	110	2.0	1100	75	250	820
PSQG LEL Conc.	10	0.6	16	0.2	460	16	31	120

<T - Trace Amount; NA - Not Available.

Bold italics indicates sediment trace metal concentrations that exceed the severe effect level (SEL).

Underlining indicates sediment trace metal concentrations that exceed the lowest effect level (LEL).

3.3.2 Benthic Community Structure and Toxicity, 1997 (Reynoldson, in prep.)

Reynoldson (in prep.) investigated toxicity and benthic community structure of sediments collected in October 1997 from 12 sites in the north channel of the St. Lawrence River at Cornwall (Figure 98). The sediments were analyzed for 28 physical and chemical parameters.

Test results were assessed using Reynoldson and Day's (1998) *Biological Guidelines for the Assessment of Sediment Quality in the Laurentian Great Lakes*. In a nutshell, this method works as follows:

1. Reynoldson and Day (1998) have identified six invertebrate communities in the Great Lakes at 252 sites from Lake Ontario to Lake Superior. These six communities represent different reference assemblages of organisms expected to occur in uncontaminated soft sediment in the Great Lakes.
2. Computer modelling can predict any new test site to one of these six groups on the basis of eleven habitat descriptor variables. The group to which the site is predicted represents the reference condition community for that site.
3. Specialized software compares the values of biological variables (i.e., toxicity test results and benthic community structure) observed for sediment at the test site with an existing database of values for the assigned group of Great Lakes reference sites.
4. The degree of difference between the test site and its matched reference condition with respect to benthic community structure or toxicity test results indicates severity of environmental stress on benthic invertebrates at the test site.
5. Test site data are analyzed for correlations between environmental and biological variables. This identifies stressors that may be responsible for observed differences in benthic community structure or toxicity relative to the reference condition.

There is some concern about the value of applying this method to sites outside the geographic area of the Great Lakes reference sites. For the Cornwall test sites, the maximum and minimum values of the predictor variables were all within the range measured at the reference sites, with the exception of longitude. It was therefore assumed that habitat conditions were not significantly different from those encompassed by the Great Lakes reference sites. However, the reference sites are lake sites whereas the Cornwall sites are all located within a large river.

The difference in longitude of reference and test sites could mean that the Cornwall sites are beyond the geographic range of some species. However, since the St. Lawrence River is directly connected to the Great Lakes the same organisms could at least potentially be present so it was deemed reasonable to try applying the Great Lakes reference sites method to the Cornwall area (Reynoldson, in prep.). Of the eight most common taxa in the comparable group of reference sites, all but one were present in the Cornwall sediment (Table 32). The one absent species, a tubeworm, may have only appeared to be absent since there were worms present that were too immature to identify to genus (Reynoldson, in prep.)

Chemical and physical characterization of sediment used in the following bioassessments was done on sediment samples taken from the surface (top 2 cm) of a mini-box corer. Sediments used in Reynoldson's (in prep.) toxicity tests had been stored for less than six months at 4° C in the dark. Reynoldson (in prep.) reported that storage for this period does not affect toxicity data. The sediment chemical and physical data provided in Table 41 are for fresh sediment analyzed prior to the six month storage period.

Table 41. Summary of sediment variables at 12 Cornwall sites, 1997. Mercury values are for top 10 cm sample; all other parameters are for top 3 cm sample. **Boldface** indicates parameters at or above the Provincial Sediment Quality Guidelines Severe Effect Level. Sample station numbers shown across first row of table. Source: Reynoldson (in prep.).

Sediment variable	Average +2SD	LEL	SEL	105	109	117	127	128	131	132	156	164	167	175	179
% Gravel	2.54			0	0	0	3.81	0	0	0	0	0	0	0	0
% Sand	108.84			14.57	5.57	50.27	70.28	45.9	14.03	22.71	46.64	16.77	26.98	26.58	17.32
% Silt	83.67			64.48	68.23	31.99	15.55	33.64	52.8	49.76	29.42	56.7	45.22	48.05	53.98
% Clay	72.72			20.95	23.2	17.74	10.37	20.46	33.17	27.53	23.94	26.53	27.8	25.37	28.7
Mean Particle Size	237.38			17.33	15.77	41.49	195.67	31.68	8.12	13.01	28.75	13.97	16.81	15.66	13.07
25% Particle Size	388.21			35.57	35.43	148.37	658.23	108.49	21.77	37.14	102.19	37.12	65.44	66.11	39.47
75% Particle Size	158.44			5.37	4.53	7.5	30.24	5.75	2.26	3.25	4.37	3.46	3.02	3.8	2.97
Sio2 (%)	85.79			53.92	54.51	59.25	61.08	55.78	50.96	54.75	59.47	54.41	58.32	53.12	54.88
Tio2 (%)	0.92			0.67	0.65	0.59	0.56	0.76	0.56	0.63	0.56	0.59	0.64	0.66	0.65
Al2o3 (%)	15.75			10.65	10.85	9.52	8.42	9.78	9.68	9.64	10.31	10.68	11.02	11.71	11.65
Fe2O3 (%)	8.23			4.24	4.05	3.47	3.45	4.17	3.38	3.46	3.29	3.73	3.51	4.65	4.51
MnO (%)	0.40			0.06	0.06	0.06	0.07	0.08	0.06	0.06	0.06	0.06	0.05	0.08	0.08
MgO (%)	5.81			3.06	3.11	2.78	3.14	2.96	2.59	2.61	2.66	2.79	2.74	3.01	3.04
CaO (%)	17.84			6.71	6.62	6.58	7.73	7.29	10.29	8.83	5.72	7.32	4.63	5.59	6.08
Na2O (%)	2.90			1.93	2.03	2.11	1.87	2.12	1.79	1.88	2.46	2.17	2.26	1.91	2.02
K2O (%)	3.45			2.55	2.59	2.32	2.34	2.31	2.23	2.42	2.64	2.64	2.57	2.82	2.82
P2O5 (%)	0.35			0.19	0.2	0.18	0.17	0.19	0.23	0.21	0.19	0.21	0.2	0.21	0.2
Total N (ppm)	6501			3850	3650	2000	1500	2140	2910	2530	1890	2790	2820	4070	3570
Total P (ppm)	1456	600	2000	952	1170	913	910	924	1110	1020	844	942	980	980	1090
Loss On Ignition (%)	26.21			14.63	14.48	12.14	10.88	13.91	18.62	15.84	11.02	14.17	13.15	14.31	13.82
Total Organic C (%)	6.59	1	10	3.71	3.97	3.22	1.67	3.69	4.22	3.46	2.52	3.48	4.32	3.42	3.08
V (ppm)	91			29	28	24	18	26	26	26	19	25	24	34	33
Cr (ppm)	95	26	110	37	40	26	14	20	29	27	19	36	25	35	33
Co (ppm)	24			7	7	6	6	8	7	7	5	9	6	10	8
Ni (ppm)	119	16	75	22	23	15	9	13	20	18	13	19	20	27	23
Cu (ppm)	56	16	110	56	69	55	21	27	56	45	22	61	35	40	36
Zn (ppm)	276	120	820	676	**823**	554	100	121	363	238	140	580	111	130	124
As (ppm)	40	6	33	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5
Cd (ppm)	2.3	0.6	10	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
Pb (ppm)	97	31	250	69	179	85	22	28	50	47	25	131	31	28	26
Hg (ppm)	n.a.	0.2	2	1.67	**4.83**	**2.0**	**4.32**	**11.2**	**19.5**	**6.78**	0.8	**3.06**	1.19	0.08	0.14

3.3.2.1 Benthic Community Structure, 1997 (Reynoldson, in prep.)

Reynoldson (in prep.) examined numbers and types of invertebrates present in sediment samples to determine whether sediment contaminants were affecting benthic community structure at any of the 12 Cornwall test sites shown in Figure 98.

As explained above, the type of benthic community that should be present at each test site was predicted on the basis of eleven habitat attributes to establish the reference or unstressed condition for the site (Reynoldson and Day 1998). All 12 Cornwall sites were predicted to reference Group 2, with the probability of group membership ranging from 51.5% at site 175 to 82.0% at site 127 (Reynoldson, in prep.).

The actual or observed assemblage of benthic organisms at each site was compared to the predicted assemblage to determine whether and to what degree the site differed from its reference condition, with the degree of difference indicating the severity of stress on the benthic community at the test site. The data were analyzed to identify any relationships between possible stressors such as sediment contaminant concentrations and the shift away from the reference condition.

Community structure was assessed at the genus level because it was not possible to identify all organisms to species. Reynoldson and Day (1998) found a total of 120 genera at the 39 Great Lakes Group 2 reference sites. Nine were common (i.e., present at over 50% of Great Lakes Group 2 sites) and are shown in Table 42 along with their occurrence at the Cornwall test sites (Reynoldson, in prep.). Of the nine common Group 2 genera only one, the tube worm *Potamothrix vejdovskyi*, was absent from all twelve Cornwall test sites. However, as previously mentioned, it may have only appeared to be absent since worms were present that were too immature to identify to genus (Reynoldson, in prep.).

Table 42. Taxa characterizing Great Lakes reference Group 2 and their occurrence at 12 Cornwall test sites in 1997. From: Reynoldson (in prep.).

Taxon	Occurrence at Great Lakes Group 2 reference sites (% of the 39 Group 2 sites at which the taxon was present)	Occurrence at 12 Cornwall test sites (number of Cornwall sites at which the taxon was present)
Procladius spp.	84.6	11
Tanytarsus spp.	82.1	9
Pisidium casertanum	79.5	6
Spirosperma ferox	56.4	5
Chironomus spp.	53.8	12 (all)
Cladotanytarsus spp.	51.3	6
Cryptochironomus spp.	51.3	11
Potamothrix vejdovskyi	51.3	0 (none)

The taxonomic data were analyzed using specialized software called "BEAST" (BEnthic Assessment of SedimenT). All information was reduced to three new descriptor variables represented by three ordination axes and plotted on an xy scatterplot. The Cornwall test sites were compared to the Group 2 reference sites in terms of their position with respect to ellipses on the scatterplot. In this way test sites can be assigned to one of four possible quality bands: Band 1 "equivalent to reference", Band 2 "possibly different", Band 3 "different" and Band 4 "very different" from the reference condition. A general example of this type of comparison is shown in Figure 99 from Reynoldson and Day (1998).

Four of the Cornwall sites (127, 164, 175, 179) were in Band 2 "possibly different" from the Group 2 reference sites (i.e., had a possibly stressed benthic community) and none were in the "different" or "very different" categories. Sites 127, 164 and 179 were in depositional Zone 2 and site 175 was located in Zone 4 as shown in Figure 98. All four sites had an increased abundance of the chironomid midges *Chironomus, Procladius, Paralauterborniella* and the mite *Piona* (which was absent from the reference sites). The fingernail clam *Pisidium* was reduced at several of the Cornwall sites and absent from three of the four sites assessed as possibly stressed.

Results from the ordination of the taxonomic data were correlated with data for 28 sediment/water/geographic variables (Table 41). Seven parameters were significantly ($P<0.01$) correlated with the three ordination axes scores: sample depth, alkalinity (water), latitude, nitrate (water), total nitrogen (sediment), copper (sediment) and zinc (sediment). This information is shown in Figure 100. Each parameter is associated with a vector (arrow) on the plot pointing in the direction towards which values for that parameter increase. For example, on Figure 100 the vector shows depth increasing from right to left. The Cornwall test stations are clustered farther away from the direction that the depth vector points than most of the reference stations. This means that the test sites were shallower than most of the reference sites.

More notably, the Cornwall stations are clustered farther along in the direction of the copper and zinc vectors, indicating that there may have been some effect of these metals on the benthic community at the test sites. However, the community structure observed at these sites was only "possibly different" from the community at the reference sites (i.e., mildly degraded and possibly stressed) but not "different" or "very different" (severely stressed).

3.3.2.2 Toxicity, 1997 (Reynoldson, in prep.)

Reynoldson (in prep.) also examined the effect of exposure to Cornwall sediment from the same sites on survival in four species of aquatic invertebrate, growth in three species, and reproduction and development in one species. Tests were performed in the lab using sieved (500 µm), homogenated sediment. Sediment used for the toxicity tests had been stored for less than six months at 4° C in the dark, which should not have an effect on the toxicity data (Reynoldson, in prep.).

Results from ten test endpoints were compared with warning level values derived from 157 Great Lakes reference sites (Reynoldson and Day 1998). Results are summarized in Table 43 and provided in more detail in Table 44.

Table 43. Test organisms and endpoints used in Reynoldson (in prep.) toxicity study of St. Lawrence River (Cornwall) sediment collected October 1997. **Boldface** indicates endpoints for which toxicity was indicated.

Test organism	Test endpoint
chironomid *Chironomus riparius*	• % survival • growth (dry wt increase/individual)
oligochaete worm *Tubifex tubifex*	• % survival • **% hatch of cocoons** • number of cocoons/adult worm • **number of live young/adult worm**
amphipod *Hyalella azteca*	• % survival • growth (dry wt increase/individual)
mayfly *Hexagenia limbata* and *H. rigida*	• % survival • growth (dry wt increase/individual)

Only two toxicity endpoints compared unfavourably with warning levels derived from the Great Lakes reference sites. Both endpoints (cocoon hatching rate and numbers of young per adult) were related to reproduction in the tube worm, *Tubifex tubifex*. This raises the question of whether measures of reproduction would be affected in any other species of benthic invertebrates. In this study, only *T. tubifex* was tested for effects on reproduction.

Five Cornwall sites (105, 127, 156, 164, 167) had a reduced *Tubifex* cocoon hatching rate. As would be expected, all five sites also had reduced numbers of live young per adult worm. Cocoon hatching, stimulated by the development of the eggs in the cocoons into young worms (embryogenesis), was reduced which resulted in reduced numbers of young at the same sites. At two additional sites (117, 128) numbers of young per adult were reduced but cocoon hatching rates were unaffected. This could be due to a toxic effect on small rapidly growing individuals. At all sites, the ability of adult worms to produce cocoons (gametogenesis) and survival of adults exposed to sediment were unaffected.

Survival and growth were unaffected in the other three species studied, which were benthic dwelling insects rather than worms. No tests related to reproduction were done for these species.

The toxicity data were analyzed by means of an ordination approach similar to that used for the community structure data. The information was reduced to three variables represented by three ordination axes. Each test site was compared to the Great Lakes reference sites toxicity data.

Five Cornwall sites (105, 117, 127, 156, 167) were "possibly different" from the reference sites as determined by their position in ordination space relative to the position of the reference sites. These five sites thereby showed evidence of potential toxicity, largely due to reduced reproduction in *T. tubifex*. Site 167 was located in depositional Zone 1 and the other four sites were in Zone 2 as shown in Figure 98.

The relationship between toxicity and sediment quality was examined for 28 sediment, water and geographic variables including physical attributes (particle size), major elements and trace metals. Six variables were significantly ($P<0.01$) correlated with the three ordination axes scores.

These were, in descending order: %silt, $\%SiO_2$, Zn, %CaO, particle size and % loss on ignition. The two variables that appeared to be associated with movement of the Cornwall sites away from the reference group in ordination space were elevated levels of zinc and a high silt content as shown in Figure 101. Similar to the community structure data, the Cornwall sites were only "possibly different" (possibly toxic) compared with the Group 2 Great Lakes reference sites.

Table 44. Comparison of individual bioassay endpoints for 12 Cornwall sites with warning level values derived from Great Lakes reference sites (Reynoldson and Day 1998). **Boldface** indicates endpoints that compared unfavourably with warning levels. Source: Reynoldson (in prep.).

Test Endpoint:	*C. riparius* growth	*C. riparius* survival	*H. azteca* growth	*H. azteca* survival	*H. limbata* growth	*H. limbata* survival	*T. tubifex* CC/ad	*T. tubifex* hatch	*T. tubifex* survival	*T. tubifex* yg/ad
Warning level:	<0.21	< 67.8	<0.23	< 67.0	<0.9	< 85.5	< 7.2	<38.1	<89.0	< 9.9
Cornwall Stn #										
CS105	0.21	93.33	0.34	100.00	4.81	100.00	11.04	**35.41**	100.00	**6.62**
CS109	0.26	70.67	0.31	96.00	4.88	100.00	11.05	48.29	100.00	25.4
CS117	0.30	78.67	0.29	97.33	4.31	100.00	11.20	39.61	100.00	**1.1**
CS127	0.34	86.67	0.42	90.67	4.06	100.00	9.80	**24.38**	100.00	**1.9**
CS128	0.32	81.33	0.48	82.67	3.26	100.00	9.45	48.51	100.00	**8.55**
CS131	0.29	93.33	0.33	92.00	1.76	100.00	10.30	48.33	100.00	10.60
CS132	0.28	76.00	0.32	94.67	1.45	100.00	10.20	45.15	100.00	13.65
CS156	0.25	92.00	0.49	97.33	4.43	100.00	11.05	**27.85**	100.00	**3.55**
CS164	0.25	88.00	0.23	93.33	3.87	98.00	11.25	**35.87**	100.00	**5.05**
CS167	0.27	96.00	0.44	96.00	4.22	100.00	10.86	**31.89**	95	**4.65**
CS175	0.30	69.33	0.25	91.67	4.10	100.00	10.65	50.51	100.00	18.15
CS179	0.30	73.33	0.23	86.67	5.08	98.00	11.00	47.77	100.00	13.00

yg/ad, # young/adult; CC/ad, # cocoons/adult; growth, dry weight increase over the test period (e.g., 0.23=23% increase in dry wt during test); survival, % of individuals alive at end of the test period; Warning level, value (growth, survival or reproductive measures) below which the result is considered different from the result expected for reference sediment.

3.3.2.3 Overall Conclusions (Reynoldson, in prep.)

Overall results are summarized in Table 45 (Reynoldson, in prep.). None of the twelve sites showed definitive evidence of an impaired community or toxic sediment. Of the total of eight sites that showed some deviation from reference condition, only one (site 127) showed evidence of both sediment toxicity and a modified invertebrate community. However, Reynoldson (in prep.) reported that concentrations of contaminants at site 127 did not exceed the normal range observed at reference sites (mean + 2 SD). The main difference between sediment from site 127 and reference sites was the high proportion of gravel and sand and larger particle size at site 127. Although mercury concentration at site 127 was two times higher than the SEL, it was over four times lower than at site 131 which was non-toxic and had an unstressed community structure (Table 45).

Mercury was two times the severe effect level in sediment from site 127 but it exceeded the SEL in sediment from five other sites as well. Four of those five were non-toxic and had an unstressed benthic community and one of the sites was possibly toxic, as shown in Table 45.

In conclusion, there was "some evidence of mild degradation of invertebrate communities at four sites and possible toxicity at five sites", with only one site (127) showing both a possibly stressed community and possible toxicity (Reynoldson, in prep.). The results "suggest that factors other than sediment quality are largely responsible for the possible changes in community structure and that sediment contamination is not resulting in toxicity in the tests performed" (Reynoldson, in prep.).

Table 45. Summary of sediment quality based on invertebrate community structure, sediment toxicity and sediment chemistry. From: Reynoldson (in prep.). * observed concentration in µg/g.

Site	BEAST assessment		Variables exceeding MOE sediment criteria		Variables > 2 SD different from reference
	Community	Toxicity	> LEL	> SEL (*)	
105	unstressed	**possibly toxic**	TP, TOC, Cr, Ni, Cu, Zn, Pb, Hg		Zn
109	unstressed	non toxic	TP, TOC, Cr, Ni, Cu, Zn, Pb	Zn (823) Hg (4.8)	Cu, Zn, Pb
117	unstressed	**possibly toxic**	TP, TOC, Cu, Zn, Pb, Hg		Zn
127	**possibly stressed**	**possibly toxic**	TP, TOC, Cu, Zn	Hg (4.3)	Gravel, 25%PS
128	unstressed	non toxic	TP, TOC, Cu, Zn	Hg (11.2)	
131	unstressed	non toxic	TP, TOC, Cr, Ni, Cu, Zn, Pb	Hg (19.5)	Cu, Zn
132	unstressed	non toxic	TP, TOC, Cr, Ni, Cu, Zn, Pb	Hg (6.8)	
156	unstressed	**possibly toxic**	TP, TOC, Cu, Zn, Hg		
164	**possibly stressed**	non toxic	TP, TOC, Cr, Ni, Cu, Zn, Pb	Hg (3.1)	Cu, Zn
167	unstressed	**possibly toxic**	TP, TOC, Ni, Cu, Zn, Hg		
175	**possibly stressed**	non toxic	TP, TOC, Cr, Ni, Cu, Zn		
179	**possibly stressed**	non toxic	TP, TOC, Cr, Ni, Cu, Zn		

3.3.3 Environmental Effects Monitoring, Domtar Fine Papers, 1993

In October 1993, Domtar Fine Papers conducted Environmental Effects Monitoring (EEM) in the St. Lawrence River at Cornwall, Ontario as required by the federal *Fisheries Act, Pulp and Paper Regulation.* The EEM included an invertebrate community survey, summarized in the EEM final report as follows (Ecological Services for Planning Ltd. 1996):

> Benthos samples (70) were collected from 12 different areas (5 reference [sites 1,2,3,4,7] ; 2 near field [sites 5,6] ; 5 far field [sites 8,9,10,11,12]) [Figure 102]. A total of 101 different taxa were identified. All samples contained healthy levels of richness and abundance of benthic organisms.
>
> Statistical analysis of the benthos data following the Interpretive Guidance Document (IGD) methodology was undertaken on the 20 most abundant taxa. There was a high level of within-site variability which made it difficult to differentiate the 3 sampling areas (reference, near field, far field).
>
> Alternative methods of benthos data analysis from the IGD approach were also investigated using 18 taxa selected on the basis of biological significance. This approach identified significant differences in the benthic community in the near field samples versus the reference and far field samples. However, the differences could be primarily related to differences in habitat features rather than due to the effects of the mill.

For more detailed information on the EEM refer to Ecological Services for Planning Ltd. (1996).

3.3.4 Oil Tank Storage Area/Tank Farm Site, 1992-1993

Chemical characterization combined with biological assessment of sediment collected in 1992 and 1993 from Zone 3 (Figure 6) indicated that sediment clean-up was not required at this site (Metcalfe-Smith et al. 1995). Intensive sampling in 1992 and 1993 showed that the degree of sediment contamination at the oil tank storage area was not severe and the area affected was very localized (Metcalfe-Smith et al. 1995).

The results of these surveys are summarized in Figures 73A&B and 74A&B. Sediment was collected from the top 3 cm only (surface grab samples). In 1992, severe effect levels were exceeded at only one station and only for chromium and copper and in 1993 the SEL was not exceeded for any parameter. Oils and greases were higher than MOE Open Water Disposal Guidelines (Persaud et al. 1992) at almost all sites in both surveys as shown in Figures 73A and 73B. Maximum concentrations of total PAHs were 30-50% of the SEL at three stations (# 89, 91, 94) in 1992; PAHs were not measured in 1993 (Metcalfe-Smith et al. 1995).

In spite of the high concentrations of oils and greases at most sites, bioassessment of the 1993 samples did not indicate that sediments from the tank farm study area were particularly toxic or that the benthic community was severely degraded. Toxic effects were identified for at least one bioassay endpoint at five of twelve sites tested. But at each of these five sites, only one or two endpoints (tests) out of ten showed statistically significant toxicity, and the decreases in growth, survival or reproduction were not extreme. Only one station had a benthic community (abundance of organisms, number of taxa, percent distribution of taxa) that was degraded relative to comparable Great Lakes reference sites (Metcalfe-Smith et al. 1995).

Based on the combined chemical and biological findings from the 1992 and 1993 in-depth studies, it was concluded that sediment clean-up (removal and disposal) was not required at the tank farm site (Metcalfe-Smith et al. 1995).

3.3.5 Benthic Invertebrate Toxicity and Contaminants, 1991

Bioassays, 1991

Surface sediment was collected in 1991 from 6 stations in the AOC (368, 368A, 369, 370, 374, 375) and 2 upstream reference stations (82, 83) and bioassayed for toxicity to benthic invertebrates and short-term metal uptake in fathead minnows (Richman 1994; Bedard and Petro 1992). Station locations are shown in Figure 103. Two batches of sediment were collected: Batch 1 in January 1991 and Batch 2 in February 1991. Clean sediment from Honey Harbour, Georgian Bay was also used for comparison in all bioassay tests.

The following conclusions were drawn from the results of toxicity testing (Bedard and Petro 1992):

- Station 368 sediment was marginally toxic to *Chironomus tentans* (midge) larvae, registering the highest percent mortality (26%). For Batch 2 sediments, station 368 midge mortality was significantly different from the reference control sediment mortality of 2% but not from the negative control (Honey Harbour) sediment mortality of 11%.

- Sediment from stations 368 and 368A consistently elicited a sublethal response in both *Hexagenia* (mayfly) nymphs and *Chironomus* larvae. Mean mayfly growth (increase in wet weight) was reduced by 79% and 73% at stations 368 and 368A respectively, compared with reference site growth. Mean midge growth was reduced by an average of 89% and 33% compared with reference sites (Richman 1994).

- Growth of midge larvae exposed to station 369 sediment was significantly lower than for reference site sediment (36% reduction).

- *Chironomus* appeared to be the most sensitive indicator organism in terms of both toxic and sublethal effects.

- There was a significant positive correlation ($p<0.01$) between toxicity and concentration of mercury, zinc, copper, lead, cadmium and pyrene. Toxicity was not correlated with physical sediment properties.

Tissue concentrations of mercury in laboratory-exposed fathead minnows were reported as higher for all downstream stations than for the upstream reference stations (Lake St. Lawrence). As well, tissue concentrations of mercury, lead and copper in exposed minnows were reported as higher for station 368 than all for all other sites. However, no tests of significant differences were provided for the metal uptake data. Minnow tissue concentrations were positively correlated with sediment bulk and corrected (for % fines and TOC) concentrations of mercury, lead and zinc (Spearman Rank Correlation analysis, $p<0.05$) (Bedard and Petro 1992).

Sediment concentrations of mercury, zinc, copper and lead exceeded the SEL at station 368. Two other stations (368A and 369) also exceeded the SEL for mercury. Given the bioassay results, it was concluded that sediment contamination had a potential to impair native species, particularly at station 368 and possibly at station 368A and 369 (Bedard and Petro 1992).

The results of the sediment bioassays were consistent with results of the field collection of benthos, discussed below, in which benthos collected from stations 368 and 368A had significantly higher mercury concentrations than benthos collected from other stations.

Contaminants in Field Collected Benthos, 1991

In 1991, benthic invertebrates (chironomid larvae and oligochaetes) collected from 13 stations in the AOC and 2 upstream reference stations (Figure 103, all stations) were analyzed for total and methyl mercury (Richman 1994). Collected invertebrates were given time for gut clearance before preservation for analysis. Results of the *in situ* contaminants testing were consistent with the bioassay results and indicated that sediment mercury is taken up by benthic invertebrates.

Concentrations of total mercury in benthos were significantly higher at stations 368 and 368A than at all other stations, as shown in Table 46. These two stations were immediately adjacent to Courtaulds Fibres as shown in Figure 103.

For all stations, there was a significant positive correlation between the mean mercury concentration in benthos and total mercury concentration corrected for total organic carbon (TOC) ($r=0.80$, $p<0.003$). TOC indicates the amount of organic material in the sediment, which affects availability of mercury to benthic organisms: organic matter binds mercury, rendering it less available to benthos. The ratio of total mercury to TOC in sediment can therefore be used to normalize sediment mercury data, effectively removing the influence of the organic material so that comparisons of mercury concentration between stations with differing amounts of organic matter can be made.

3.3.6 Benthic Invertebrate Communities, 1986

Benthic invertebrates were collected from 10 St. Lawrence River stations in September 1986 as shown in Figure 104, as part of an MOE benthic environmental assessment program (Jaagumagi 1987). It was concluded that benthic invertebrate distribution and density appeared to be most affected by the organic content of the sediments and occurrence of aquatic macrophytes. There was no evidence of effects due to other factors, even though a "black, sticky, oil-like substance" was observed in sediment from stations 366, 368, 368B and 371A and metal concentrations (mercury not examined) were above MOE guidelines at many sites.

Table 46. Total mercury concentration (ng/g wet weight) in benthos collected from the St. Lawrence River, 1991 (Richman 1994).

Station number	Total mercury concentration in benthos (ng/g wet wt)		Ratio of total mercury/TOC in sediment
	mean	std deviation	
82 (upstream reference)	26.5	4.9	4.2
83 (upstream reference)	11.9	2	8.5
365	31.8	4.4	42.6
366	12.2	0.5	4.4
368	55.8	26.8	88.7
368A	68	26.3	100.0
369	24.3	9.4	42.6
369A	24.4	7.8	42.6
370	7.7	2.3	3.6
370A	24	1.6	84.3
371	19.6	3	29.7
372	6.4	0.8	8.4
376	14	1.5	19.7
373	4.7	0.9	4.8
374	9.2	1	4.4

3.4 Other Biota

Environment Canada, Canadian Wildlife Service has done a number of studies on biota other than fish and benthic invertebrates in the Cornwall area. These are summarized here.

3.4.1 Mudpuppies

The mudpuppy (*Necturus maculosus*) is a freshwater amphibian resembling a salamander. Body burdens of organochlorine contaminants and incidence of digital deformities have been studied in mudpuppies collected along the south (Akwesasne) and north (Cornwall) shores of the St. Lawrence River. South shore results are related to PCB contamination from US sources and are therefore not included here.

Mudpuppies from the north shore were collected in April 1996 from the sites shown in Figure 105. Incidence of digital deformities and concentrations of chlorinated hydrocarbons were measured. Deformity results are shown in Table 47 (Bishop 1997).

The sample size was small (15 animals) because there were apparently very few mudpuppies in the area. Fifty traps set on each of two nights yielded only 15 mudpuppies. In contrast, the same number of traps over two nights in April 1994 yielded 50-70 animals in wetlands of Lake Erie and eastern Lake Ontario. The habitat along the Cornwall waterfront is not that suitable for mudpuppies, which prefer marshy areas. It is possible that populations of mudpuppies at Cornwall may never have been very large or they may have been reduced by the impact of habitat loss or contaminant exposure (Bishop 1997).

Table 47. Incidence of digital deformities in mudpuppies collected at five trap lines along the Cornwall waterfront in April 1996 (Bishop 1997).

Site	# individuals with deformities	# caught (N)
Line #1	2	3
Line #2	1	4
Line #3	1	4
Line #4	1	2
Line #5	1	2

Among the few mudpuppies that were caught the incidence of extra or missing digits was high (7/15 individuals). However two of the three individuals caught at the upstream location also had deformities, suggesting this may not have been a sufficient control site (Bishop 1997). The site was only about 2 km upstream of Cornwall industrial point sources and the mudpuppies in this area could ingest contaminants from prey (fish and their eggs) ranging over a larger area than the mudpuppies themselves. The home range of the mudpuppies is less relevant for sampling purposes than the range of their prey.

The 46% incidence of deformities in the April 1996 collection from the Cornwall waterfront was much higher than the incidence (<10%) found at sites of low organochlorine and mercury contamination on the Ottawa River and St. Lawrence River in 1992, 1993 and 1995 (Gendron et al. 1994).

3.4.2 Northern Leopard Frogs

The following text is reproduced verbatim from the conference abstract of results presented by C.W. Bishop, Environment Canada (Bishop et al. 1999):

> In 1998, deformity rates in northern leopard frogs were assessed at Hoasic Creek (reference site), Gray's Creek and Cooper's Marsh. Newly hatched tadpoles were collected from Guindon Park, Cornwall. Fifty tadpoles were placed in each of four cages per study site. Tadpoles were fed boiled lettuce from mid-May to mid-July when they had completely transformed into frogs. Survival and deformity rates in these caged amphibians showed few differences among sites. The percentage of tadpoles surviving to metamorphosis at each site was: 79% (Cooper's Marsh); 93% (Gray's Creek); 96% (Hoasic Creek). Percentages of animals that transformed into frogs and showed at least one gross morphological deformity were: 1.3% (Cooper's Marsh); 2.2% (Gray's Creek); 0% (Hoasic Creek). Background rates of deformities in wild frogs are usually 1%.

3.4.3 Tree Swallow Eggs

The following information is excerpted from Bishop et al. (1999):

In 1998, several wildlife toxicology studies were initiated in the St. Lawrence Area of Concern. Contaminants were measured in snapping turtle and tree swallow eggs and investigations of the biological effects of these contaminants began. Tree swallow eggs were collected from four sites, three on the north channel and one on the south channel of the river. Snapping turtles were studied only in the south channel of the river and are therefore not discussed here. The study continues in 1999.

Tree swallow eggs were collected from three north channel sites (Fly Creek; near the Cornwall WPCP outfall; Canadian side of Cornwall Island) and one south channel site (Loran's Marina, Hogansburg). Eggs were collected from a variety of nest boxes at each site; egg contents were pooled and analyzed as a single sample.

Total PCB concentrations in tree swallow eggs were 0.9-2.2 µg/g wet weight at the north channel sites. Total mercury concentrations were low and not variable (0.22-0.31 µg/g wet wt) at all four sites. Sex ratios in 16 day-old tree swallow chicks were 1:1 at Fly Creek and the north shore of the St. Lawrence but at Cornwall Island the ratio was 2:1 favouring females. The Cornwall Island ratio is quite unusual as all sites studied within the St. Lawrence Area of Concern and seven sites previously studied by Environment Canada in southern Ontario showed 1:1 sex ratios in 16 day-old tree swallows.

3.4.4 Marsh Monitoring

Marsh bird and amphibian communities in the AOC have been monitored through the Marsh Monitoring Program, a joint undertaking of Environment Canada, the Long Point Bird Observatory and the US Great Lakes Protection Fund. Volunteers are trained to identify specific marsh birds and amphibians by sound and to record numbers of calls of each type in a set period of time at specific points along a monitoring route. The numbers recorded are extrapolated to provide an estimate of abundance of specific birds and amphibians within the marsh.

Overall, based on the results of 1994, 1995 and 1996 marsh monitoring, the AOC appeared to be unimpaired in terms of its ability to support healthy marsh bird and wetland amphibian populations. The following brief summary is drawn from a summary report on 1995-1996 results (Long Point Bird Observatory 1998).

In total, 8 amphibian and 7 marsh bird monitoring routes have been established in the AOC. The number of amphibian species present ranged from 4-7 per route, with an overall total of 9 species recorded. Densities of most species were moderate to high. Of the expected five amphibian indicator species, only one (mink frog) was absent. Three species scored as average or above average in abundance and chorus frog was below average. In terms of overall amphibian presence, there appeared to be some impairment because two mainly terrestrial species (tree frog and American toad) were absent. However, with respect to the wetland species of amphibians, the AOC was not impaired.

The number of marsh nesting bird species ranged from 7-20 per route. Overall, 25 species of marsh nester and 10 foraging (feeding but not nesting in the marsh) species were recorded. In general, densities were equal to or greater than Great Lakes basin averages outside Areas of Concern. Eleven of the twelve marsh bird indicator species were present; only Least Bittern was not. Abundances of 9 species were average or above average. Sora and American Bittern were below average in abundance.

While the degree of impairment varied for individual marshes, overall, the AOC does not appear to be impaired in its capacity to support healthy marsh bird and amphibian populations.

4. INFORMATION REQUIREMENTS

A great deal of data has been gathered and examined over the past thirty years for the Cornwall section of the St. Lawrence River. As in any complex scientific investigation there is always additional data that would be valuable.

At present, the biological effect of sediment contaminants is the most poorly understood area of study related to Cornwall sediment contamination. This is not unique to Cornwall, however. See Jaagumagi and Persaud (1996) for a discussion of the complexities of biological investigations and contaminated sediment.

Donna Bedard, MOE (pers. comm., October 26, 1999) emphasized the following points for future studies of biological effects at Cornwall:

- A more extensive sportfish sampling program would be useful to acquire a sample size sufficient for comparisons among regions of high, moderate and low mercury contamination. Also analysis on different age groups may help project the rate of accumulation and determine whether current trends are different from past trends as an indication of ongoing source problems.
- Sportfish sampling should include an analysis of gut content both in terms of mercury concentrations and prey identification. This will provide a clearer picture of routes of uptake and degree of biomagnification.
- Caged fish and caged mussels should be used for long-term exposure studies in areas of high, moderate and low mercury contamination. Different sets of cages can be placed at the sediment surface versus suspended in the water column to examine the extent of uptake derived from pelagic *versus* deposited + pelagic routes. The suspended solids data suggest concentrations peak at certain times of the year and should be factored into the study design.
- Sampling and tissue analysis of *in situ* benthos should be continued with a sample size large enough to allow for rigorous statistical interpretation and current conditions.

Bedard (pers. comm., 1999) also noted that "the use of laboratory sediment bioassays is probably not the best approach for identifying biological effects associated with [either] mercury or maximum bioaccumulation. There is probably no need to include such tests in future studies. Both MOE and EC test results indicate few significant effects, that were limited to only one species and one endpoint."

N. Rukavina (Environment Canada, National Water Research Institute, pers. comm. October 1999) identified the following additional types of missing data:

- Information on bioturbation of sediments and its possible effect on the transport of contaminants to the surface.
- Data on flow rates, their seasonal and annual changes and the effect on temporal changes in suspended sediment concentrations.
- Data on shoreline or bottom changes as a result of Seaway development which could affect sediment transport or accumulation in Zone 2.
- Information about sediment accumulation rates in depositional zones located to the east and southeast of Pilon Island (Figure 6).

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LIST OF ACRONYMS

ADCP	acoustic Doppler current profiler. An instrument used to acoustically determine direction and velocity of water flow at various depths throughout the water column.
AOC	Area of Concern (Great Lakes/St. Lawrence River Remedial Action Plan Program)
BSAF	biota-sediment accumulation factor. The ratio of contaminant concentrations in sediment versus tissue of organisms exposed to the sediment.
EC	Environment Canada
EEM	Environmental Effects Monitoring. Environmental monitoring at pulp and paper plant sites as required by federal legislation under the Fisheries Act.
GIS	geographic information system. A type of computerized system for analyzing and mapping environmental data.
HMDS	hybrid multi-dimensional scaling (a method of data ordination (i.e., data sorting).
ICI	ICI Forest Products (formerly CIL)
LEL	"lowest effect level" from 1992 Provincial Sediment Quality Guidelines. This is the level of sediment contamination that can be tolerated by the majority of benthic organisms. Concentrations above this level indicate that the benthic community may be impaired.
LSF	Lake St. Francis
LSL	Lake St. Lawrence
LTSS	long term sensing study
MISA	Municipal-Industrial Strategy for Abatement
MNR	Ministry of Natural Resources (Ontario)
N	number of samples
MOE	Ministry of the Environment (Ontario)
NWRI	Environment Canada National Water Research Institute, Burlington, Ontario
PAH	polyaromatic hydrocarbon
PCA	principle components analysis
PCB	polychlorinated biphenyl

PWQO	Provincial Water Quality Objective
RAP	Remedial Action Plan. A Canada-US program to restore environmental quality in 43 Great Lakes/St. Lawrence River Areas of Concern.
SD	standard deviation
SEL	"severe effect level" from 1992 Provincial Sediment Quality Guidelines. This is the level of sediment contamination that is expected to be detrimental to the majority of benthic organisms.
SFC	St. Francis Centre (LTSS site)
SFS	St. Francis South (LTSS site)
SFN	St. Francis North (LTSS site)
SPM	suspended particulate matter
TCTI	Transport Canada Training Institute (LTSS site offshore from NavCanada, formerly TCTI)
TOC	total organic carbon
TPH	total petroleum hydrocarbons
ULN	upper limit of normal
WPCP	water pollution control plant (=sewage treatment plant)