The Impacts of Municipal Wastewater Effluents on Canadian Waters: A Review

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Abstract

Domestic sewage is a major threat to receiving waters throughout the world. In Canada, a high proportion of the population (81%) is served by municipal wastewater treatment facilities. Nevertheless, discharges from wastewater treatment plants, stormwater sewers and combined sewers have caused some adverse impacts on lakes, rivers and coastal waters. The most publicly recognized impacts are shellfish harvesting restrictions and beach closures resulting from microbial contamination. Habitat degradation and contamination also occur and these, in turn, have altered the abundance and diversity of aquatic organisms. Our findings on the effects of municipal wastewater discharge suggest that there is a need to review sewage treatment requirements in Canada. Further research is also required on the interactive and cumulative responses to habitat degradation and to long-term exposure to persistent and bioaccumulative pollutants. Finally, an integrated approach to wastewater management is needed that addresses loadings from treatment plants, stormwater sewers, CSOs and other wastewater sources.

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Introduction

Rivers, lakes and coastal waters have long been used as receptacles for diluting and dispersing domestic waste. Water pollution related to sewage discharge probably can be traced back to the foundation of the first cities 7000 years ago along the Tigris-Euphrates and Indus rivers. However, it only became a severe problem during the industrial revolution when wastes produced from industrial processes along with domestic sewage from an increasing urban population were poured untreated into the nearest waters. Classic examples of long-term water pollution from domestic sewage are described for the Thames River, England (Gameson and Wheeler 1977), and Boston Harbor and Chesapeake Bay, USA (National Research Council 1993). Yet, while there have been considerable efforts since the 1970s by developed nations to improve sewage treatment, discharge of untreated or poorly treated sewage is still a concern in many parts of the world, particularly developing countries and those experiencing transition economies (e.g., Russia and central Europe). In the case of developing countries, >90% of urban sewage is discharged directly into surface water without treatment (World Resources Institute 1996). However, even in many developed countries, only a portion of municipal sewage receives conventional treatment. For example, only 60% of the total population of countries belonging to the Organisation for Economic Co-operation and Development was connected to a sewage collection system and served by domestic wastewater treatment in the late 1980s (OECD 1995).

The impacts of domestic wastewater discharge on receiving water are numerous. Sewage poses a direct health risk from pathogens such as bacteria (e.g., cholera, salmonella, shigellas), viruses (e.g., hepatitis, enteroviruses, poliovirus, Norwalk) and parasites (e.g., protozoans such as *Giardia* and *Cryptosporidium*, helminths*)* (World Health Organization 1993; World Resources Institute 1996). Indirect human health risks include consumption of fish or shellfish rendered toxic by bacteria, metals or organic compounds associated with sewage (Waldichuck 1989), or exposure to microbially-contaminated waters during recreational activities (Edsall and Charlton 1996). In addition to human health risks, domestic wastewater discharge poses environmental risks. Loading of nitrogen and phosphorus can lead to eutrophication resulting in radical changes in productivity and biodiversity, inputs of contaminants can cause acute or chronic toxicity of organisms in receiving waters, while high loads of oxygen-demanding material can reduce dissolved oxygen to levels that threaten the survival of aquatic organisms (e.g., Meybeck et al. 1989; National Research Council 1993).

The aim of this study was to review the consequences of municipal wastewater discharge to Canadian lakes, rivers and coastal waters. Canada is often seen as a privileged country because of the abundance and quality of its water resources. However, concerns raised by international agencies about sewage disposal world-wide (United Nations 1992a; UNEP1995) and by public groups about the Canadian situation (Sierra Legal Defence Fund 1994; Kapitain 1995; Nantel 1995, 1996a,b) emphasized the need to evaluate the current status of municipal wastewater discharges and their effects on the Canadian environment. In this review, we summarized the current risks to human health and the environment from municipal wastewater discharge in Canada. Whereas the historical focus

of municipal wastewater management has been on discharges from municipal wastewater treatment plants (MWTPs), it is now recognized that municipal wastewater management should also take into account stormwater discharges and combined sewer overflows. Thus, in this paper, we examined the effect of discharges from MWTPs, stormwater and combined sewer overflows. Discharges from lagoons and septic systems have not been considered as their risks to human health and the environment are difficult to generalize due to a paucity of Canadian information.

Municipal Wastewater Status in Canada

Municipal Wastewater Treatment Plants

In 1992, there were approximately 2800 MWTPs in Canada that discharged approximately 1 x 10^7 m^3/d (300 L/capita/d) of municipal wastewater to the environment. These ranged from primary wastewater treatment facilities (employing physical processes to decrease influent biochemical oxygen demand (BOD) by 20 to 30% and total suspended solids (TSS) by about 60%), to secondary treatment facilities (employing additional biological treatment, such as anaerobic and/or aerobic microorganisms, to decrease influent BOD and TSS by 80 to 95%), to tertiary treatment facilities (employing advanced chemical or biological treatment to remove specific compounds or materials that remain after secondary treatment). In addition to that, some MWTPs chlorinate their effluent on a seasonal or continuous basis. Of all MWTPs in Canada, approximately 14% discharge chlorinated effluent for a total chlorinated discharge of 6.11 x 10^6 m^3/day (Government of Canada 1993). The volume of effluent discharged from municipal sources in Canada is large in comparison with wastewater discharge from many industrial sectors (Table 1). In 1986, domestic wastewater discharge in Canada totaled 5.2 km^3 and was the single largest source of wastewater, with the exception of electricity production (United Nations 1992b).

Table 1. Wastewater generation in Canada by sector for 1986 (United Nations 1992b).

Sector	**Wastewater Generation (km^3)**
Domestic	
non-treated	1.5 (estimate)
treated	3.7
Industrial[1]	
Total mining sector	0.4
Manufacture of paper and paper products	2.8
Manufacture of chemical and chemical products	1.6
Manufacture of metallic mineral products	1.8
Other manufacturing	1.4
Electricity production	25.1
Other	1.5

[1]Industrial sector classifications are based on the International Standard Industrial Classification (ISIC) of all economic activities.

The percentage of Canadians served by wastewater treatment has increased in recent years. As of 1994, 81% of Canadians were served by some level of treatment (Table 2), whereas < 56% of Canadians were served in 1980. In Quebec alone, the population served by municipal wastewater treatment increased from 2 to 75% between 1980 and 1994 (MEFQ 1995). The proportion of the population served by wastewater collection and treatment varies across the country. Municipal wastewater discharges to inland waters in Canada generally receive secondary or tertiary treatment, whereas discharges to coastal waters often receive only primary or no treatment. As of 1986, approximately 5 x 10^5 m^3/day of municipal waste in Atlantic Canada were discharged into coastal waters with only 30% receiving wastewater treatment (Environment Canada 1986). On the Pacific coast, the British Columbia government registered over 250 marine sewage discharges in 1986. In 1990, there were over 300 permits or applications for sewage discharge to coastal waters, accounting for about 20% of the effluent discharged to British Columbia coastal waters from all sources (Wells and Rolston 1991). In the Arctic, Wells and Rolston (1991) reported that in the 1980's untreated or primary treated municipal wastewater from nine communities was discharged directly to the marine environment or by percolation through lagoon substrates with subsequent leaching into surface drainage system. In the Northwest Territories, alone, 35 of the 59 organized communities (or 31% of the 56808 population in these communities) have no wastewater treatment, with approx. 14 communities (approx. 15% of total population) discharging to coastal waters (UMA 1993).

Table 2. Percentage of Canadian population served by wastewater treatment in 1994. (Data obtained from D. Tate, Environment Canada, Hull, Quebec, pers. com.)

Region	None[1] (no sewer)	None (sewers)	Primary	Secondary	Tertiary
Atlantic	22	40	11	26	<1
Quebec	11	14	36	30	9
Ontario	10	<0.1	5.7	15	69
Prairies	5	<1	7	61	27
BC & Territories	18	2	54	20	6
Total	12	7	20	27	34

[1] May have individual on-site systems.

Stormwater and Combined Sewer Overflows

Surface runoff from urban areas is usually transported in separate storm sewer systems (i.e., separated from sewage) or combined sewer systems (i.e., combined with sewage). The volume of surface runoff varies in proportion to the degree of imperviousness of the ground

area. For an entire urban area, 30 to 50% of the total rainwater volume may be converted into surface runoff that is then fed into a separate or combined sewer system (Falk 1983). Stormwater from a separate sewer system is either discharged directly to the receiving water or passed through stormwater management facilities which may reduce flows and improve water quality (Marsalek and Kok 1997). In the case of a combined sewer system, however, the whole flow is conveyed to a wastewater treatment facility under low flows but, during high flows that would otherwise exceed the sewer system or MWTP capacity (i.e., during heavy rainstorms), some flow is diverted out of the sewer system to receiving waters at overflow structures. These overflows are referred to as combined sewer overflows (CSOs) and contain both surface runoff and municipal sewage. CSOs may be highly polluted and typically are discharged to nearby receiving waters without treatment. In relatively few locations, however, special facilities for storing and treating CSOs have been built (Marsalek and Kok 1997). MWTP bypasses are also found in some cities. As these divert flow immediately upstream of the MWTP, bypass characteristics are similar to those of CSOs.

Detailed data are not available on the proportion of the Canadian population served by storm sewer versus combined sewer systems. However, most urban areas developed prior to the early 1940s are served by CSOs. Waller (1969) estimated that about 6.7 million Canadians were served by combined sewers in 1969. In large cities, tens of CSO outfalls exist and contribute to a great spatial extent of CSO impacts on a single receiving waterbody. For example, in Greater Vancouver, British Columbia, 252 stormwater outfalls and 53 CSOs discharge to the lower Fraser River and estuary (UMA 1994, 1995). Given that combined sewers were used mostly in the older sections of cities where populations are now declining and that sewer separation programs have been undertaken by some communities during the past 25 years, the current Canadian population served by combined sewers is likely smaller than in 1969.

The quantities of both stormwater and CSOs vary temporally and with location, and depend on local climate, sewer design and drainage practices. CSO or stormwater discharges are not routinely monitored in Canada; therefore, estimates of their volume and impact on receiving waters are rare. However, detailed measurements of stormwater or CSO quality can be integrated over a large drainage area to give approximate discharges and loads. Thus, large scale estimates gave an average annual stormwater discharge of about 760 L/capita/d for the Great Lakes basin (Marsalek and Schroeter 1988). However, the stormwater discharges would be 2000 to 3000 L/capita/d if the averaging was done just for the wet-weather days. The average annual discharge was about 473 L/capita/d for urban runoff and CSOs in the Greater Vancouver Regional District (Environment Canada 1992 from GVRD 1988). These flows greatly exceed the average municipal sewage flow of 300 L/capita/d.

Municipal Wastewater Effluent Characterization and Loadings

Municipal Wastewater Treatment Plants

Characterization

Effluents from MWTPs are derived from both household and industrial sources and consist of suspended solids, microorganisms, debris, and about 200 chemicals (Environment New Brunswick 1982; Birtwell et al. 1988; OMOE 1988). While municipal wastewater effluent (MWWE) contains a wide variety of constituents, they can generally be described by the following categories: solids; suspended and dissolved substances that exert a biochemical oxygen demand; nutrients; pathogens; organic chemicals; metals; oil and grease; and, plastics and floatables. Of these, total suspended solids (TSS), biochemical oxygen demand (BOD) or chemical oxygen demand (COD), nutrients in the form or phosphorus (P) and/or nitrogen (N), pathogenic bacteria, and plastics and floatables are the primary targets for removal through wastewater treatment. Concentrations of these conventional parameters are given in Table 3 for selected Canadian cities and different treatment types.

In addition to conventional substances, recent surveys (OMOE 1988; Rutherford et al. 1994; Golder Associates Ltd. 1995a,b) have shown that toxic substances including metals and organic chemicals are present in MWWEs across Canada (Tables 4 and 5). In particular, a wide variety of synthetic organic chemicals have been found in MWWEs in Canada (Table 6). While their concentrations might not be very high, many of these organic chemicals are toxic and persistent in the environment. In Ontario, for example, a study of 37 MWTPs with various types of treatment reported up to 24 base/neutral and acid extractables, seven dioxins and furans, 30 pesticides and herbicides, and 19 volatiles, along with 15 metals and cyanide (OMOE 1988). Effluents from two MWTPs in Edmonton contained 19 metals and 3 to 7 organic compounds (Golder Associates Ltd. 1995a,b) whereas up to 48 organic chemicals including detergents, solvents, chlorinated compounds, plasticizers, fecal-derived compounds such as dihydrocholesterol, and miscellaneous compounds such as caffeine and nicotine were found in effluent from four Nova Scotia MWTPs (Rutherford et al. 1994). Industries are often a significant source of metals and organic compounds; however, some of the metals and organic compounds in MWWE are derived from household waste. For example, metals such as copper, zinc, iron, cobalt, manganese and molybdenum are essential elements in human nutrition. Consequently, small concentrations of metals and other toxic chemicals that are by-products of human physiological functions will always be found in municipal wastewater. In addition, aluminum can be introduced from cooking ware and antacids, tin can be introduced from canned foods, while household cleaning agents can introduce a variety of chemicals. Thus, U.S. EPA (1986) found that approximately 20% of metals in U.S. wastewaters are from domestic sources.

Table 3. Concentrations (mg/L) of conventional parameters and annual discharge (x 10^6 m^3) for selected municipal wastewater treatment plants in Canada. (Blanks indicate data are not available; sample years given in brackets after parameter value if different from year(s) given in "Date" column.)

Location and MWTP type	Date	BOD	COD	Total Suspended Solids	Ammonium	Nitrite	Nitrate	Total Phosphorus	Effluent Discharge
Alberta									
Calgary secondary[1]	1980	16	65	14	16			3.5	132
Calgary secondary + P removal[2]	1985							<1	
Goldbar Edmonton secondary[3]	1982-83	22	38					3.7	90
Goldbar Edmonton secondary[4]	1980-93			9	20	0.5		3.1	114
Capital Region Edmonton tertiary[4]	1980-93			5	0.6	0.4		5.4	17
Grande Prairie primary[5]	1988-93	8.6						5.0	4
British Columbia									
Iona Island Greater Vancouver primary[6] Annacis	1985	81		57	8.8			2.9	170 (1987)
Island Greater Vancouver primary[6]	1985	156		71	16			4.5	107 (1987)
Lulu Greater Vancouver primary[6]	1985	139		64	20			5.3	15 (1987)
Prince George secondary[7]	1985-91	19[7]		30[7]	26	1.2	1.6	5.0	11 (1993-94)
Macaulay Point Victoria primary[8]	1992	224		320	28				} total 36x10^6 m^3 for
Clover Point Victoria primary[8]	1992	192		186	18				both outfalls
Nova Scotia[9]									
2 secondary	fall 1991	<5-50		25-45	<0.14-28				
Eastern Passage primary	fall 1991	80		31	18				4.4 (1986)
Lakeside tertiary	fall 1991	<5		12	2.0				0.5 (1986)
Ontario[10]									
7 primary	1987	48	109	30	10	0.00	0.05	1.34	291
28 secondary	1987	21	53	10	3.9	0.22	2.33	0.68	1044
1 tertiary	1987	25	99	32	18	0.05	0.14	1.56	16
Quebec[11]									
Montreal primary + physical/chemical treatment with ferric chloride	1993	38	105	20	5.9			0.5	412
Saskatchewan									
Saskatoon primary	1985-89	83[13]		74[13]	19 (1987)[12]			5.2[13]	32 (1987)[12]
Saskatoon primary + P removal[13]	1993-95	78		24				0.96	34
Saskatoon secondary[13]	1996	44		19				1.12	30
Yukon[14]									
Carmacks secondary	Oct 93	18	56	10	0.2	0.11	27		0.03

[1] Hamilton and North 1986
[2] Charlton and Bayne 1986
[3] Anderson et al. 1986
[4] Golder Associates Ltd. 1995a,b
[5] Chambers 1996; Chambers and Mills 1996
[6] Environment Canada 1992
[7] French and Chambers 1995; City of Prince George records (BOD values are carbonaceous BOD (CBOD); CBOD and TSS from 1991-1994)
[8] EVS Consultants 1992; Taylor et al. 1995
[9] Rutherford et al. 1994, except discharge from Environment Canada 1986
[10] OMOE 1988
[11] CUM 1994; Deschamps et Ceijka 1993; PAHs and PCBs from Pham and Proulx 1996
[12] Saskatchewan Environment and Public Safety 1989
[13] City of Saskatoon records
[14] Enns and Soprovich 1995
[15] Environmental Management Assoc. & Hydroqual Laboratories Ltd. 1993

Table 4. Metal concentrations (µg/L unfiltered total, unless otherwise indicated) for selected municipal wastewater treatment plants in Canada. (Footnotes as for Table 3. Blanks indicate data are not available; nd is below detection limits.)

Location and MWTP type	Date	Al	As	Cd	Cr	Co	Cu	Fe	Pb	Hg	Mn	Mo	Ni	Se	Ag	Zn	Sr
Alberta																	
Calgary secondary[1]	1980	60	1	<1	48	2	21	115	18		34		8			114	
Calgary secondary + P removal[2]	1985																
Goldbar Edmonton secondary[3]	1982-83								3		43		2			53	
Goldbar Edmonton secondary[4]	1980-93	60[a]	0.8	2	21	1	5	110	4	0.1	54	10	22	0.2	20	53	
Capital Region Edmonton tertiary[4]	1980-93	76[a]	1	2	9	1	10	89	3	0.1	75	3	8	0.2	20	85	
British Columbia[6]																	
Iona Island Greater Vancouver primary	1985				nd		101	889	43		60		nd			120	
Annacis Island Greater Vancouver primary	1985	799			nd		141	1799	41		99					171	
Lulu Greater Vancouver primary	1985	2789		1.7	150		160	2692	58		90		150			340	
Nova Scotia[9]																	
2 secondary	fall 1991	280-12000					20-120										
Eastern Passage primary	fall 1991	320					20										
Lakeside tertiary	fall 1991	340					30										
Ontario[10]																	
7 primary	1987	550	nd	2.5	11	6.5	18		21	0.05		7	9	17	6.4	70	305
28 secondary	1987	102	17	2.1	9	6.4	13		17	0.03		7	22	17	6.9	53	341
1 tertiary	1987	1252	Nd	6	69	9	55		56	0.17		11	26	nd	6.6	960	1171
Quebec[11]																	
Montreal primary + physical/chemical treatment with ferric chloride	1993	1009	1	1.8	8		26	2000	7				14		1.8	58	
Saskatchewan[12]																	
Saskatoon primary + P removal	spring92		0.5	<1	11	<1	33		<5	0.09		13	4	2	2		
	summ92		<.5	<1	5	<1	44		9	0.31		5	26	2	1		
	fall 92		3.2	<1	14	<1	51		<5	0.16		21	4	<1	3		
Yukon[14]																	
Carmacks secondary	Oct93	690					90	510			210					160	390

[a] Extractable

Table 5. Concentrations of organic contaminants for selected municipal wastewater treatment plants in Canada. (Footnotes as for Table 3. Blanks indicate data are not available; nd is below detection limits.)

Location and MTWP type	Date	Chlorinated Solvents (µg/L)		PCB (ng/L)	PAHs (µg/L)		Phenols (µg/L)	
		Tetrachloro ethylene	Tricholoro ethylene				Chlorinated	Non-Chlorinated
Alberta								
Goldbar Edmonton secondary[3]	1983						16[a]	
Goldbar Edmonton secondary[4]	1992-93	1.5					13[a]	
Capital Region Edmonton tertiary[4]	1992-93						8[a]	
British Columbia								
Iona Island primary Greater Vancouver[6]	1985						30[a]	
Annacis Island Greater Vancouver primary[15]	1993 1985	5.9	1.5				51[a] (1985 data[4]) 39[a]	
Lulu Greater Vancouver primary[6]								
Ontario[10]								
7 primary	1987	4.39	1.71	30	nd[b]	nd[c]	nd[d]	1.78
28 secondary	1987	1.18	1.12	20	1.08[b]	1.61[c]	2.71[d]	1.65
1 tertiary	1987	3.50	1.26	50	1.29[b]	1.85[c]	3.10[d]	1.99
Quebec[11]								
Montreal primary + physical/chemical treatment with ferric chloride	1993			1.10[e]	0.00[b] 0.66[f]	0.03[c]	nd	12

[a] all phenols; [b] benzo(*a*)anthracene; [c] pyrene; [d] pentachlorophenol; [e] for 13 PCBs; [f] for 21 PAHs.

Table 6. Organic chemicals detected in municipal wastewater treatment plant effluents.

(1,1'-Biphenyl)-2-ol [1]	Benz(a)Anthracene [2]
1,1,1-Trichloroethane [2,3]	Benzo(k)Fluoranthene [2]
1,1-Dichloroethane [2]	Beta-hexachlorocyclhexane [2,4]
1,1-Dichloroethene [2]	Bis(2-Chlorethoxy)Methane [2]
1,2,4-Trichlorobenzene [2]	Bis(2-Chloromethyl)Ether [2]
1,2-Benzenedicarboxylic acid, diethyl ether [1]	Bis-2-ethylhexylphthalate [4]
1,2-Benzenedicarboxylic acid dibutyl ester [1]	Bromodichlorobenzene [2]
1,2-Benzenedicarboxylic acid, bis(2-ethylhexyl) ester [1]	Butylbenzylphthalate [2,4]
1,2-Dichlorobenzene [3]	Caffeine [1]
1,2-Dichloroethane [2]	Carbon tetrachloride [2]
1-Methyl-5-(3-pyridinyl)-2-pyyrolidinone [1]	Chlorobenzene [2]
1-Octene [2]	Chloroform [3]
2,4,5-Trichlorophenoxyacetic acid [2]	Chlorodibromomethane [2]
2,4,6-Trichlorophenol [2]	Cholest-5-en-3-ol [1]
2,4-D-propionic acid [4]	Cis-1,2-Dichloroethylene [2]
2,4-Dichlorophenol [2]	Cyclododecane [1]
2,4-Dichlorophenoxyacetic acid [2]	Cyclohexadecane [1]
2,4-Dichlorophenoxybutyric acid [4]	Decanoic acid [1]
2,4-Dimethylphenol [2]	Di-N-butylphthalate [4]
2,4-Dinitrotoluene [2]	Di-N-octylphthalate [3]
2,6-Dinitrotoluene [2]	Diacetin [1]
2-(2-Butoxyethoxy)-ethanol [1]	Dichlorobenzoic acid [1]
2-(Methylthio)benzothiazole [1]	Dichlorodifluoromethane [2]
2-Butoxy ethanol [1]	Dichlorophenol [1]
2-Butoxy, phosphate ethanol [1]	Dieldrin [2]
2-Butyl-phosphate ethanol [1]	Diethyl Ether [2]
2-Chlorophenol [2]	Dihydroxycholesterol [1]
2-Ethyl-1-hexanol [1]	Dimethyl Phthalate [2]
2-Methyl-3-hydroxy-2,4-trimethyl propanoic acid [1]	Dimethyl-(methylethyl)benzene [1]
2-Methyl-4,6-Dinitrophenol [2]	Dimethylethylbenzene [1]
2-Nitrophenol [2]	Dodecanol [1]
3-(1-Methyl)-2-pyrrolodinyl) pyridine [1]	Endosulphan I [2]
4-Nitrophenol [2]	Endosulphan II [2]
9-Hexadecanoic acid [1]	Endosulphan sulphate [2]
9-Octadecenoic acid [1]	Endrin [2]
Alpha-Chlordane [2]	Ergost-5-en-3-ol [1]
Alpha-Chlorotoluene [2]	Ethylbenzene [2]
Alpha-hexachlorocyclhexane [2]	Ethyl methylbenzene [1]
Alpha-terpineol [1]	Fluorene [2]
Androsterone [1]	Gamma-Chlordane [2]
Atrazine [1]	Gamma-hexachlorocyclohexane [2,4]
Benzeneacetic acid [1]	Heptachlor [2,4]
Benzeneethanol [1]	Heptachlorodibenzodioxin [2]
Benzenemethanol [1]	Hexachlorobenzene [4]
Benzenepropionic acid [1]	Hexachlorocyclopentadiene [2]

[1] Rutherford et al. 1994
[2] OMOE 1988
[3] Golder Associates Ltd. 1995a,b
[4] Orr et al. 1992

Table 6. Organic chemicals detected in municipal wastewater treatment plant effluents. (cont'd)

Hexachloroethane [2]	o-Xylene [2]
Hexadecanoic acid [1]	Octachlorodibenzodioxin [2]
Hexadecanoic acid hexadecyl ester [1]	Octachlorodibenzofuran [2]
Hexadecanol [1]	p-Chloro-m-Cresol [2]
Hexadecenoic acid [1]	PCB (total) [2]
Hexanoic acid [1]	Pentachlorophenol [2]
Hydroxybenzoic acid [2]	Phenanthrene [2]
m- and p-Xylenes [2]	Phenol [2]
m-Cresol [2]	Phenoxyethanol [1]
Methoxychlor [2,4]	p,p'-DDD [2,4]
Methyl(methylethyl)benzenes [1]	p,p'-DDE [2]
Methyl-(methylethyl)cyclohexanol [1]	p,p'-DDT [2]
Methyl-(methylethyl)cyclohexen-1-ol [1]	Pyrene [2]
Methylbenzoic acid [1]	Silvex [2]
Methylene chloride [3]	Stigmast-5-en-3-ol [1]
Methylphenol [1]	Styrene [2]
Mirex [2]	Tetrachlorodibenzofuran [2]
N,N-bis(hydroxyethyl)dodecanamide [1]	Tetrachloroethylene [2,3]
N,N-Diethyl-3-methylbenzamide [1]	Tetradecanoic acid [1]
N,N-Diethylmethylbenzamide [1]	Tetradecanol [1]
N-Nitroso-di-N-propylamine [2]	Tetramethylbenzene [1]
Naphthalene [1,2]	Toxaphene [2]
Nicotine [1]	Trans-1,3-Dichloropropene [2]
Nitrobenzene [2]	Trichloroethylene [2]
Nonanoic acid [1]	Trimethylbenzene [1]
o-Cresol [2]	Trimethylpentanediol [1]

[1] Rutherford et al. 1994
[2] OMOE 1988
[3] Golder Associates Ltd. 1995a,b
[4] Orr et al. 1992

In addition to chemicals introduced from domestic and industrial sources, chemicals in MWWE may also be derived from the treatment process. For example, strontium, aluminum and ferric chloride are used as chemical precipitants and are consequently high in effluents receiving these types of treatment (e.g., strontium and aluminum in many Ontario effluents, iron in the Montreal effluent; Table 4). Another example is total residual chlorine (TRC) which is a measure of the amount of chlorine remaining in the final effluent after chlorination treatment for disinfection (Table 7).

Concentrations of chemicals in MWWE can differ considerably, despite similarities in treatment (Tables 3, 4 and 5) While the degree of pollutant removal often increases from primary to secondary to tertiary treatment (particularly for conventional parameters such as BOD and TSS), it is difficult to characterize the chemical content of MWWE on the basis of treatment type as concentrations depend upon many factors, including domestic *vs.* industrial sources, types of industries, surface area served, and volume treated. In addition to variability in effluent characteristics among MWTPs, there may be temporal (e.g., daily, weekly and seasonal) variability in effluent within a plant. At three Ontario MWTPs, effluent variability was associated with daily rhythms in raw sewage

characteristics: organic substances, particularly phenolics, exhibited the greatest degree of variability in effluents; metals were the next most variable and conventional parameters the least variable (OMOE 1991). Variability may also be related to seasonal use of certain treatment processes (e.g., chlorination) or to different operating conditions between winter and summer (e.g., Table 7 for winter *vs.* summer TRC concentrations for eight MWTPs in Ontario).

Table 7. Concentrations (µg/L) of total residual chlorine (TRC) and annual discharge (x 10^6 m^3) for selected municipal wastewater treatment plants that use chlorine for disinfection. (Blanks indicate data are not available.)

Location and MWTP type	**Date**	**TRC**		**Effluent Discharge**
Nova Scotia[1]				
2 secondary	fall 1991	5-850		
Eastern Passage primary	fall 1991	250		4.4 (1986)
Lakeside tertiary	fall 1991	120		0.5 (1986)
Ontario[2]		summer:	winter:	
Bracebridge secondary	1989	70	9	0.9
Huntsville secondary	1989	530	950	1.4
Walkerton secondary	1989	40	nd	1.7
Stratford secondary	1989	6	nd	8.1
Toronto North secondary	1989	310	190	12.7
Toronto Highland Creek secondary	1989	5	46	55.3
Midland secondary	1989	260	nd	4.1
Wallaceburg secondary	1989	81	190	2.2
Saskatchewan[3]				
Saskatoon primary	summer 1987	1892		32

[1] Rutherford et al. 1994, except discharge from Environment Canada 1986
[2] Orr et al. 1992 - 1989 TRC data, 1988 discharge
[3] Saskatchewan Environment and Public Safety 1989

Loadings

Comparison of effluent load (effluent concentration x effluent discharge) to the load of the receiving water provides an estimate of the long-term potential for an effluent to affect the receiving water, especially for persistent and bioaccumulative substances which can cause cumulative impacts. For conventional parameters, a 1991 study of 387 Ontario MWTPs found that the highest load was for BOD (41,000 tonnes/yr) (OMOE 1993). For metals, strontium (1.3 tonnes/d), aluminum (0.8 tonnes/d), and zinc (0.2 tonne/d) were the greatest contributors to loadings among all treatment types in a study of 37 Ontario MWTPs (OMOE 1988). Ten other metals had smaller loads (less than 0.5 tonne/d) but are more toxic (e.g., cadmium, copper, chromium, lead, nickel). Cadmium loadings from 37 Ontario MWTPs totalled less than 10 kg/d and mercury loadings totaled less than 150 g/d. Three Greater Vancouver MWTPs (Environment Canada 1992), two

Edmonton MWTPs (Golder Associates Ltd. 1995a,b) and a MWTP in the Yukon (Enns and Soprovich 1995) showed a similar pattern in metal loadings with high aluminum (up to 233 kg/d) and zinc (up to 56 kg/d) and low cadmium and mercury loadings.

Although numerous organic pollutants have been detected in MWTP effluent (OMOE 1988; Rutherford et al. 1994; Golder Associates Ltd. 1995a,b), total loadings of organic chemicals are generally lower than for metals. Total loadings averaged 132 kg/d for base/neutral and acid extractable organics, 107 kg/d for volatile organic compounds, 1.4 kg/d for pesticides, 0.082 kg/d for PCBs and 0.004 kg/day for dioxins and furans for 37 MWTP in Ontario (OMOE 1988). The average load of all 21 PAHs in Montreal MWTP effluent was 1.2 kg/d, while the load for the sum of 13 PCBs averaged 2.5 g/d (Pham and Proulx 1996). In the case of PCBs, this represented only 1% of the PCB load measured in the St. Lawrence River at Quebec City. Although the loadings of PCBs and other organic pollutants are relatively small, they are a cause for concern because they are persistent and have the potential to bioaccumulate and biomagnify in the food chain.

Stormwater, CSOs and MWTP bypasses

Characterization

Stormwater and CSOs have not been routinely monitored because the diffuse and intermittent nature of these sources makes large-scale monitoring programs prohibitively expensive. However, some information is available for Ontario because of work done in support of the Canada/US and Canada/Ontario Agreements respecting Great Lakes Water Quality and for British Columbia because of the Fraser River Estuary Management Plan. Stormwater and CSO discharges are characterized by high flows during or shortly after periods of wet weather or during periods of snowmelt, high quantities of suspended solids, and significant quantities of nutrients, toxic substances (e.g., heavy metals, chlorides, hydrocarbons) related to traffic and road maintenance, and microorganisms.

The main pollutants of concern in stormwater are suspended solids, nutrients (particularly P), heavy metals, hydrocarbons, and fecal bacteria. A recent review by Makepeace et al. (1995) of 140 studies mostly from the USA, several European countries and Canada identified the 28 most important contaminants in stormwater that have the potential to affect aquatic life and human health, mostly through contamination of drinking water supplies (total solids, TSS, Al, Be, Cd, Cl, Cr, Cu, Fe, Pb, Mn, Hg, N, Ag, Zn, DO, PCBs, bis(2-ethylhexyl) phthalate, δ-BHC, chlordane, heptachlor, heptachlor epoxide, total PAHs, benzo(a)pyrene, tetrachloroethylene, fecal coliforms, fecal streptococci, enterococci). Concentrations of water quality constituents reported in Ontario, British Columbia and Calgary, Alberta stormwater are listed in Table 8.

The chemical composition of CSOs has been studied much less than for stormwater, in part because CSOs are more difficult to monitor than stormwater. During the early phase of a runoff event, when sewage sludge may be scoured from the sewer bottom by high flows,

Table 8. Typical concentrations of various constituents found in stormwater and annual stormwater discharge.

Parameter	Ontario		Calgary, Alberta (mean[d])	Lower Mainland, British Columbia (mean[e])	
	Stormwater (mean or range)	CSO[a] (mean)	Urban Runoff	Urban Runoff	CSO
total suspended solids (mg/L)	170[a]	190	1691	48	59
biochemical oxygen demand (mg/L)	14[a]	41	17	14	60
chloride (mg/L)	230-340[b]		7		
ammonia (mg/L)	0.30-0.75[b]		0.74	0.15	3.8
total phosphorus (mg/L)	0.35[a]	1.4	1.52	0.1	1.9
total nitrogen (mg/L)	3.5[a]	83	6.83		
cadmium (µg/L)	1.5-6.8[b]			4.8	3.2
copper (µg/L)	43.4-47.2[b]		33	40	77
iron (µg/L)	5710-6960[b]		22300		
lead (µg/L)	97-233[b]		458	55	89
mercury (µg/L)	29-104[b]				
nickel (µg/L)	28-39[b]		9		
zinc (µg/L)	234-307[b]		170	98	81
oil and grease (mg/L)	2.14-5.37[b]				
PAHs (µg/L)	2.1-9.1[b]				
benzo(a)pyrene (µg/L)	0.3[b]				
fecal coliform (MPN/100 mL)	12000 - 51000[c]	1000000		2043	390000
E.coli (MPN/100 mL)	800 - 61000[c]				

MPN = Most Probable Number; numbers in brackets indicate sample size; NA = Not Available

[a] Waller and Novak 1981
[b] Marsalek and Ng 1989
[c] Marsalek et al. 1992
[d] Hamilton and North 1986
[e] Environment Canada 1992 from Greater Vancouver Regional District 1988a

CSO composition resembles or even exceeds pollutant concentrations in raw sanitary sewage. After this first flush, pollutant concentrations in CSOs subside. The main parameters of concern are suspended solids, BOD, nutrients (N and P), fecal bacteria and possibly some other chemicals originating from local municipal and industrial sources. Concentrations of water quality constituents reported in Ontario and British Columbia CSOs are listed in Table 8. In comparison to stormwater, constituent concentrations in CSOs are similar for TSS (Marsalek et al. 1993), but greater for BOD (Marsalek et al.1993), total N and total P (Marsalek et al. 1993), and generally smaller for unconventional pollutants, including heavy metals, PAHs and some other trace organic contaminants released from industrial sources (Marsalek and Ng 1989).

Loadings

Loadings from stormwater and CSOs depend on the drainage area, land-use activities in that area and, in the case of CSOs, the nature of the sewage generated in the area. One of the more extensive summaries of stormwater loadings was published for the Canadian Great Lakes basin (Marsalek and Schroeter 1988). Here, annual loading was greatest for TSS (10^8 kg/y), followed by oil and grease (10^5 - 10^6 kg/y), inorganics (mostly heavy metals, 10^2 - 10^5 kg/y), PAHs (10^0 - 10^2 kg/y), and some trace organic contaminants (10^{-1} - 10^1 kg/y).

For CSOs, conventional pollutant loadings in the Canadian Great Lakes basin were estimated to be 17,400 tonnes/y for TSS, 3,700 tonnes/y for BOD, 760 tonnes/y for total N and 130 tonnes/y for total P (Waller and Novak 1981). These estimates are consistent with more recent studies of CSOs in Sarnia and Windsor (Marsalek and Ng 1989) which gave the combined annual CSO discharge from both municipalities as 6.2 x 10^6 m^3/y and corresponding annual loads of 1200 tonnes/y TSS, 51 tonnes/y total N, and 8.7 tonnes/y total P. These loads represent about 7% of the total estimated loads of TSS, total N and total P to the Great Lakes basin, which is consistent with the fact that Sarnia and Windsor represent about 9% of the basin population assumed by Waller and Novak (1981) in their calculations.

Impacts of Municipal Wastewater Effluents

MWWEs have the potential to affect both human use of water resources and the structure and functioning of aquatic ecosystems. Impacts from the discharge of MWWE include: (a) restrictions on fish and shellfish consumption; (b) degradation of aquatic and wildlife populations and their habitat (including water and bottom sediment quality); (c) eutrophication or undesirable algal growth; (d) isolated incidences of water-borne disease caused by sewage contamination of drinking water supplies in communities dependent on high quality raw water supplies; (e) beach closures; (f) degradation of aesthetics; and, (g) added costs to agricultural, industrial and municipal users for treatment of unacceptable water.

Impacts may be acute and manifest themselves over short time-frames or cumulative (long-term) and manifest themselves only after extended periods of time (Hvitved-Jacobsen 1986; Harremoes 1988). Acute impacts are generally caused by toxic concentrations of ammonia (Lijklema et al. 1993), total residual chlorine (Orr et al. 1992) or heavy metals (Lijklema et al. 1993) in the receiving waters; BOD or COD loads that reduce dissolved oxygen to concentrations insufficient for the survival of aquatic organisms (Lijklema et al. 1993); high flows from urban runoff (Borchhardt and Statzner 1990) and; bacterial contamination that render shellfish unsuitable for human consumption (Fisheries and Oceans / Environment Canada 1992). The frequency of acute effects attributable to MWWE is determined by the nature and quantity of industrial and residential inputs, treatment type, disinfection regime and whether the MWTP capacity is surpassed. Furthermore, cumulative toxic impacts may be caused by metals and trace organic contaminants in the receiving water (Lijklema et al. 1993).

For stormwater and CSOs, the frequency of acute impacts depends upon the frequency and intensity of rain and snowmelt events causing overflows and can therefore be estimated from local climatic data and wastewater collection/treatment system capacity. Stormwater and CSO effects are most serious in small urban creeks, where the aquatic biota can be severely damaged by sediment erosion and deposition resulting from increased runoff, elevated temperatures, and toxic concentrations of chemicals, fecal bacteria and pathogens (Horner et al. 1994). In these small streams, the dilution of stormwater and CSO discharges is minimal. Acute impacts caused by stormwater and CSO discharges occur less frequently in large water bodies because of large dilution of the effluent. For example in the St. Marys River, Ontario, no increase in fecal bacterial concentration was observed downstream of Sault Ste. Marie because of stormwater dilution by the high river discharge (Dutka and Marsalek 1993). However, even in the case of CSO discharges to large rivers, dilution is not always an effective mechanism for improving water quality, particularly in the near-shore zone. Marsalek et al. (1996) observed poor microbial water quality in the Detroit River along a 22 km section in Windsor, Ontario (a city with 25 CSOs) and in the St. Clair River along a 9.5 km section in Sarnia, Ontario (with five CSOs). A study of 252 stormwater discharges in Greater Vancouver, British Columbia, identified 40 of the receiving waters as having high environmental sensitivity (based on presence of an endangered or protected habitat, spawning or rearing habitat for recreational or commercially utilized species, or use for primary contact water activities), 96 as moderately sensitive (generally characterized by moderate river flow, moderate fisheries resources, and secondary water contact such as water skiing), and 116 as low sensitivity (good river flow, low public use and minimal fisheries resource) (UMA 1994). In the Great Lakes basin, in 1994, the potential for problems from stormwater and CSO pollution was rated as “medium” to “high” for 11 of the 17 Canadian Areas of Concern (AOCs) and “very high” for Hamilton Harbour and the Toronto Waterfront AOCs (Weatherbe and Sherbin 1994).

Human Health Issues

Restrictions on recreational water uses

High loads of bacteria in MWWE, stormwater or CSOs can result in restricted recreational activities in the receiving waters (Table 9a). Many beaches in urban areas are frequently closed for several days during and immediately after rainfall events because of microbial contamination caused by stormwater and CSO discharges (Dutka and Marsalek 1993; Marsalek et al. 1994; Nix et al. 1994; Tsanis et al. 1995). For example, between 1986 to 1994, forty-four percent of Ontario's Great Lakes beaches had "closure notices", with the majority of these (79 of 369) on Lake Ontario (Edsall and Charlton 1996). A 1993 study of the Thames River near London, Ontario, likewise showed that 10 of 18 beaches exceeded a geometric mean count of 100 *E. coli* per 100 mL with one beach having counts up to 3300 per 100 mL (Burns and Reffle 1995). Similarly, on the Red River from 1980 to 1989, the Recreational Water Quality Objective for *E. coli* was exceeded 57 to 96% of the time downstream of the Winnipeg, MB secondary MWTP compared to 3.2 to 7.5% at upstream locations (Gurney 1991). However, disinfection of MWWE reduces the risk of microbial pollution. In a study of three Ontario MWTPs, the combination of secondary treatment and chlorination was found to be effective in destroying more than 99% of indicator organisms (total coliforms, fecal coliforms, fecal streptococci, total staphylococci and *Pseudomonas aeruginosa*) (Seyfried et al. 1984).

Microbial contamination resulting in restrictions of recreational water use is typically determined by exceedance of the Canadian Recreational Water Quality Guideline (Health and Welfare Canada 1992) or a more stringent provincial limit (e.g., Ontario, for *E. coli* and enterococci bacteria). *E. coli* and fecal coliform are generally used as indicators of contamination by pathogens such as hepatitis B, enteritis, cholera, and typhoid. Compliance with the Canadian Recreational Water Quality Guidelines is determined by comparing a geometric mean of at least 5 samples, taken for a period not to exceed 30 days, to indicator organism limits (Health and Welfare Canada 1992).

The potential public health risks of microbial pollution in Canadian recreational waters are not well understood as there are few epidemiological data. In an epidemiological study of swimming-related illness at 10 beaches in the Great Lakes basin, Seyfried (1985a,b) reported that 7% of swimmers became ill compared with 3% of non-swimmers. The most frequently reported illnesses were respiratory ailments followed by gastrointestinal, eye, ear, skin, and allergic symptoms. Similarly, a 1984 study of windsurfers from sewage-contaminated waters of the Baie de Beauport on the St. Lawrence River near Quebec City reported that the relative risk for occurrence of symptoms of otitis, gastroenteritis, conjunctivitis and skin infection increased with number of falls into the water (Dewailly et al. 1986). The current Canadian Guidelines for Recreational Water Quality state that the seasonal risk of gastrointestinal illness is between 1 and 2% for an *E. coli* concentration of 200 colonies/100 ml (Health and Welfare Canada 1992). The actual risk, however, depends on the nature of recreational activities - the highest being for swimmers, and the lowest for those engaged in wading.

Table 9. Human health and environmental issues associated with discharge of municipal wastewater effluent, the component(s) of the effluent causing the problem, the relevant guidelines for drinking water or surface water quality or from the Canadian Shellfish Sanitation Program, and *in situ* examples of the problem.

(a) Human health issues

Issue	Primary Causal Factor(s)	Guidelines	Examples
Restricted recreational use	bacterial contamination	< 200 *E. coli* and <35 enterococci per 100 ml as geometric means of 5 samples taken over 30 d. (plus other parameters).[1]	Beach closures around the Canadian Great Lakes (Edsall and Charlton 1996) and Thames River near London, ON (Burns and Reffle 1995) caused by exceedance of the provincial guideline of 100 *E. coli* per 100 mL; marine beach closures near Victoria, BC prior to 1993 caused by exceedance of provincial guideline of 200 fecal coliforms per 100 mL due to stormwater runoff (Taylor et al. 1995).
Fish and shell fish closure	bacterial contamination	< 14 fecal coliforms per 100mL (median or geometric mean Most Probable Number (MPN)) in water where shellfish collected and not more than 10% of samples exceeding MPN of 43/100mL for a five-tube decimal test; and comprehensive water sanitary survey.[2]	fish and shellfish restrictions for Lower Fraser River and estuary, BC (Birtwell et al. 1988); shellfish closures on Atlantic and Pacific coasts (Menon 1988; Wells and Rolston 1991); in 1992, approx. 3,018 km^2 closed to shellfish harvesting in Canada due to bacterial contamination (Environment Canada fact sheet, undated).
	marine biotoxins	< 80 μg/100g Paralytic Shellfish Poison (PSP); < 20 μg /g domoic acid (ASP) in shellfish and considerations[2]	Intermittent PSP closures are posted at various times of year near Victoria, BC (Thomson and Wilson 1995)
	metal contamination	0.01 ppm Hg in seafood[2]	1 shellfish closure reported on Atlantic due to Cd and Pb (Environment Canada fact sheet, undated)
	organic compounds	5 ppm DDT, 2 ppm PCB, 20 ppt dioxin; 0.1 ppm mirex, 0.1 ppm all other agricultural chemicals, in seafood [2]	2 shellfish closures reported on Pacific coast due to dioxin (Environment Canada fact sheet, undated)

Drinking water contamination	bacterial contamination	0 cfu/ 100 ml for total coliforms or <10cfu/100 ml for total coliforms if none are fecal coliforms.[3]	rare isolated accounts across Canada associated with the lack of, or poorly functioning, drinking water treatment facilities (e.g., O'Neil 1984; O'Neil et al. 1985)
	metal contamination	1.0 mg/L Ba; 0.005 mg/L Cd; 0.05 mg/L Cr; 1.0 mg/L Cu; 0.3 mg/L Fe; 0.01 mg/L Pb; 0.05 mg/L Mn; 0.001 mg/L Hg; 0.01 mg/L Se; 0.1 mg/L U; 5.0 mg/L Zn.[3]	d/s Ile-aux-Vaches, Que. sewer outfall (L'Italien et al. 1991)
	nitrate contamination	10 mg/L N as NO_3. [3]	nitrate contamination of well water on Ontario farms (Goss and Barry 1995). Many possible sources of contamination, including sewage.
	organic compounds	extensive list[3]	water supplies from Great Lakes (Manno et al. 1995)

(b) Environmental degradation

Issue	Primary Causal Factor(s)	Guideline for Protection of Aquatic Life[4]	Examples from *in situ* observations
Eutrophication	nutrient addition	Phosphorus and nitrogen - no numerical value; excessive amounts to be avoided	Bow River d/s Calgary, AB (Culp 1992); South Saskatchewan River d/s Saskatoon, SK (Chambers 1993; Chambers and Prepas 1994); d/s Duncan-North Cowichan MWTP, BC (Perrin et al. 1988); Georgian Bay, Lake Huron (Nicholls and Heintsch 1992; Gemza 1995; Sherman and Brown 1995); Lake Ontario (Sly 1991)
Changes to physical habitat	temperature change	Thermal additions should not alter thermal stratification or turnover dates, exceed maximum weekly average temperature and exceed maximum short-term temperature.	
	total suspended solids (TSS)	effluent TSS: ≤ 10 mg/L for receiving waters with TSS ≤ 100 mg/L, or 10% of receiving water concentration when receiving water > 100 mg/L	Extensive degradation of intertidal sand flats in outer estuary of Fraser River when Iona plant was discharging there (Birtwell et al. 1988)
	multiple causes		changes in benthic infaunal community near a deep-sea outfall in BC (Macaulay Point, Victoria) as a result of organic matter enrichment of sediments (Chapman et al. 1996); changes to fish community structure before and after MWTP upgrades near Toronto, ON (Wichert 1995)
DO stress	BOD addition	Cold water biota: 9.5 mg/L DO for early life stages 6.5 mg/L DO for other life stages Warm water biota: 6.0 mg/L DO for early life stages 5.0 mg/L DO for other life stages	daily mortality of fisheries resources (crab, flounders and salmon) when Iona plant was discharging onto intertidal flats in outer estuary of Fraser River (Birtwell et al. 1983)

(c) Direct Toxicity and Bioaccumulation

Issue	Primary Causal Factor(s)	Guideline for Protection of Aquatic Life[4]	Examples from *in situ* toxicity tests, bioaccumulation and observations
Toxicity	unionized ammonia (NH_3)	1.37 - 2.2 mg/L[a]	toxicity of MWWE from Stratford, ON, likely due to both unionized ammonia and TRC (Flood et al. 1984)
	nitrite (NO_2)	0.06 mg/L	
Toxicity	total residual chlorine (TRC)	2.0 μg/L	Toxicity to fish (Servizi and Martens 1974); MWWE from Turner Valley, AB (Osborne et al. 1981; Osborne and Davies 1987)
Toxicity/ Bioaccumulation	metals	5-100 μg/L Al[b]; 50 μg/L As[c]; 0.01-0.06 μg/L Cd[d]; 2-20 μg/L Cr; 2-4 μg/L Cu; 300 μg/L Fe; 1-7 μg/L Pb[e]; 0.1 μg/L Hg; 25-150 μg/L Ni[e]; 1 μg/L Se; 0.1 μg/L Ag; 30 μg/L Zn; 5 μg/L free cyanide	toxicity of effluent from CFB Cornwallis, NS, attributed to aluminum, copper and low pH (Rutherford et al. 1994); guidelines exceeded for some metals for the lower Fraser River and associated tributaries (Environment Canada 1992); bioconcentration of some metals in benthos and fish in the lower Fraser River (Swain 1986, Swain and Walton 1989, Harding et al. 1987 and EVS Consultants 1986 as reported in Environment Canada 1992); toxic sediments due to Hg and 1,4 - dichlorobenzene near a deep-sea outfall from Victoria, BC (Chapman et al. 1996); Hg bioaccumulated in invertebrates and fish 1-3 km below Macauley and Clover Points MWTPs in Victora, BC (Colodey et al. 1992); bioaccumulation of Cu, Pb and Zn in mussels from Halifax Inlet near MWTP (Ward 1990)
Toxicity	chlorinated solvents	110 μg/L tetrachloroethylene[f] 20 μg/L trichloroethylene[f]	
Toxicity/ Bioaccumulation	PCBs	1 ng/L PCB 10 ng/L PCB for marine waters	PCB contamination of marine biota (Birtwell et al. 1988)

Toxicity/ Bioaccumulation	PAHs	6 μg/L acenaphthene[e] 4 μg/L acridine[e] 0.01 μg/L anthracene[e] 0.04 μg/L fluoranthene[e] 3 μg/L fluorene[e] 1 μg/L naphthalene[e] 0.02 μg/L pyrene[e] 3 μg/L quinoline[e] 0.02 μg/L benzo(a)anthracene[e] 0.01 μg/L benzo(a)pyrene[e] 0.4 μg/L phenanthrene[e]	PAHs in sediment and biota near MWTP in BC (Harding et al. 1988 as reported in Wells and Rolston 1991)
Toxicity/ Bioaccumulation	chlorinated phenols	7 ug/L monochlorophenol 0.2 ug/L dichlorophenol 18 ug/L trichlorophenol 1 ug/L tetrachlorophenol 0.5 ug/L pentachlorophenol	tetra- and pentachlorophenols present in Iona (BC) effluents accumulating in exposed salmons (Birtwell et al. 1988)

[a] Guideline varies with pH and temperature (CCREM 1987)
[b] Guideline varies with pH, calcium and DOC concentrations (CCREM 1987)
[c] guideline under review (CCREM 1987)
[d] revised guideline (CCME 1996a)
[e] guideline varies with hardness (CCREM 1987)
[g] interim guideline (CCME 1996b)
[f] interim guideline (CCREM 1987)

[1] Guidelines for Canadian Recreational Water Quality (Health and Welfare Canada 1992)
[2] Canadian Shellfish Sanitation Program (Fisheries and Oceans / Environment Canada 1992)
[3] Guidelines for Canadian Drinking Water Quality (CCREM 1987)
[4] Canadian Water Quality Guidelines for Protection of Freshwater Aquatic Life (CCREM 1987)

Contamination of fish and shellfish areas

Restrictions in fish and shellfish consumption by humans are usually associated with contamination by bacteria, metals or organic compounds. While bacterial contamination is generally associated with MWWE or urban runoff, the source of metal and organic contamination to a fishery or shellfishery is difficult to ascribe due to the many sources of these substances in the environment. Compounds such as PCBs, PAHs, chlorinated phenols, chlorinated benzenes, and mercury are associated with MWTP effluents and urban runoff (Birtwell et al. 1988; Marsalek and Schroeter 1988; OMOE 1988); they are persistent in the environment and have the potential to bioaccumulate and biomagnify in aquatic organisms rendering them unsafe for human consumption. For example, in 1983 tetra- and pentachlorophenols present in the effluents from the Iona MWTP in Greater Vancouver, BC were found to be rapidly bioaccumulating in exposed juvenile chinook salmon (Birtwell et al. 1988) (Table 9a).

The relationship between shellfish contamination and MWWE is more obvious than for fish contamination (Table 9a). Bacterial contamination of shellfish harvesting areas is widespread and pervasive on both the Atlantic and Pacific coasts. This contamination has resulted in the closure of harvesting areas because of concerns about transmission of disease, ranging from mild gastroenteritis to typhoid fever and hepatitis, resulting from the consumption of molluscan shellfish (e.g., oysters, clams, and mussels) (Nelson 1994). In 1992, approximately 3,018 km^2 were closed to harvesting due to bacterial contamination in the three Canadian shellfish regions (Environment Canada fact sheet, undated). The main factor causing the deterioration in marine water quality is untreated or poorly treated domestic sewage due to neglected or outdated sewage and other wastewater treatment infrastructure (Nelson 1994).

On the Atlantic coast (excluding Quebec), 35% or 2011 km^2 of the areas surveyed as suitable for direct harvesting of shellfish were closed in 1995, with about $10-12 million lost to the local economy (M.P. Guilcher, Environment Canada, Atlantic Region, pers. com.). Municipal effluents were identified as the direct cause for 20% of all shellfish closures in the Maritimes (Menon 1988). In Quebec, of the 156 shellfish zones evaluated in 1991, 67 (43%) were permanently closed and 42 (27%) were open conditionally (i.e., closed from June 1 to September 30) (Environnement Canada 1991). Fecal contamination has also affected bivalve aquaculture production. For example, in Newfoundland, 30% of proposed aquaculture sites representing 127 km of coastline were rejected because of high coliform levels in 1989 (Government of Canada 1991).

In British Columbia, Wells and Rolston (1991) reported that municipal discharges were identified as the sole source of approximately 15% of all shellfish harvesting closures, and implicated as a contributing factor in a further 78% closures. In 1989, bacterial contamination was responsible for closures of 705 km^2 along 730 km of the BC coastline. In 1994, 26% of areas surveyed or approximately 730 km^2 were closed to shellfish harvesting of which about 550 km^2 were in the Georgia Strait. This represents

about a $4 million loss to the local economy from the intertidal fishery alone (H. Nelson, Environment Canada, Pacific and Yukon Region, pers. com.). The number of closures due solely to marine sewage outfalls has approximately doubled since 1972 (Nelson 1994).

In the Arctic, Wells and Rolston (1991) identified municipal sewage disposal as one of the concerns in the Western Arctic, Lancaster Sound and Tuktoyaktuk Harbour. They noted that a public health concern may exist for communities which consume shellfish from contaminated waters or butcher marine mammals on contaminated shorelines.

Contamination of drinking water

Since municipalities treat and disinfect water used for drinking, explosive outbreaks of water-borne disease are rare in Canada (Health Canada 1995a,b). Isolated incidences of microbial contamination of drinking water in Canada from CSOs, stormwater and inadequately treated MWWEs have been reported (Table 9a). For example, an epidemic of gastroenteritis lasting 24 to 48 hours affecting approximately 3000 persons in an Alberta city of 6500 people occurred in 1983 following discharge of raw sewage caused by a lift station failure (O'Neil et al. 1985). With increasingly sensitive analytical methods for detection of parasites and viruses, concern has arisen about the safety of water meeting current drinking water quality standards. In an epidemiological study conducted in the Montreal Urban Community, Payment et al. (1991) reported the risk of gastrointestinal illness was higher among persons drinking tap water (incidence of 0.76) originating from sewage-contaminated surface waters but which met Canadian Drinking Water Quality Guidelines than among persons drinking the same water which had been passed through a domestic reverse-osmosis filtration unit (incidence of 0.50).

Environmental Degradation

Dissolved Oxygen (DO) Stress

Discharge of MWWEs with high BOD loads into receiving waters can cause immediate reductions in DO in the water column as well as longer-term impacts (on the scale of months or years) due to build-up of oxygen-consuming material in the bottom sediments (i.e., sediment oxygen demand) (Hvitved-Jacobsen 1982). DO threats to fish and other organisms often occur during summer because the solubility of oxygen in water decreases with increasing water temperature. However, in colder climates where rivers and lakes are ice-covered for many months, DO threats can occur during winter due to ice cover preventing reaeration (Chambers et al., in press).

DO reduction can cause ecological impacts such as reduced biological diversity and loss of species (Table 9b). In July 1980, DO concentrations 1700 and 3200 m from a sewage outfall in the Fraser River estuary, BC ranged between 0.6 and 0.9 mg/L (well

below the Canadian Water Quality Guideline, Table 9b), and high mortality of Chinook salmon was observed (Birtwell et al. 1983). In the Bow River, AB prior to 1983, dense macrophyte beds were supported by the nutrient-rich effluent from Calgary MTWPs (Charlton et al. 1986). Culp et al. (1992) speculated that diel fluctuations in DO concentrations caused by macrophyte photosynthesis and respiration (4-5 mg/L DO to supersaturation) along with high ammonia concentrations may have been responsible for fish kills prior to the 1982 upgrading of the secondary treatment systems to advanced phosphorus removal.

Eutrophication of receiving waters

CSOs, stormwater and MWTP effluents contribute nutrients (N and P) to receiving water bodies and thus cause eutrophication (Table 9b). Because nutrients can accrue in the bottom sediments and be released into the water at a later time, nutrient loading has a cumulative as well as an immediate effect (Harremoes 1988). The impact of added nutrients on aquatic ecosystems is a major concern because excess nutrients can increase the growth of primary producers (algae and rooted aquatic plants) to levels that result in impairment of the ecosystem (e.g., changes in energy dynamics and food web structure, changes in habitat, loss of species). These ecological changes, in turn, can affect human use of aquatic resources including water-based recreational activities, fisheries, and water quality to municipal, industrial and agricultural users (Lijklema et al. 1993). Yet, despite the clear consequences of excessive nutrient loading, the concentrations of P or N that change a lake, river or coastal water from acceptable to unacceptable conditions are difficult to define because they depend upon the particular ecosystem and the goals of its users. N and P are required to support a diverse ecosystem and their concentrations will naturally increase over hundreds of years as an aquatic ecosystem ages. However, once a baseline concentration of nutrients is present, there is a wide range of acceptable concentrations (and consequently, acceptable abundance and composition of aquatic organisms) before excessive nutrient concentrations result in clear environmental degradation. Thus management of nutrients in the aquatic environment ranges from addition of P to nutrient-poor coastal streams in B.C., so as to enhance salmonid production, to P control in Lake Erie and Lake Ontario to reduce algal blooms and undesirable changes in fish community composition.

Eutrophication was a common problem in the early 1970s before the recognition of the critical role of nutrients in regulating the productivity and trophic status of lakes, rivers and marine waters (e.g., Schindler et al. 1971; Cole 1973; Dillon and Rigler 1974; US-EPA 1983; Peterson et al. 1985; National Research Council 1993). These studies identified P as the nutrient that is in shortest supply in most inland waters; therefore, its availability usually controls aquatic plant growth and, hence, eutrophication. However, in most marine waters, N is the nutrient which regulates aquatic plant growth. With the recognition of the role nutrients play in eutrophication, many MWTPs discharging to inland waters have upgraded to reduce P loading (particularly MWTPs around the Great Lakes, in the Prairie provinces, and in the British Columbia Okanagan basin).

Reductions in P loading from MWTPs and other sources have been successful in controlling or reversing eutrophication in many Canadian lakes. For example, in the early 1970s, spring total P concentrations in Lake Ontario reached peak values of 50 µg/L, but in response to nutrient controls under the Canada - United States Great Lakes Water Quality Agreement of 1978, concentrations decreased to about 10 µg/L in the late 1980s (Sly 1991). These changes reflect the influence of both reformulations of laundry detergents and greatly improved municipal sewage treatment, i.e., to less than 1.0 mg/L total P in plants discharging a million gallons per day or more (Sly 1991). However, eutrophication remains a problem in specific areas in the basin. For example, Severn Sound, a group of bays in southeast Georgian Bay, Lake Huron, continued to exhibit signs of eutrophication despite the construction of several sewage treatment plants since 1969. Nutrient regeneration from sediments is thought to be partly responsible for the delay in lake recovery following advanced and improved phosphorus treatment (Gemza 1995).

Eutrophication continues to be a pervasive problem for many rivers. Until recently, studies on the effects of nutrient addition to inland waters focused largely on lakes and reservoirs with comparatively little work conducted on streams and rivers. Yet, nutrient dynamics and food web responses to nutrient addition can differ markedly between running and standing water systems. For example, unlike in lakes, nutrients in rivers are believed to be repeatedly taken up by aquatic plants, released during tissue decomposition and then translocated downstream to be taken up by other plants ("nutrient spiralling") (Newbold et al. 1981; Mullholland et al.1990). The ecological importance of nutrient spiralling is evident when considering nutrient-limited streams in which periphyton biomass often exceeds that predicted from empirical P-biomass relationships.

Our poorer understanding of eutrophication in rivers has meant that MWWE management strategies aimed at reducing eutrophication in receiving waters have met with variable success. The situation in the Canadian Prairies is a good example. During the 1970's and early 1980's, increases in P concentrations and macrophyte biomass were reported for the South Saskatchewan River downstream of Saskatoon (City of Saskatoon 1986; Chambers 1993) and the Bow River downstream of Calgary (Cross et al. 1984). In the case of the Bow River, MWWE loads increased total P concentrations from <10 μg/L upstream of Calgary to, on average, 328 μg/L in winter and 132 μg/L in summer approximately 12 km downstream of the City (1979-1982, Hamilton and North 1986). In response to these higher P concentrations, periphyton biomass increased eight-fold from upstream to downstream of Calgary while rooted aquatic plant biomasses exceeded 1000 g/m^2 dry weight downstream of Calgary compared to negligible values upstream (Charlton et al. 1986). In response to these findings, many cities in the Canadian prairies implemented advanced P removal: Regina in 1976, Calgary in 1982, and Saskatoon in 1990. Yet despite the well-advanced technologies for nutrient removal, the receiving rivers have shown only slow and variable recoveries. For example, the biomass of periphyton in the Bow River had not decreased significantly by 1988 (six years after the implementation of advanced P removal) at sites downstream of Calgary while macrophyte biomass had decreased at some but not all sites (Sosiak 1990). The slow and

variable response of prairie rivers to P reductions likely relates to the fact that primary production in these systems is often dominated by rooted aquatic plants which can take up nutrients from the riverbed sediments (Chambers et al. 1989) and, thus, show a cumulative response to nutrient loading. Hence, nutrient enrichment continues to be an issue in prairie rivers as the recovery of these systems will likely depend upon reductions in sediment nutrient concentrations.

An additional complexity in evaluating the effects of nutrient addition on aquatic organisms is the potential for higher order effects related to nutrient-dissolved oxygen-contaminant interactions. Whereas nutrient addition will have a direct effect by increasing primary production, this enhanced plant growth may, in turn, accentuate diel swings in DO (due to daytime oxygen production from photosynthesis *vs.* nighttime oxygen consumption from respiration) and increase sediment oxygen demand (due to bacterial-mediated decomposition of dead plant material at the sediment-water interface). Thus, the nutrients in MWWEs could increase primary productivity and consequently food availability for secondary producers, but the same effluents may release contaminants or lower DO levels that separately or synergistically act as a stress to biota. These cumulative interactions would result in any one of a variety of endpoints, including a reduction in productivity, an increase in productivity, or a lack of change. To isolate the effects of nutrients, contaminants and DO stress on complex foodwebs, many researchers have advocated the use of experimental stream systems (e.g., Lamberti and Steinman 1993; Culp and Podemski 1996). Recent studies using these systems have isolated the effects of DO stress from those due to contaminants (Lowell and Culp 1996) and contaminant stress from nutrient effects (Culp et al. 1996).

Changes to physical habitat

Habitat structure (e.g., stream morphometry, stream discharge, water temperature, particle size structure of bottom sediments) can be modified as a result of discharges from MWTP, stormwater or CSOs. These physical changes can, in turn, cause changes in food web structure and loss of critical species (Table 9b). For example, Wichert (1995) in a study of fish associations in streams near Toronto, Ontario, observed that upgrades to MWTP outfalls promoted colonization of sensitive fish species (characterized by greater tolerance for changes to physical habitat, turbidity, chlorine and DO). A study of habitat degradation in the Great Lakes basin reported that approximately 15% of the river and lake areas had been damaged by MWWE, resulting in impaired nearshore wildlife habitats and feeding areas (World Wildlife Fund 1995).

Habitat changes can occur in any receiving water, however the most serious impacts occur in small urban creeks where repeated and sudden changes in flow, usually as a result of stormwater or CSO discharges, can cause instantaneous (i.e., flooding, washout) or long-term (i.e., changes in channel morphometry) impacts (Schueler 1987; Borchardt and Statzner 1990). CSO and stormwater discharges are known to increase turbidity of receiving waters (Villeneuve and Lavallée 1986), and stormwater discharges can cause erosion and increased turbidity and sediment transport in Canadian streams (MacRae and Marsalek 1992; Lorant 1988). Suspended solids, in turn, can cause a number of direct

and indirect environmental effects including reduced sunlight penetration and consequently reduced photosynthesis, infilling of spawning grounds, and harm to fish (Horner et al. 1994). CSO and stormwater discharges to small rivers and creeks can also result in thermal enhancement of the receiving environment, particularly during periods of low flows. This may modify the composition of algae, e.g., from cold-water diatoms to warm-water filamentous green and blue-green species, invertebrate or fish communities (Galli 1991; Horner et al. 1994).

Direct Toxicity

The toxicity of municipal effluents is dependent upon a variety of factors including the size and extent of industrial and urban development; the type and efficiency of the treatment and disinfection processes; and the physical, chemical and biological characteristics of the receiving waters. In the case of MWWE, toxicity is usually attributed to ammonia, total residual chlorine (in the case of chlorinated effluents), cyanide, sulfides, phenols, surfactants and several heavy metals (including copper, zinc, chromium and nickel). Other factors such as temperature, pH, hardness, alkalinity and DO tend to modify toxicity of chemical constituents. In addition, compounds can interact and the resulting toxicity may not reflect the toxicity of the individual compounds. Thus, because of the numerous factors involved and their interactions as well as the site-specificity of the effects in the receiving environment, it is not easy to arrive at broad generalizations regarding toxicity of MWWEs.

Laboratory toxicity tests using planktonic algae, zooplankton and/or fish have been conducted for effluents from many Canadian MWTPs to determine concentrations that are lethal (i.e., median lethal concentration, LC50) or cause physiological or behavioural changes (i.e., median effect concentration, EC50). While responses to complex effluents clearly differ among organisms, unionized ammonia is the most frequent cause of toxicity for MWWE, as well as total residual chlorine (TRC) in the case of chlorinated effluents (Table 10). Occasionally, toxicity is also attributed to high metal concentrations (e.g., Alexander et al. 1977; Rutherford et al. 1994; Wong et al. 1995) or surfactants (Alexander et al. 1977; Higgs 1977b). However, while it is sometimes possible to ascribe toxicity to a particular chemical or group of chemicals in a complex effluent, often toxicity shows no clear relationship with the concentrations of known toxicants. For example, Orr et al. (1992) in a study of acute and chronic toxicity of effluents from Ontario MWTPs (two lagoons and eight secondary plants) found that, when all data were considered, less than 50% of the observed variations in the toxicity test data could be explained by variations in chemical concentrations. However when test results were interpreted by evaluating each contaminant independently, TRC and un-ionized ammonia were the most important toxicants with respect to rainbow trout survival; elevated sulphide levels associated with lagoons were also toxic to trout; TRC was the only parameter associated with effluent toxicity to *Daphnia magna*, while un-ionized ammonia represented a major toxicant to *Ceriodaphnia dubia*; metals (e.g., copper and aluminum) may have contributed to effluent toxicity; and, with the exception of

Table 10. Summary of the results of laboratory toxicity tests measuring median lethal or median effects concentrations for effluents from MWTPs across Canada.*

Location and Treatment	Date	Toxicity Issue?	Probable Cause of Toxicity
Nova Scotia[1]			
Greenwood secondary chlorinated	fall 1991	Yes	NH_3
Lakeside tertiary chlorinated	fall 1991	No	
Cornwallis CFB secondary chlorinated	fall 1991	Yes	Al, Cu, low pH, TRC?
Cornwallis CFB secondary chlorinated	fall 1991	Yes	TRC
Eastern Passage secondary chlorinated	fall 1991	Yes	NH_3 and TRC
Ontario[2]			
Bracebridge tertiary + P removal, chlorinated	1989	Yes - summer	?
Toronto Highland Creek secondary + P removal, chlorinated	1989	Yes	NH_3
Huntsville secondary + P removal, chlorinated	1989	Yes	NH_3, TRC
Lindsay primary + P removal	1989	Yes - winter	NH_3
Midland secondary + P removal, chlorinated (seasonal)	1989	Yes	TRC
North Toronto secondary + P removal, chlorinated	1989	Yes - summer	NH_3, TRC
Perth primary + P removal	1989	Yes - winter	NH_3
Stratford tertiary + P removal, chlorinated (seasonal)	1989	Yes - summer	NH_3, TRC
Walkerton secondary + P removal, chlorinated (seasonal)	1989	No	
Wallaceburg secondary + P removal, chlorinated	1989	Yes - winter	NH_3, TRC
Manitoba			
North End Winnipeg secondary[3]	1975	Yes	NH_3, Zn, Cu, surfactants
South End Winnipeg secondary[4]	Oct77	Yes	NH_3, nitrite
Alberta			
Capital Region Edmonton secondary[5]	1989-90	No	
Capital Region Edmonton secondary[6]	1992-94	Yes	NH_3?
Goldbar Edmonton secondary[5]	1989-90	No	
Goldbar Edmonton secondary[6]	1992-94	Yes	NH_3?
British Columbia			
Macaulay Point Victoria preliminary[7]	1992	Yes	NH_3, low DO
Clover Point Victoria preliminary[7]	1992	Yes	NH_3, low DO
Iona Island Greater Vancouver primary, chlorinated[8]	Aug76	Yes	anionic surfactants, TRC
Annacis Island Greater Vancouver primary,	Oct76	Yes	NH_3, anionic surfactants
chlorinated+dechlorinated[9]	July76	Yes	NH_3, TRC, Cu?, Zn?, Cy?
Penticton secondary + P removal, chlorinated[10]	Sept76	Yes	NH_3, anionic surfactants
Cache Creek secondary, chlorinated[10]	July76	No	
Mission secondary, chlorinated[10]	Sept76	No	
Prince George secondary, chlorinated[10]	Aug76	No	
Williams Lake secondary, chlorinated[10]	Aug76	No	
Clinton secondary[10]	Apr92	Yes	NH_3, Al, Cu, Zn
Annacis Great Vancouver primary[11]			

*The test organisms ranged from planktonic algae to zooplankton to fish. The question "Is there a toxicity issue?" was answered "Yes" if LC50 or EC50 for at least one test was less than 85% (i.e., an 85:15 volume:volume mixture of effluent to water). Probable cause of toxicity is the assessment by the author(s) of each study of the compound(s) likely responsible for toxicity. (? indicates questionable.)

[1] Rutherford et al. 1994
[2] Orr et al. 1992
[3] Alexander et al. 1977
[4] Spink and Thackeray 1979
[5] Moore et al. 1993
[6] Golder Associates Ltd. 1995a,b
[7] EVS Consultants 1992
[8] Higgs 1977b
[9] Higgs 1977c
[10] Higgs 1977a
[11] Environmental Management Associates & Hydroqual Laboratories Ltd. 1993

methoxychlor, most of the organic contaminants detected in the effluent samples were at levels below those expected to be toxic to the test organisms.

Laboratory toxicity tests offer the advantages of controlled conditions for comparing toxicity among effluents. However, they provide little ecological insight as to the effects of an effluent on organisms in the receiving water. The most obvious environmental property that affects toxicity *in situ* is dilution capacity. Although an undiluted effluent may be acutely lethal in laboratory tests, receiving systems with large dilution capacity may dilute effluents to non-lethal levels. However, the dilution capacity of a receiving water may also vary with time. For example, tides can reverse the river flow and cause pooling of effluents near Annacis and Lulu MWTPs in the lower Fraser River, BC, thereby subjecting fish to multiple dosing of effluents (Churchland et al. 1982; Birtwell et al. 1988). In addition to dilution, factors such as hardness, pH, temperature, and organic matter in the receiving environment can affect the bioavailability and, thus, the toxicity of organic and inorganic chemicals (e.g., Erickson 1985; Mayer and Ellersieck 1988; Gobas and Zhang 1994; Mayer et al. 1994). Moreover, the potential impact of contaminants on organisms in the receiving water may be exacerbated or mitigated by other stressors, such as low DO or very low or very high nutrient concentrations (e.g., Taylor et al. 1991; Munawar et al. 1993). Thus, while determination of the causal factor(s) of toxicity *in situ* may be difficult, assessments conducted in the receiving water offer the advantage of identifying responses under natural conditions of caged test species (i.e., *in situ* toxicity tests) or native communities (i.e., *in situ* monitoring and observation).

In situ toxicity tests and observations of community diversity and abundance have identified biological responses to MWWE discharge downstream of many Canadian cities (Table 9c). Some of these responses can be ascribed to specific chemicals in the effluents, primarily ammonia, TRC and metals. As with MWWE, the toxicity of stormwater and CSOs is often attributed to ammonia but also toxic metals (copper, lead, iron and zinc), hydrocarbons (particularly PAHs), and pesticides (Hall and Anderson 1988; Dutka et al. 1994a,b). However, stormwater and CSO toxicity has not been studied as extensively as MWWE toxicity because of the highly variable nature of their flows and chemical concentrations, the resulting variability in flows and concentrations in the receiving waters, and the fact that traditional 96-h bioassays are not applicable to stormwater and CSO discharges which are of much shorter duration. Alternative approaches for assessing stormwater and CSO toxicity have been used, e.g., breathing responses of caged fish (Seager and Abrahams 1990; Ellis et al. 1995), mortality of invertebrates (Hall and Anderson 1988), and battery of bioassays on several media (Dutka et al. 1994a,b). However, many of these tests are difficult and expensive to conduct and hence have received only limited consideration.

In addition to toxicity associated with surface waters, bottom sediments located near municipal outfalls may have concentrations that render them toxic to aquatic organisms. For example, mercury and 1,4-dichlorobenzene were found in sediments at concentrations of possible concern, up to 100 m away from the deep sea outfall from the Macauley Point MWTP in Victoria, BC (Chapman et al. 1996). While sediment toxicity tests showed no

effect on survival, growth and development was reduced for laboratory organisms exposed to sediments from that area. In the same study area, the benthic infaunal community structure within 100 m of the outfall was also found to show a classic organic enrichment pattern of increasing species richness and decreased abundance with distance from the outfall. Toxicity was also observed for sediments collected at the mouth of the Nashwaak River 800 to 1000 m downstream of the Fredericton, NB secondary MWTP, and from Halifax Harbor 30 to 200 m from the Eastern Passage, NS primary MWTP outfall (Rutherford et al. 1995). Comparison of sediment metal concentrations with Canadian Sediment Quality Guidelines (Environment Canada 1995) also indicated that concentrations of cadmium, chromium, copper, lead, mercury, nickel and zinc in sediments transported by runoff from 12 urban areas in southern Ontario (Marsalek and Schroeter 1988), collected from downstream of CSO discharges in Greater Vancouver, BC, and collected near the Macauley Point MWTP outfall, Victoria, BC exceeded interim trace metal guidelines for the protection of aquatic life (J. Ellis, Greater Vancouver Regional District, Burnaby, BC, pers. com.) (Table 11). Furthermore, many of the concentrations also exceeded the concentrations above which adverse biological effects are expected to occur (probable effect level, PEL).

One of the major issues with toxic chemicals is providing protection of both human and environmental health from the chronic effects of long-term low-level loading of persistent and bioaccumulative toxic chemicals. Historical management practices have assumed that water protected from chemical contaminants through guidelines and regulations assures chemical, physical and biological integrity. However, within a watershed, the potential degradation begins in the headwaters and accumulates with progressive downstream developments, and also includes far-reaching effects such as atmospheric transport of contaminants. This can result in a build-up of contaminants in downstream receiving waters, i.e., cumulative effects on a spatial scale. In addition cumulative environmental impacts can also occur on temporal (seasonal, among years, and decadal time frames) and organizational (cellular, individual, population or community levels) scales. The assessment of the cumulative effects of these chemicals on aquatic biota is poorly understood and is complicated by the same range of point and non-point sources of pollution and the complexity and dynamic nature of the ecological pathways of the receiving waters. Yet, it is clear that with continued population growth and industrial expansion, there is a need to better understand the additive and synergistic effects of multiple environmental stressors acting at different temporal and spatial scales.

An emerging issue: presence of endocrine disrupters

3.2.3.1.3 Endocrine Disruptors

Environmental contaminants can affect the reproduction of fish and wildlife through a wide variety of mechanisms. Recently, concern has arisen about chemicals that can bind to the estrogen receptor, thereby regulating the activity of estrogen-responsive genes. Estrogens play a critical role in controlling reproductive processes in fish. One of the functions of natural estrogens in fish is to stimulate the liver to produce vitellogenin, a large phospholipoprotein, which is released into the blood stream and sequestered by developing oocytes for production of egg yolk. In maturing female fish, vitellogenin is a

Table 11. Concentrations of trace metals in sediments located near municipal sources and Canadian interim freshwater Sediment Quality Guidelines (SQGs) for the protection of aquatic life and Probable Effect Levels (PELs) for eight trace metals. (Blanks indicate data are not available.)

Trace Metal	SQG[1] (mg/kg)	PEL[1] (mg/kg)	Sediment Concentration (mg/kg)				
			12 southern Ontario urban areas[2]	Below CSOs in the Greater Vancouver Regional District[3]	Near the Macauley Point outfall, Victoria, BC[4]		
					Reference Site	0-400 m downstream	800-1600m downstream
As	5.9	17.0	8.2	3.6	4.9 to 5.0	6.0-12	5.2-5.7
Cd	0.6	3.53	2.0	<1.2	< 0.1	<0.1-0.7	< 0.1
Cr	37.3	90.0	110.0	44	36-39	32-59	38-39
Cu	35.7	196.6	67.0	196	13-15	19-197	17-19
Pb	35.0	91.3	470.0	127	7.0-8.0	9.4-129	8.2-11
Hg	0.174	0.486	0.24	1.21	< 0.05	0.03-0.98	<0.05-0.3
Ni	18.0	35.9	50.0	30	--	--	
Zn	123.1	314.8	400.0	176	61-66	68-198	69-73

[1]Environment Canada 1995
[2]Marsalek and Schroeter 1988
[3]J.Ellis, pers.com., Greater Vancouver Regional District, Burnaby, B.C.
[4]EVS Consultants 1992

major constituent of the blood proteins whereas in male fish it is not normally present in any appreciable amount. However, if male fish are exposed to estrogens, vitellogenin can be produced at similar levels to that found in maturing females. Although the implications of this induction on reproductive function are not fully understood, the presence of vitellogenin has been used as a sensitive indicator of exposure of fish to exogenous estrogens.

Sewage effluents appear to have the potential to cause endocrine disruption in fish. Studies in the early 1980's in the River Lea, UK, reported a low incidence of intersex in the roach (*Rutilus rutilus*) exposed to sewage effluents. Follow-up studies have demonstrated that the effluents were strongly estrogenic with many of the exposures resulting in plasma vitellogenin levels in fish being elevated to the levels expected in gravid females (Purdom et al. 1994). This response has been observed in fish collected immediately downstream of many sewage treatment plants in the UK and the response persists in some cases several kilometers downstream of the outfalls (Harries et al. 1997). This response has also recently been reported in wild fish at North American sites (Folmar et al. 1996).

There are many compounds which are capable of binding to estrogen receptor and causing estrogenic responses, e.g., alkylphenol polyethoxylates, bisphenolics, phthalates, *p,p'*-DDE (White et al. 1994; Soto et al. 1995; Jobling et al. 1996; Routledge and Sumpter 1996). Municipal effluents are complex mixtures which contain a variety of contaminants which may be capable of endocrine disruption. There is also growing evidence that non-ionic surfactants in detergents can act as estrogen mimics (Jobling and Sumpter 1993). Measurable quantities of nonylphenol ethoxylates and their degradation products have been found in a variety of MWWEs and sludge in southern Ontario, Quebec and Atlantic Canada, and in surface waters in the Great Lakes Basin (Bennie et al. 1996, 1997; Lee et al. 1997). Exposure of rainbow trout to several alkylphenolics resulted in the synthesis of vitellogenin and inhibition of testicular growth (Jobling et al. 1996). Nonylphenol has recently been shown to cause intersex in Medaka (Gray and Metcalfe 1997). Alkylphenol polyethoxylates and several degradation products which are estrogenic are expected to partition to sewage sludge and aquatic sediments and be persistent in the environment (White et al. 1994; Jobling and Sumpter 1993). Although elevated vitellogenin production may be caused by several estrogenic compounds in effluent, the majority of the response in a sewage effluent has recently been isolated into a single chromatographic fraction (Routledge et al. 1995). This suggests that the response is largely due to a single chemical or group of very similar chemicals, at least for the effluent studied. Following a toxicity identification evaluation approach, Desbrow et al. (1996), were able to isolate and identify natural and synthetic estrogens in the bioactive fractions (e.g., 17α-ethynylestradiol, 17β-estradiol and estrone). Although these chemicals are found at very low concentrations, they were at levels (low ng/L) which would explain the vitellogenin induction observed in fish (Desbrow et al. 1996). Schweinfurth et al. (1996) reported similar low levels of 17α-ethynylestradiol in surface waters. Although this work strongly suggests that natural and synthetic estrogens are likely responsible for the estrogenic responses observed near sewage outfalls, the levels of nonylphenols in municipal effluents which receive textile or other industrial effluent inputs may be high enough to explain the responses at these sites. Blackburn and Waldock (1995) measured concentrations of nonylphenol (24-53 μg/L) downstream of a wool scouring plant, discharging through a municipal treatment plant, which were sufficiently high to explain the observed effects (Harries et al. 1997).

In Canada, the extent of estrogenic effects attributable to sewage effluents has not been established. Although some chemical characterization of effluents has recently been undertaken (Bennie et al. 1996, 1997; Lee et al. 1997), the causal relationship between the chemical constituents, such as nonylphenols and synthetic estrogens, and possible estrogenic effects has not been established. There are potentially several responsible chemicals and their toxicity and bioavailability are dependent on the characteristics of the effluent and the receiving environment. A comprehensive toxicity (estrogenicity) analysis of MWWE coupled with a detailed survey of the physiological status of fish from receiving waters is required to fully evaluate the extent and potential impact of endocrine disrupting compounds in the Canadian environment.

Conclusion

Discharge of municipal effluents (sewage, stormwater and CSO) continues to have significant adverse impacts on Canadian receiving waters, despite the relative high proportion of the population (81%) served by sewage treatment facilities. The environmental consequences of municipal wastewater discharge to the Canadian environment are, however, difficult to

generalize largely due to regional variations in level of sewage treatment and the nature of the receiving waters (river, lake or coastal water). The most publicly recognized direct impacts of sewage discharge are shellfish harvesting restrictions and restrictions on recreational water uses such as beach closures resulting from microbial contamination (Table 9a). Contamination of shellfish harvesting beds is widespread and pervasive on both the Atlantic and Pacific coasts. Closures of shellfish beds on both coasts have increased in recent years, with significant economic loss.

In many parts of Canada, municipal sewage discharge has led to habitat degradation and contamination which, in turn, have altered the abundance and diversity of aquatic organisms (Table 9b). Destruction of fish and wildlife habitats has been documented as a result of physical alterations to the environment, particularly as a result of stormwater or CSO discharge. MWWEs are also significant sources of toxic substances, some of which are persistent and bioaccumulative (Table 6). While reports of acute toxicity *in situ* are now virtually non-existent for Canadian receiving waters, the chronic effects to aquatic organisms of continuous long-term loading of low doses of persistent and bioaccumulative pollutants are little known and may pose a substantial threat to aquatic habitat and communities. In addition, little is known of the cumulative environmental impacts occurring at a variety of interacting spatial (i.e., geographic) and temporal (years or decade) scales, and of the additive or synergistic effects resulting from the complexity of chemicals in MWWE. Substances capable of disrupting fish reproduction (i.e., endocrine disrupting substances) have also been detected in MWWE; however the scant information on their occurrence and responses to exposure make it difficult to assess the risk of endocrine disrupting substances in MWWE at this time. Other potential issues for which there is little Canadian information are the concentrations of medicinal drugs that enter surface waters from household wastewater and their human health and environmental consequences (Franke et al. [1995] and Wilken et al. [1997] provide European data), and the ecological significance of toxic marine dinoflagellate, such as the recently discovered *Pfiesteria piscicida*, which caused major fish kills in shallow nutrient-rich estuaries along the eastern United States (Lewitus et al. 1995, Noga et al. 1996).

Our review of MWWE in Canada indicates that gross environmental degradation associated with sewage discharge is rare. However, deleterious environmental and human health effects do occur downstream of specific urban communities that have minimal or no treatment of large quantities of sewage or stormwater, receiving waters with low flushing rates, or both. In addition, problems still exist in servicing widely-dispersed communities with low population, for example in rural and northern areas. Little or no information is available on responses to long-term low level chemical exposure or habitat degradation, or cumulative environmental impacts.

At the international level, there is a recognition that marine pollution is significantly affected by riverine sources of pollution, including municipal wastewater discharges. UNEP has launched the “Global Plan of Action for the Protection of the Marine Environment from Land-Based Activities”. This initiative involves a Global Programme of Action which was endorsed by governments in 1995 (UNEP 1995). Recent United Nations conferences (United Nations 1992a, UNEP1995) identified actions, policies and measures to maintain or restore inland and coastal waters, and protect human health from the consequences of sewage discharge. These include:

- adopting primary, secondary and, where appropriate and feasible, tertiary treatment for municipal sewage discharged to rivers, estuaries and oceans;

- establishing regular monitoring of water-related pollution impacts, including epidemiological surveillance;

- establishing waste treatment and disposal quality criteria, objectives and standards based on the nature and assimilative capacity of the receiving water;

- locating sewage outfalls so as to obtain or maintain agreed environmental quality criteria so as to avoid exposing shellfisheries, water intakes and bathing areas to pathogens, and sensitive environments to excess nutrient loads;

- eliminating the discharge of persistent bioaccumulative compounds and reducing the discharge of other synthetic organic compounds; and

- moving toward a precautionary and anticipatory approach rather than a reactive approach to the management of coastal and inland waters.

Adoption of these actions would entail new integrated initiatives at local, national and, in the case of coastal waters, global levels to manage MWWE and assess environmental risks.

Our findings on the effects of municipal wastewater discharge in Canada, along with actions recently proposed by the United Nations (1992a, 1995), suggest that there is a need to review sewage treatment requirements in Canada. These requirements would need to recognize local receiving water considerations, including water chemistry, biodiversity and human water resource uses. Our assessment of the impacts of municipal wastewater discharge also emphasizes the need for research on the interactive and cumulative responses to long-term chemical exposure and habitat degradation. Finally, our review reinforces the need for an integrated approach to wastewater management that addresses loadings from treatment plants, stormwater and CSOs as well as other wastewater sources. In areas with advanced sewage treatment, management of stormwater and control or treatment of CSOs is also advanced, recognizing that without an integrated approach to controlling both point and nonpoint sources, further improvements to receiving water quality are not attainable.

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