RWAB Meeting November 8, 2017
Agenda Item # 8.1.1 - Discussion re water quality monitoring review

Questions for RWAB Consideration

1. Is HRM's current beach monitoring program adequate?

2. What changes would you make to improve the current beach monitoring program?

3. Should HRM link the beaches monitoring program with a larger corporate monitoring program, and if so, for what purposes and how?

References:

1. HRM 2017 Beach Monitoring Protocol
2. HRM 2017 Beach Closure Summary
3. Guidelines for Canadian Recreational Water Quality, 3rd Edition, 2012.

Halifax Beach Water Quality Monitoring Protocol
Summer 2017

Beach Management

The Municipality's supervised outdoor swim (beaches) program is offered as a public service during the summer months of every year, from July 1 through August 31. This service, offered at 19 locations throughout HRM in 2017 (see Appendix A) is highly valued by our residents, and is one of the signature recreational services available during summer months. Public services offered in natural environments can only be offered when an adequate measure of public safety can be assured. Both freshwater and marine aquatic environments pose potential threats to human health, due to the possibility of contact with various chemicals or biological materials, and physical hazards.

The primary hazard posed by water quality is the potential for contact with microorganisms associated with fecal contamination. The best way to manage this risk is through the effective operation of a water quality monitoring program, including the use of risk awareness measures, appropriate guidelines and standards for collection, handling, analysis, and reporting.

In 2017, the water quality monitoring program will also cover four beaches that were formerly supervised. The affected beaches are: Black Rock, Dingle, Government's Wharf, and Kinsman.

Beach Operation

Supervised beaches are to be open to the public except in the following circumstances:

- The geometric mean (hereafter, "geomean") of five test results for a given beach is above the limit for the appropriate indicator bacteria (i.e., based on measured bacterial counts)
- Beach personnel suspect water quality concerns (precautionary – all sites)
- Notification of wastewater treatment infrastructure overflows (precautionary – for Halifax Harbour sites only).

Beach staff responses to circumstances triggering beach closures are described later in this protocol.

The opening and closure of beaches to the public is driven by water quality test results. Halifax personnel are responsible for the collection, handling and delivery of the samples to the analytical laboratory and associated documentation, and the lab is responsible for confirming documentation and analytical procedures, conducting analytical procedures, and reporting analytical results to Halifax staff. Halifax and lab staff responsibilities are described separately below.

Water Quality - Halifax Front End: Sample Collection, Handling, Delivery & Documentation

Sample collection is the process of obtaining an uncompromised sample of water from within a supervised beach area. Samples are best collected from the position in the water nearest the greatest concentration of bathers. The specific location, and the depth at that location, varies from beach to beach. At designated sampling locations, the open bottle should be submerged below the water surface approximately 30 cm, with the open end facing downwards until the

bottle has reached 30 cm (1 foot) below the surface. The most important consideration in sample collection is to avoid contaminating the sample. Human skin naturally harbours several varieties of microorganisms, including bacteria, even when freshly washed. If a hand touches the inside of the bottle or the inside of the lid, these bacteria could be transferred to the water sample, and may cause false test results, which could result in unnecessary beach closures, further testing requirements and unnecessary expense.

Beach program supervisors have the option to use telescoping sampling poles to enable them to collect samples from the appropriate locations within the beach while remaining dry on shore – but these poles should only be used if the supervisors can reliably obtain full samples in a single immersion while orienting bottles properly throughout the procedure.

Halifax strives to meet the intent of the Canadian Recreational Water Quality Guidelines (Health Canada, 2012), and this protocol has been developed in consultation with the Nova Scotia Department of Health and Wellness (NSDHW) and Nova Scotia Environment (NSE).

Beaches will be sampled on a weekly basis. Five samples will be collected during each monitoring event, one at each of five stations (Station A, Station B, etc.). A geometric mean will be calculated from the results from each station. For most beaches, where only one parameter is tested, one sample will be collected from each station.

Halifax has beaches in both freshwater and saltwater (marine) environments. Monitoring protocols are identical in these environments, except for two factors, as depicted in Table 1:

Table 1. General Beach Monitoring Indicators and Maximum Concentrations

Beach Type	**Indicator Organism**	**Maximum Allowable Concentration**
Freshwater	E. Coli	200
Marine	Enterococci	35

Additional monitoring requirements are in effect for the Municipality's official Blue Flag beach (Birch Cove Beach). In addition to standard E. Coli testing, two enterococci samples will be collected at Stations A and E once each week. A summary of all monitoring requirements at all beaches is provided in Appendix B.

Beach Supervisors receive hands-on training on proper sample collection procedures from HRM's water quality coordinator at the start of each beach season. Only Beach Supervisors should collect water samples from supervised beaches. In extenuating circumstances, they may delegate collection to other staff, if delegates are informed of proper collection procedures and care is taken to ensure they are followed.

Handling procedures for water samples are intended to ensure safe, secure, and controlled collection of samples from the time they are collected until delivered to the lab. They include proper bottle labelling and storage – including refrigeration. Bottles must be labelled with the following information:

- Date and Time of sample collection
- Sample ID: Beach Name, Station Name
- Sample Type: FW (Fresh Water) **or** SW (Salt Water)
- Analysis: E. Coli (FW) **or** Enterococci (SW)

The lab prefers water samples to be 10°C or cooler, so freshly collected samples should be immediately placed into a cooler upon return to the vehicle. If electronic coolers are unavailable or non-functional, Beach Supervisors should immediately make alternate arrangements, such as the use of standard 'picnic' coolers and crushed ice to cool the samples during transport.

The same type of bottle is used for both E. coli and Enterococci samples. This bottle contains a substance, Sodium Thiosulphate ($Na_2S_2O_3$) in crystalline (powder) form. This substance is immaterial to our concerns – it is used as a dechlorination agent for treated drinking water, and consequently does not matter if the powder escapes the bottle upon sample collection. This material will not contact HRM personnel under normal handling procedures. If the material does get onto someone's skin, thorough washing with soap and water is recommended. The Material Safety Data Sheet (MSDS) for Sodium Thiosulphate is given as Appendix C.

Documentation of water samples is critical because incomplete, inaccurate or false paperwork can lead to confusing, misleading, or useless sample results. In some circumstances, however unlikely, these could result in the swimming public being exposed to unsafe waters under municipal supervision – a condition that Halifax strives to avoid.

The primary documentation to be made for water samples is on the Chain of Custody (COC) form supplied by the lab, customized for the Halifax Beaches program – see Appendix D. This form requires the following information:

- Field Sample Identification (i.e., Beach name & Station #)
- Matrix (i.e., Fresh Water or Salt Water)
- Date & Time of sample collection
- # and Type of Bottles (per site) – you will need to report each sampling station on its own line (row) *****
- (Identification of Analysis Required) - check either E. coli, Enterococci, or both
- Name of (HRM staff delivering samples), plus date and time of delivery

*** Seven samples will be collected at Birch Cove Beach, HRM's only designated Blue Flag beach. Two samples will be collected at each of stations A and E (one E. Coli, one Enterococci), and one E. coli sample at each of stations B through D. Mark the number of bottles per station accordingly.**

Water samples are only valid for bacteriological analysis within 24 hours of sample collection, so Halifax staff must ensure the delivery of all samples to the lab on the same day they are collected. If this is not possible, samples will be stored in a refrigerated environment, preferably a dedicated refrigerator, or in a cool location with ice changes as needed. Samples must be delivered to the lab as soon as possible the following morning, BEFORE 24 hours has elapsed from the earliest sample collection event the previous day. Samples delivered more than 24

hours after collection are not reliable indicators of water quality, cannot be used to determine the safety of the beach and should be repeated.

Water Quality – Laboratory Back End: Documentation Confirmation, Analysis & Reporting

Halifax contracts with only accredited and certified laboratories for the testing of water samples through the Beaches program. These labs apply thorough quality control and assurance programs at all stages of their work, which begins with sample reception. Reception staff are responsible for confirming that the number and type of bottles received match those reported on the COC form, and for following up on any inconsistencies, errors, or uncertainties with the client. Normally this would be the Halifax Beach Supervisor who is delivering and signing for the samples at the time of sample submission, but it may also or instead be the primary client contact (i.e., Cameron Deacoff).

Upon satisfactory receipt and confirmation of all samples, the lab conducts the appropriate analysis as requested on the COC. Maxxam Analytics, which performs all HRM's laboratory analysis for this program, conducts all analyses on-site.

As indicated in the previous section, samples remain viable for analysis only when delivered to the laboratory within 24 hours of sample collection. It is therefore critical to accurately observe and record the collection time, and to deliver samples less than 24 hours later. Table 2, below, identifies the key drop times and corresponding reporting periods for both parameters on a weekly basis.

Table 2. Sample Drop-off Times for Maxxam Labs

Drop-Off Day	**Parameter**	**Preferred Drop Time (same day)**	**Latest Acceptable Drop Time**	**Results Availability Time & Day**
Monday – Thursday	E. coli	4pm @ Bedford; 2pm Burnside	10am (day after sample collected) at Bedford	Noon on Day 2
	Enterococci	4pm @ Bedford; 2pm Burnside	10am (day after sample collected) at Bedford	Noon on Day 3
Friday	E. coli & Enterococci	n/a	3pm (same day) – only at Bedford	Noon on following Monday
The tests for E. coli and Enterococci must run for 24 hours and 48 hours, respectively. If samples are received in the afternoon the results will not be ready by noon the following day; they will be reported by noon on the second day. (i.e., if sample dropped off Monday at 3pm the results will be reported on Wednesday by noon).				
Notice for sample drop-off beyond regular cut-off times should be given as early as possible. During business hours, please contact				

Maxxam sends analytical results to the following personnel via email: Cameron Deacoff, Josh Weagle & Rhonda Dea.

Water Quality Results

Halifax uses the best available scientific guidance, in consultation with NSDHW and NSE, to determine the bacteria levels at which swimming and other primary contact recreation is safe. This guidance comes from Health Canada, which publishes Guidelines for Canadian Recreational Water Quality. The current edition of these guidelines was published in 2012.

E. coli is recommended as the best indicator of fecal contamination in fresh water, and as a suitable indicator in marine waters. Enterococci is recommended as the best indicator of fecal contamination in salt water, and as a suitable indicator in fresh waters.

In concordance with the Guidelines, Halifax uses E. coli as indicator of water quality for freshwater beaches, and Enterococci as the indicator of water quality for marine and brackish beaches. Kinap Beach is the Municipality's only brackish beach this year and is exclusively managed via Enterococci results. All other beaches are exclusively managed based on E. coli results.

HRM Response to Water Quality Results

Lab results are received by Halifax Coordinators or Supervisors with the Beach program.

For supervised beaches, where bacteriological results exceed guideline limits, Beach Supervisors should arrange to retest the affected beach as soon as possible and follow the steps outlined in Table 3.

Table 3. Action Items for Excess Indicator Bacteria Levels at Supervised Beaches

Step #	Action	Person(s) Responsible
1	Notify HRM Coordinator of Aquatic Services	Aquatics Specialist or Beach Supervisor on Office duty
2	Notify staff on affected beach location	Beach Staff
3	Place appropriate signage at site	Lifeguard(s) on site
4	Notify HRM Public Affairs Office: Email:	Beach Staff
5	Remain on station for at least 7 days for Public Relations	Lifeguard(s) on site
6	Direct all media questions to the Coordinator Aquatic Services or designate. Staff to maintain "no comment" unless otherwise directed.	All staff
7	Notify NSE (Sara Rumbolt or alternate)	Water Quality Coordinator

As mentioned previously, the Municipality will conduct water quality monitoring at the following four beaches that were previously supervised: Black Rock, Dingle, Government's Wharf, and Kinsman. For these unsupervised beaches, the Municipality will respond by following steps outlined in Table 4.

Table 4. Action Items for Excess Indicator Bacteria Levels at Unsupervised Beaches

Step #	Action	Person(s) Responsible
1	Notify Coordinator of HRM Aquatic Services	Aquatics Specialist or Beach Supervisor on Office duty
2	Place appropriate signage at site	Beach Supervisor
3	Notify HRM Public Affairs Office to publish PSA: Email:hrmcommunicationoffice@halifax.ca	Beach Staff
4	Direct all media questions to the Coordinator Aquatic Services or designate.	All staff
5	Notify NSE (Sara Rumbolt or alternate)	Water Quality Coordinator

In the event of sewage overflows at Halifax Water Pumping Stations or other locations, Halifax Water has developed a response procedure for Dingle Beach and Black Rock Beach. See Appendix E for this procedure.

Beach Retesting in Case of Closure

When water sample results lead to closures of any municipal beach, supervised or unsupervised, affected beaches are re-sampled on soon as practical, typically the following weekday. Beaches will remain closed until the geomean of five samples is equal to or less than the guideline limits (200 E. coli, 35 Enterococci).

When wastewater (sewage) system overflows are the cause of beach closures, water samples (E. Coli and Enterococci) should be collected on the next weekday after the overflow. Beaches will remain closed until the geomean of five samples is equal to or less than guideline limits.

During retesting conditions, Beach Supervisors should consider documenting the following conditions to assist in interpreting water results as necessary:

- Was it raining at the time of collection or at any time during the previous 24-hour period?
- How clear or turbid was the water?
- Were ducks or other waterfowl present? How many?
- Did you see any other possible signs of water contamination, or possible causes?

Halifax Beach Program Contacts:

Water Quality Coordinator	Cameron Deacoff HRM Energy & Environment – Planning & Development
Coordinator Aquatic Services	Rhonda Dea HRM Beaches – Parks, Recreation and Communities
Aquatic Supervisor	Josh Weagle HRM Beaches – Parks, Recreation and Communities
Beach Office	(Rotating position – Beach Supervisors) HRM Beaches – Parks and Recreation 902.490.5458
Beach Supervisor	Jacob Hamilton HRM Beaches – Parks, Recreation and Communities
Beach Supervisor	Emily Dunn HRM Beaches – Parks, Recreation and Communities
Beach Supervisor	Lewis Jenkins HRM Beaches – Parks, Recreation and Communities
Beach Supervisor	Jean-Luc Lemieux HRM Beaches – Parks, Recreation and Communities

List of Appendices:

Appendix A.

Municipal Beach Locations 2017

Area	Beach Name	Civic Address	Community	Associated Lake/Watercourse & Park (if any)
Supervised Beaches				
Halifax	Campbell Point Beach	187 Lakewood Dr.	Brookside	Hatchet Lake
	Chocolate Beach	2 Melwood Ave.	Halifax	Chocolate Lake
	Cunard Beach	121 Williams Lake Rd.	Halifax	Cunard Pond
	Kearney Beach	Unaddressed - Hamshaw Dr.	Halifax	Kearney Lake
	Kidston Beach	94 Fieldstone St.	Halifax	Kidston Lake
	Long Pond Beach	869 Herring Cove Rd	Herring Cove	Long Pond
Dartmouth	Albro Beach	199 Albro Lake Rd.	Dartmouth	Albro Lake
	Birch Cove Beach	46 Oakdale Cres.	Dartmouth	Lake Banook (Birch Cove Park)
	Penhorn Beach	70 Penhorn Dr.	Dartmouth	Penhorn Lake
	Shubie Beach	30 John Brenton Dr.	Dartmouth	Lake Charles (Shubie Park)
Bedford/Sackville	Sandy Beach	115 Smiths Rd.	Bedford	Sandy Lake
	Saunders Beach	105 Millrun Crescent	Bedford	Paper Mill Lake (Scott Saunders Memorial Park)
	Springfield Beach	Lakeview Ave.	Sackville	Springfield Lake
Eastern Shore	Kinap Beach*	181 Greenough Dr.	West Porters Lake	Porters Lake
	Lake Echo Beach	3170 Highway 7	Lake Echo	Lake Echo
	Mallay Falls*	Lochaber Mines Rd., off Highway 374	Sheet Harbour	East River Sheet Harbour
	Pleasant Drive Beach	183 Pleasant Dr.	Gaetz Brook	Petpeswick Lake
	Webber's Beach	738 Upper Lakeville Rd.	West Petpeswick	Lake Charlotte
Oakfield	Oakfield Beach	366 Oakfield Park Road	Grand Lake	Grand Lake +B3:E25 (Oakfield Provincial Park)
Unsupervised Beaches				
Halifax	Black Rock Beach	5718 Point Pleasant Dr.	Halifax	Halifax Harbour (Point Pleasant Park)
	Dingle Beach	Dingle Rd.	Halifax	Northwest Arm, Halifax Harbour (Fleming Park)
Eastern Shore	Government Wharf	169 West Petpeswick Rd.	West Petpeswick	Musquodoboit Harbour
Bedford/Sackville	Kinsmen Beach	31 First Lake Dr.	Sackville	First Lake

* weekday supervision only

Appendix B.

Beach Monitoring Summary 2017

Blue Flag Beach

Beach	Water Type	Sample Type	# of Stations	Sample Collection	# Bottles
Birch Cove Beach	Fresh (Lake)	E. coli & Enterococci	5	**Stn A+E:** E. coli & Enterococci; **Stns B-D:** E. coli	7

Other Beaches

Beach	Water Type	Sample Type	# of Stations*	# Bottles / Station	# Bottles
Albro Beach	Fresh (Lake)	E. coli	5	1	5
Campbell Point Beach	Fresh (Lake)	E. coli	5	1	5
Cunard Beach	Fresh (Lake)	E. coli	5	1	5
Kearney Beach	Fresh (Lake)	E. coli	5	1	5
Kidston Beach	Fresh (Lake)	E. coli	5	1	5
Kinap Beach	Marine (ocean)	Enterococci	5	1	5
Lake Echo Beach	Fresh (Lake)	E. coli	5	1	5
Long Pond Beach	Fresh (Lake)	E. coli	5	1	5
Mallay Falls	Fresh (Lake)	E. coli	5	1	5
Oakfield Beach	Fresh (Lake)	E. coli	5	1	5
Penhorn Beach	Fresh (Lake)	E. coli	5	1	5
Pleasant Drive Beach	Fresh (Lake)	E. coli	5	1	5
Sandy Beach	Fresh (Lake)	E. coli	5	1	5
Saunders Beach	Fresh (Lake)	E. coli	5	1	5
Shubie Beach	Fresh (Lake)	E. coli	5	1	5
Springfield Beach	Fresh (Lake)	E. coli	5	1	5
Webber's Beach	Fresh (Lake)	E. coli	5	1	5

Version Date: July 18, 2017

Appendix C.

No Acid

Anachemia

255 Norman.
Lachine (Montreal), Que
H8R 1A3

Material Safety Data Sheet

EMERGENCY NUMBERS:

(USA) CHEMTREC : 1(800) 424-9300 (24hrs)
(CAN) CANUTEC : 1(613) 996-6666 (24hrs)
(USA) Anachemia : 1(518) 297-4444
(CAN) Anachemia : 1(514) 489-5711

WHMIS	Protective Clothing	TDG Road/Rail
WHMIS CLASS: D-2B		Not controlled under TDG (Canada). PIN: Not applicable. PG: Not applicable.
		

Section I. Product Identification and Uses

Product name	**SODIUM THIOSULFATE, ANHYDROUS**
Chemical formula	Na2S2O3
Synonyms	Sodium hyposulfite, Sodium thiosulfate, AC-8547, 85786
Supplier	Anachemia Canada. 255 Norman. Lachine (Montreal), Que H8R 1A3
Material uses	For laboratory use only.

CI#	Not available.
CAS#	7772-98-7
Code	AC-8547
Formula weight	158.11
Supersedes	

Section II. Ingredients

Name	CAS #	%	TLV
1) SODIUM THIOSULFATE	7772-98-7	100	Not established by ACGIH: ACGIH (Sulfur dioxide) TWA 2 ppm (5.2 mg(SO2)/m3); STEL 5 ppm (13 mg(SO2)/m3)

Toxicity values of the hazardous ingredients

SODIUM THIOSULFATE:
INTRAPERITONEAL (LD50): Acute: 5200 mg/kg (Mouse).

Section III. Physical Data

Physical state and appearance / Odor	Clear to white granules or crystals. Odorless.
pH (1% soln/water)	8.6
Odor threshold	Not available.
Percent volatile	0% at 21°C
Freezing point	Transition at 48°C
Boiling point	Decomposes at >100°C.
Specific gravity	1.66-1.73 (Water = 1)
Vapor density	Not applicable.
Vapor pressure	Not applicable.
Water/oil dist. coeff.	Not available.
Evaporation rate	Not applicable.
Solubility	33% (in H2O)

Section IV. Fire and Explosion Data

Flash point	Not available.
Flammable limits	Not available.
Auto-ignition temperature	Not available.
Fire degradation products	Oxides of sulfur and sodium. Hydrogen sulfide. Sodium sulfide.
Fire extinguishing procedures	Use DRY chemical, carbon dioxide, foam or water spray. Wear adequate personal protection to prevent contact with material or its combustion products. Self contained breathing apparatus with a full facepiece operated in a pressure demand or other positive pressure mode. Disperse vapors with water spray if they have not ignited. Cool containing vessels with flooding quantities of water until well after fire is out.
Fire and Explosion Hazards	The sensitivity to impact is not applicable. The sensitivity to static discharge is not applicable. Heating above 100°C yields a flammable residue sodium sulfide. Contact with oxidizers may cause fire and/or explosion. Emits toxic fumes under fire conditions.

Section V. Toxicological Properties

Routes of entry	Inhalation and ingestion. Eye contact. Skin contact.
Effects of Acute Exposure	May be harmful by ingestion, inhalation, or skin absorption. Irritant.
Eye	May irritate or burn eyes and cause temporary conjunctivitis.
Skin	May cause skin irritation. Aqueous solutions or dust may cause irritation from repeated or prolonged contact.
Inhalation	Dust or mist may cause severe irritation to the respiratory tract. Exposure may cause coughing, chest pains, and difficulty in breathing. If heated to the point where sulfur dioxide gas is driven off, then this gas is highly irritating to the respiratory tract.
Ingestion	May cause gastrointestinal irritation. May cause nausea, vomiting, purging, cyanosis. Doses of 8 g/kg (oral, rat) were non-toxic.

Section V. Toxicological Properties

Effects of Chronic Overexposure	Carcinogenic effects: Not available. Mutagenic effects: Not available. Teratogenic effects: Not available. Toxicity of the product to the reproductive system: Not available. To the best of our knowledge, the chemical, physical, and toxicity of this substance has not been fully investigated.

Section VI. First Aid Measures

Eye contact	Immediately flush eyes with copious quantities of water for at least 15 minutes holding lids apart to ensure flushing of the entire surface. Seek immediate medical attention.
Skin contact	Immediately flush skin with plenty of water and soap for at least 15 minutes while removing contaminated clothing and shoes. Call a physician. Wash contaminated clothing before reusing.
Inhalation	Remove patient to fresh air. Administer approved oxygen supply if breathing is difficult. Administer artificial respiration or CPR if breathing has ceased. Seek immediate medical attention.
Ingestion	If conscious, wash out mouth with water. Have conscious person drink several glasses of water to dilute. Seek immediate medical attention. Never give anything by mouth to an unconscious or convulsing person.

Section VII. Reactivity Data

Stability	Stable. Conditions to avoid: High temperatures, sparks, open flames and all other sources of ignition, contamination.
Hazardous decomp. products	Not available.
Incompatibility	Oxidizing agents (e.g., nitrates, sodium nitrite, halogens) cause vigorous exothermic reactions. Acids release sulfur dioxide gas. Water-reactive materials such as sodium, cause strong exothermic reaction. Mercury salts, lead, silver, iodides, iodine, mercury.
Reaction Products	Sulfur dioxide gas which is toxic, corrosive, and an oxidizer, is driven off above 100°C leaving, a sodium sulfide residue which is flammable, a strong irritant to skin and tissue and is also incompatible with acids. Hazardous polymerization will not occur.

Section VIII. Preventive Measures

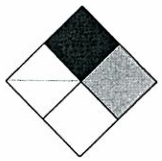
Protective Clothing in case of spill and leak	Wear respirator, chemical safety goggles, rubber boots and heavy rubber gloves.
Spill and leak	Evacuate the area. Sweep up and place in container for disposal. Avoid raising dust. Ventilate area and wash spill site after material pick up is complete. DO NOT empty into drains. DO NOT touch damaged container or spilled material. Avoid run off.
Waste disposal	According to all applicable regulations.
Storage and Handling	Store in a cool place away from heated areas, sparks, and flame. Store in a well ventilated area. Store away from incompatible materials. Do not add any other material to the container. Do not wash down the drain. Do not breathe dust. Keep away from direct sunlight or strong incandescent light. Keep container tightly closed and dry. Manipulate in a well ventilated area or under an adequate fume hood. Avoid raising dust. Handle and open container with care. Minimize dust generation and exposure - use dust mask or appropriate protection. This product must be manipulated by qualified personnel. Do not get in eyes, on skin, or on clothing. Wash well after use. In accordance with good storage and handling practices. Do not allow smoking and food consumption while handling.

Section IX. Protective Measures

Protective clothing	Splash goggles. Impervious gloves, apron, coveralls, and/or other resistant protective clothing. Sufficient to protect skin. If use conditions generate dusts, wear a NIOSH-approved respirator appropriate for those emission levels. Appropriate respirators may be a full facepiece or a half mask air-purifying cartridge respirator with particulate filters, a self-contained breathing apparatus in the pressure demand mode, or a supplied-air respirator. Do not wear contact lenses. Make eye bath and emergency shower available. Ensure that eyewash station and safety shower is proximal to the work-station location.
Engineering controls	Local mechanical exhaust ventilation capable of minimizing dust emissions at the point of use. Do not use in unventilated spaces.

Section X. Other Information

Special Precautions or comments Irritant! Do not breathe dust. Avoid all contact with the product. Avoid prolonged or repeated exposure. Manipulate in a well ventilated area or under an adequate fume hood. Keep away from heat, sparks and flame. Handle and open container with care. Container should be opened only by a technically qualified person.
RTECS NO: XN6472000.



NFPA

Prepared by MSDS Department/Département de F.S.. | **Validated 28-Jul-1999**

Telephone# (514) 489-5711

Appendix D.

Maxxam Analytics
200 Bluewater Road, Suite 105, Bedford, Nova Scotia B4B 1G9 Tel: 902-420-0203 Fax: 902-420-8612 Toll Free: 1-800-565-7227
www.maxxam.ca
E-mail: Clientservicesbedford@maxxam.ca

MAXXAM Chain of Custody Record

COC#: Page of

This column for lab use only :

Client Code

Maxxam Job#

Cooler ID	Seal Present	Seal Intact	Temp 1	Temp 2	Temp 3	Average Temp

Integrity YES NO

Integrity/Checklist by:

Labelled by:

Location/Bin#

INVOICE INFORMATION:

Company Name: Halifax Regional Municipality

Contact Name: Cameron Deacoff

Address: PO Box 1749, Halifax, NS B3J 3A5

Email: deacofc@halifax.ca

Ph: **Fax:**

REPORT INFORMATION (if differs from invoice):

Company Name: same

Contact Name:

Address:

Email:

Ph: **Fax:**

PO# 2070744653

Project # / Phase#

ProjectName/Site Location

Quote

Site #

Task Order#

Sampled by

TURNAROUND TIME

Standard ☒

10 day ☐

If RUSH Specify Date:

Pre-schedule rush work

Charge for # Jars used but not submitted

Guideline Requirements/ Detection Limits/ Special Instructions

Send results by email only.
CC results to Josh Weagle & Rhonda Dea

Field Sample Identification	Fresh (FW) or Marine (SW)	Date/Time Sampled	# & type of bottles	E. Coli	Enterococci
1 Birch Cove - Station A	FW	July 1, 0930	2	x	x
2 Birch Cove - Station B	FW	July 1, 0930	1	x	
3 Birch Cove - Station C	FW	July 1, 0930	1	x	
4 Birch Cove - Station D	FW	July 1, 0930	1	x	
5 Birch Cove - Station E	FW	July 1, 0930	2	x	
6 Penhorn - Station A	FW	July 1, 1045	1	x	
7 Penhorn - Station B	FW	July 1, 1000	1	x	
8 Penhorn - Station C	FW	July 1, 1000	1	x	
9 Penhorn - Station D	FW	July 1, 1200	1	x	
10 Penhorn - Station E	FW	July 1, 1200	1	x	

RELINQUISHED BY: (Signature/Print) **Date** **Time**

RECEIVED BY: (Signature/Print) **Date** **Time**

White : Maxxam Yellow : Mail Pink : Client

Appendix E.

PROCEDURE FOR RESPONDING TO SEWAGE OVERFLOWS - Where Dingle or Black Rock Beaches may be affected.

Pumping Stations of Concern:

Dingle Tower PS
Chain Rock PS
Armdale PS
Pier A PS

Other Overflow points:

Armdale Roundabout
Jubilee Road
Coburg Road
Fairfield Holding Tank

PROCEDURE:

1. **Overflow event occurs, as detected by Halifax Water (HW) Wastewater Collection Services who notify HW Regulatory Services:**

   Tony Blouin, Manager, Regulatory Compliance

   OR

   Kenda MacKenzie, P.Eng., Director Regulatory Services

2. **HW Regulatory Services begins notification, to NSE, and, if the overflow/malfunction lasts for 1 hour or greater, to HRM by telephone & email as noted below:**

   Weekdays, call **Josh Weagle, Aquatics Supervisor**

   And

   Cameron Deacoff, Environmental Performance Officer

   Weekends, call the **Beach Line** 902.490-5458

   Recreation Supervisors check messages first thing in the morning and in the afternoon.

3. HRM Aquatics staff will initiate the process of posting the beaches.

Cameron or Josh will notify Corporate Communications of the overflow. Corporate Communications will post a public service announcement (PSA) to advise the media. The PSA will list a representative from HRM Beaches and James Campbell, or designate from HW, as media contacts.

HW Communications
James Campbell

4. HRM Energy & Environment (Cameron Deacoff) will advise NSE that the beach has been posted and provide any available information on sampling.

Sara Rumbolt
Environmental Health Consultant

Bill Rideout (Sara's alternate)

5. <u>HRM Aquatics will resample beach for bacteria indicators by the end of the day following notification.</u>

The beach will remain closed until subsequent water test results indicate that Enterococci levels have fallen to within acceptable limits.

Municipal Sampling Procedure in Event of a CSO/Malfunction/Shutdown Event

Black Rock (Point Pleasant Park) and Dingle (Fleming Park) Beaches

In the event of an overflow notification from HW Regulatory Services, HRM Parks & Recreation will close the corresponding beach(es)

When an overflow/discharge notification is received by HRM Recreation from Halifax Water (HW) Wastewater Collection Services, the role of HRM Recreation is to conduct daily sampling at the closed beach(es) for fecal enterococci in accordance with field monitoring protocols established by HRM Energy & Environment. Field sampling is through grab samples taken at approximately 30cm below the water surface, in water that is approximately 1.0 to 1.5 metres deep. Water samples from this position can best be collected with a sampling pole from shore. When a sampling pole is unavailable, samplers may wade to a water depth of approximately 1 meter. Regardless of method, samplers are to collect 1 sample bottles at each of two stations per beach. Sampling will commence on the day of notification if time permits, or on the day following notification if not. .

Samples are to be delivered to the Maxxam Lab in Halifax, which provides 2-day turnaround for enterococci results.

Daily sampling will continue until both samples of enterococci falls below the swimming guideline level (35cfu/100mL). HRM Energy & Environment and HRM Recreation staff receive simultaneous lab result notifications from Maxxam Analytics, and both will independently confirm that bacteria levels are safe and the beach(es) may re-open.

The HRM Aquatics Supervisor will advise the lifeguards that the beaches have reopened, update the HRM website, update beach signage, and, through HRM Corporate Communications, update the media regarding beach reopening.

HRM Energy & Environment will notify HW & NSE that beaches are re-opening.

Closure #	Beach Name (Watercourse Name)	Date Closed[1]	Date Re-Opened[2]	Closure Duration (days)[3]	Closure reason	Indicator Used[4]
1	Kinsmen (First Lake)	June 29	n/a	62	Bacteria Results	E. coli
2	Kinap (Porters Lake)	June 30	July 17	18	Bacteria Results	Enterococci
3	Birch Cove (Lake Banook)	July 5 (1145am)	July 7 (1100h)	2	Bacteria Results	E. coli
4	Lake Echo (Lake Echo)	July 7 (2pm)	July 17 (2pm)	10	Bacteria Results	E. coli
5	Birch Cove (Lake Banook)	July 18 (415pm)	July 26 (12:00h)	8	Bacteria Results	E. coli
6	Government's Wharf (Petpeswick Inlet)	July 21 (11am)	August 22 (1200)	32	Bacteria Results	Enterococci
7	Kinap (Porters Lake)	August 1 (11 am)	August 22 (1200)	22	Bacteria Results	Enterococci
8	Black Rock (Halifax Harbour)	August 1 (11 am)	August 17 (11am)	16	Bacteria Results	Enterococci
9	Dingle (Halifax Harbour)	August 1 (11 am)	August 4 (1215pm)	3	Bacteria Results	Enterococci
10	Birch Cove (Lake Banook)	August 2 (2pm)	August 14 (10am)	13	Bacteria Results	E. coli
11	Albro Lake	August 2 (2pm)	August 4 (1115am)	2	Bacteria Results	E. coli
12	Albro Lake	August 11 (11am)	August 15 (3pm)	4	Bacteria Results	E. coli
13	Birch Cove (Lake Banook)	August 15 (4pm)	August 17 (5pm)	2	Bacteria Results	E. coli
14	Oakfield Beach (Grand Lake)	August 17 (1:30pm)	August 22 (1200)	6	Bacteria Results	E. coli
15	Springfield Beach (Springfield Lake)	August 17 (10am)	August 22 (1200)	6	Bacteria Results	E. coli
16	Birch Cove (Lake Banook)	August 23 (5pm)	August 29 (9am)	5	Bacteria Results	E. coli
17	Dingle (Halifax Harbour)	August 25 (1230pm)	n/a	7	Bacteria Results	Enterococci
18	Kinap (Porters Lake)	August 28 (130pm)	n/a	4	Bacteria Results	Enterococci
19	Birch Cove (Lake Banook)	August 29 (12 noon)	n/a	3	Algae presence	visual assessment

Last Updated: August 31, 2017

Notes.

1. The date on which the beach was closed by lifeguards. This date may precede the publication of Public Service Announcements of beach closures on the Halifax website (https://www.halifax.ca/home/news).

2. The date on which the beach reopened by lifeguards. This date may precede the publication of Public Service Announcements.

3. Refers to the total number of days on which a beach was closed, including partial days where beaches re-opened the day after they were closed. Tends to overstate the actual closure duration by up to one full day.

4. E. Coli is used for freshwater lakes and Enterococci for ocean / estuarine bodies (ocean, harbour).

5. Five tests are taken per beach at one time; these results are presented as pairs. Beaches are closed when results exceed the maximum acceptable concentrations of 200 (E. Coli) and 35 (Enterococci).

Closure #	Indicator Concentration(s)[5]	Notes
1	591; 1287; 219; 390; 287; 2060; 585; 239; 296	Beach unsupervised in 2017
2	51; 102; 112; 33	
3	440; 29	
4	302; 30	
5	433; 272; 31	
6	42; 241; 67; 23	Beach unsupervised in 2017
7	46; 270; 42; 153; 53; 126	
8	93; 108; 139; 19	Rainfall on day preceding initial sample collection (?)
9	71; 26	Rainfall on day preceding initial sample collection (?)
10	777; 305; 85; 181	
11	269; 77;	
12	206; 11	
13	697; 64	
14	489; 41	
15	205; 13	
16	528; 82	
17	151	Did not reopen. Season ended August 31.
18	71; 67	Did not reopen. Season ended August 31.
19		Did not reopen. Season ended August 31.

Last Updated

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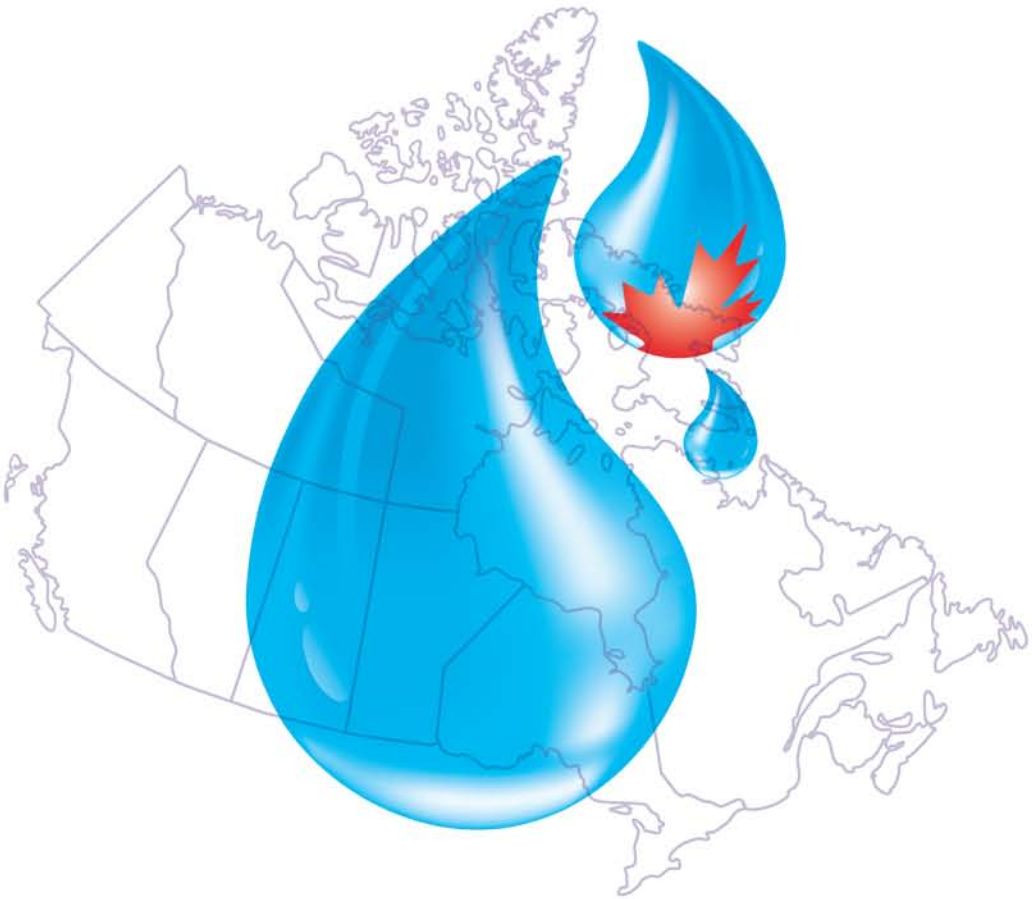
Guidelines for Canadian Recreational Water Quality

Third Edition



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Guidelines for Canadian Recreational Water Quality

Third Edition

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on Recreational Water Quality
of the
Federal-Provincial-Territorial Committee on
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Executive Summary

The primary goal of the *Guidelines for Canadian Recreational Water Quality* is the protection of public health and safety. This document provides guidance on the factors that can interfere with the safety of recreational waters from a human health perspective. It is intended to guide decisions by provincial and local authorities that are responsible for the management of recreational waters.

Recreational water quality generally falls under provincial and territorial jurisdiction. Responsibility for the safe management of recreational waters can be shared between the provincial-territorial authorities and the beach managers or service providers. The division of duties (e.g. responsibility for monitoring or the communication of results) may vary depending on provincial-territorial policies in place. Recreational waters are considered to be any natural fresh, marine or estuarine bodies of water that are used for recreation. These include lakes and rivers, as well as human-made constructions (e.g. quarries, artificial lakes) that are filled with untreated natural waters. The principal health risk associated with exposure to recreational water quality hazards is infection as a result of contact with pathogenic microorganisms. Other risks include injury or illness due to the physical or chemical properties of the water.

The *Guidelines for Canadian Recreational Water Quality* consider the human health risks associated with recreational activities—primary contact activities, such as swimming (this includes bathing/wading for the purposes of this document), windsurfing and waterskiing, as well as secondary contact activities, such as canoeing or fishing—in natural waters through intentional or incidental immersion. They establish guideline values for specific parameters used to monitor recreational water quality, including the bacteriological indicators of faecal contamination, cyanobacteria and their toxins, and values for physical and aesthetic objectives. This document also outlines a risk management approach to safe recreational water quality and describes the current scientific knowledge regarding the water quality hazards that can be encountered in the natural recreational water environment. It discusses pathogenic microorganisms of concern, water sampling and analysis, as well as emerging issues, such as faecal contamination of beach sand and faecal pollution source tracking.

Management of recreational waters

The protection and safe management of recreational waters require the cooperation of all stakeholders. The best approach is based on a preventive risk management strategy that focuses on the identification and control of water quality hazards and their associated risks before users could be exposed. As with drinking water, the multi-barrier approach provides this preventive strategy through an integrated system of procedures, actions and tools that collectively reduce the risk of human exposure to recreational water quality hazards. The effectiveness of these procedures, actions and tools is then verified or confirmed by monitoring results and the application of guideline values. The success of this approach rests primarily with the establishment of multiple barriers to protect watersheds.

Potential water quality hazards or risk scenarios that can affect the recreational water area need to be identified through an Environmental Health and Safety Survey. The results of this survey are then used to identify the appropriate procedures or actions that should be put in place as

barriers. These may include physical actions, such as beach cleanup and grooming, or processes or tools to improve the effectiveness of the recreational water management program, such as monitoring, guidelines and standards, and education and communication strategies.

Guideline values and technical information

Guideline values for a variety of water quality parameters are one important component of the overall risk management approach to safe recreational water quality. They should be used together with the appropriate technical documentation provided for these parameters. Table 1 outlines the guideline values for the recommended water quality parameters. Considerations are also provided for parameters and water quality hazards for which guideline values cannot be established.

Table 1. Guidelines for Canadian recreational water quality: summary table

Guidelines		
Parameter	Considerations	Guideline
Escherichia coli (Primary-Contact Recreation)[*]	Geometric mean concentration (minimum 5 samples) Single sample maximum concentration	≤ 200 *E. coli* /100 mL ≤ 400 *E. coli* /100 mL
Enterococci (Primary-Contact Recreation)[*]	Geometric mean concentration (minimum 5 samples) Single sample maximum concentration	≤ 35 Enterococci /100 mL ≤ 70 Enterococci /100 mL
Pathogenic Microorganisms (bacteria, viruses, protozoa)	Testing only needed when there is epidemiological or other evidence to suggest that this is necessary	No numerical guideline value
Cyanobacteria Cyanobacterial toxins	Total Cyanobacteria Total Microcystins	≤ 100,000 cells/mL ≤ 20 µg/L
Other Biological Hazards (e.g. schistosomes causing swimmer's itch; aquatic vascular plants and algae)	Recreational activities should not be pursued in waters where the responsible authority deems the presence of these organisms poses a risk to the health and safety of the users	No numerical guideline value
pH	For waters used for primary contact recreation	5.0 to 9.0
Temperature	Should not cause an appreciable increase or decrease in the deep body temperature of swimmers	No numerical guideline value
Chemical Hazards	Risks associated with specific chemical hazards will be dependent on the particular circumstances of the area and should be assessed on a case-by-case basis.	No numerical guideline value
Aesthetic Objectives		
Parameter	Considerations	Aesthetic Objective
Turbidity	To satisfy most recreational uses	50 NTU
Clarity	Clarity should be sufficient for users to estimate depth and to see subsurface hazards	Secchi Disc visible at a depth of 1.2 m
Colour	Colour should not be so intense as to impede visibility in areas used for swimming	No numerical value

[*] Advice regarding waters intended for secondary-contact recreational activities is provided in Section 4.2.

Oil and Grease	Should not be present in concentrations that can be detected as a visible film, sheen, discolouration or odour; or that can form deposits on shorelines or bottom sediments that are detectable by sight or odour	No numerical value
Litter	Areas should be free from floating debris as well as materials that will settle to form objectionable deposits	No numerical value

Indicators of faecal contamination—primary contact recreation

Escherichia coli is the most appropriate indicator of faecal contamination in fresh recreational waters, and enterococci is the most appropriate indicator of faecal contamination in marine recreational waters. Guideline values for *E. coli* and enterococci have been developed based on the analysis of epidemiological evidence relating concentrations of these organisms to the incidence of swimming-associated gastrointestinal illness observed among swimmers. The values represent risk management decisions based on the assessment of possible health risks for the recreational water user and the recognition of the significant benefits that recreational water activities provide in terms of health and enjoyment. The guidelines advised that recreational water areas routinely used for primary contact recreation be monitored at a minimum of once per week, with increased monitoring recommended for those beaches that are highly frequented or are known to experience high user densities. Similarly, under certain scenarios, a reduction in the recommended sampling frequency may be justified.

Advice regarding water intended for secondary-contact recreational activities

Due to increased interest from jurisdictions in distinguishing between primary contact activities and secondary contact activities, this current edition of the Guidelines takes an initial step at providing advice for secondary contact activities and faecal indicator concentration. There are insufficient epidemiological data available to derive precise health-based faecal indicator limit values intended to protect users engaged in secondary contact recreational activities from exposure to faecal contamination. Secondary contact is defined as recreational activity in which only the limbs are regularly wetted and in which greater contact (including swallowing water) is unusual. Because a lower degree of water exposure can be expected at most times during the majority of secondary contact recreational activities, there may be some waters in which a secondary contact use designation with separate water quality values is desired and considered acceptable to management and regulatory authorities. Advice is provided that the application of a factor of 5 to the existing geometric mean faecal indicator concentration used to protect primary contact recreation users may be used as an approach to establish faecal indicator limits. These values represent a risk management decision based on the assessment of the expected exposure scenarios and potential health risks for the recreational water user, and represents a tolerable and reasonable approach to protecting users engaged in a voluntary activity.

Other potential indicator organisms

The organisms most widely discussed as potential recreational water indicators include *Bacteroides* spp., *Clostridium perfringens*, F^+ RNA coliphages and bacteriophages infecting *Bacteroides fragilis*. At present, none of these organisms meets a sufficient number of the requirements necessary to be successfully used as a routine indicator of recreational water

quality. These organisms appear to be better suited as possible pathogen indicators or as faecal source indicators. Advances in detection and enumeration methods may improve the understanding of these organisms and the roles they may play in future recreational water monitoring programs.

Pathogenic microorganisms (bacteria, viruses, protozoa)
The challenges associated with the detection of pathogenic microorganisms in recreational waters are currently too great to recommend this practice as part of a regular monitoring program. Surveillance is necessary only during special circumstances, such as during waterborne disease outbreak investigations. Faecal indicators such as *E. coli* and enterococci are the best available indicators for the possible presence of enteric pathogenic microorganisms. However, the absence of the recommended faecal indicators should not be interpreted to mean that all pathogenic microorganisms are also absent.

Cyanobacteria and their toxins
Swimmer illnesses have been reported following exposure to toxic cyanobacterial blooms in recreational waters. Guideline values for cyanobacteria and their toxins (microcystins) have been established to protect against both the risk of exposure to microcystins as well as any harmful effects that may be possible as a result of exposure to high densities of cyanobacterial material. Waters shown to exceed the established guideline values or those in which a bloom has developed may result in human exposure to cyanobacterial material or cyanotoxins in amounts sufficient to be harmful to human health. A swimming advisory may be issued at the discretion of the responsible authority. Contact with waters where an advisory has been issued should be avoided until the advisory has been rescinded.

Other biological hazards
Recreational water activities should not be pursued in areas where other biological hazards are present in significant quantities such that they pose a risk to the health and safety of recreational water users. Examples include the presence of organisms responsible for swimmer's itch and dense growths of aquatic plants.

Physical, aesthetic and chemical characteristics
Physical, aesthetic and chemical characteristics of water can have an impact on recreational water users. Recreational waters should be of good aesthetic quality and should be free from substances that impair its aesthetic appreciation. Aesthetic components can also impact the health and safety of recreational water users where visibility has become significantly impaired.

Guideline values for specific chemical parameters in recreational waters cannot be specified. In general, potential risks from exposure to chemical parameters will site specific and be much smaller than microbiological risks. It is important for beach operators or service providers to have a mechanism in place to ensure that risks from potential chemical hazards are identified and adequate action is taken.

Faecal contamination of beach sand
Beach sand can present an important non-point source of faecal contamination to recreational waters. Sand may provide a favourable environment for microorganisms of faecal origin,

permitting them to survive for longer periods than in the adjacent waters. Physical factors such as wave action, storm surges, tidal activity and high swimmer load can result in the transference of faecal microorganisms from foreshore and nearshore sand and sediments to waters used for swimming.

Further research is needed to determine the relationships between faecal indicator bacteria and the possible presence of faecal pathogens in beach sand, as well as the potential implications for human health. Barriers that collectively reduce the risk of exposure for beach users could include public education campaigns, improved beach sanitation practices, appropriate sand grooming practices and actions designed to discourage the activities of animals (birds and other wildlife) within the beach area.

Faecal pollution source tracking

Faecal pollution source tracking is an emerging field that focuses on understanding the specific sources of faecal contamination affecting an area. Numerous chemical and microbiological source-tracking tools have been described. A good understanding and formulation of the nature of the faecal contamination problem are required before any faecal source tracking study can be considered.

Introduction

Recreational water quality generally falls under provincial and territorial jurisdiction. Responsibility for the safe management of recreational waters can be shared between the provincial or territorial authorities and the beach managers or service providers. The division of duties (e.g. responsibility for monitoring or the communication of results) will vary depending on provincial or territorial policies in place. The Federal-Provincial-Territorial Working Group on Recreational Water Quality was established by the Federal-Provincial-Territorial Committee on Health and the Environment to review and evaluate current scientific information on recreational water quality and develop up-to-date guidance. This has resulted in the development of an updated version or Third Edition of the *Guidelines for Canadian Recreational Water Quality*, which incorporates current science and outlines a recommended risk management approach. In preparing this document, the Working Group re-evaluated the criteria for existing indicators of recreational water quality and conducted reviews of the literature published on the topic of recreational water quality and human health and safety, including research papers, reports of epidemiological investigations, published texts, disease surveillance reports and guideline documentation developed by other government and multinational organizations worldwide.

The primary goal of the *Guidelines for Canadian Recreational Water Quality* is the protection of public health and safety. The document is aimed primarily at responsible authorities and decision-makers. It provides guidance on the factors that can interfere with the safety of recreational waters from a human health perspective. It recommends the adoption of a preventive risk management strategy that focuses on the identification and control of water quality hazards prior to the point of contact with the recreational water user. It also recommends the use of a multi-barrier approach as the most effective means for protecting users from exposure to water quality hazards in recreational waters.

Recreational waters can be considered as any natural fresh, marine or estuarine bodies of water where a significant number of people use the water for recreation. These include human-made constructions using untreated natural waters (e.g. artificial lakes, quarries).

Recreational water activity can be classified as any activity involving intentional or incidental immersion in natural waters. These are further defined (adapted from WHO, 2003a) as follows:

- *Primary contact:* Activities in which the whole body or the face and trunk are frequently immersed or the face is frequently wetted by spray, and where it is likely that some water will be swallowed (e.g., swimming, surfing, waterskiing, whitewater canoeing/rafting/kayaking, windsurfing, subsurface diving).

.

- *Secondary contact:* Activities in which only the limbs are regularly wetted and in which greater contact (including swallowing water) is unusual (e.g., rowing, sailing, canoe touring, fishing).

This document does not include treated recreational water facilities (e.g., swimming pools, hot tubs, whirlpool baths, hydrotherapy pools) or tertiary contact uses of water, where no contact

with water is expected (e.g., walking along the shore, sunbathing). It does not address water-related injuries such as drowning or diving injuries. It does not address issues specific to particularly sensitive individuals or population groups. Individuals concerned about their health status or the health of vulnerable population groups should consult with their health care provider or regional health unit in order to make an informed decision before engaging in any recreational water activities.

The document is divided into two parts:

- Part I (Management of Recreational Waters) provides guidance on the management of recreational waters, including approaches for water quality hazard assessment, water quality monitoring and the implementation of preventive or corrective actions.

- Part II (Guideline Technical Documentation) establishes guideline values and aesthetic objectives and provides related technical and scientific information on the water quality parameters and hazards of importance for Canadian recreational waters.

The guideline values and aesthetic objectives established in this document should not be regarded as legally enforceable standards, except where adopted by the appropriate provincial/territorial or federal agency. Further, the jurisdictional authority may wish to apply more stringent values and objectives as deemed necessary. This document is intended to guide authorities responsible for developing operational standards as part of a comprehensive beach management plan. The *Guidelines for Canadian Recreational Water Quality* may be periodically revised or adjusted as necessary to ensure that they continue to remain protective of the health and safety of all Canadians.

Part I: Management of Recreational Waters

The division of duties pertaining to the safe management of recreational waters will differ depending on the provincial or territorial policies in place. The authority overseeing the day-to-day operations of the recreational water area generally possesses the most comprehensive knowledge of the area and is therefore in the best position to take the actions necessary to ensure the safe operation of the facilities. The management information in this section is more pertinent to managed beaches (either public or private); however the same principles can be applied to any natural recreational water area.

Effective recreational water management requires the cooperation of all of its stakeholders, including beach operators and service providers, governments, local businesses and industry, as well as users. All stakeholders are expected to become informed about their roles and responsibilities in the safe management of recreational waters.

A preventive multi-barrier approach to management that focuses on the identification and control of water quality hazards and their associated risks before the point of contact with the recreational water user represents the best strategy for the protection of public health from risks associated with recreational waters. Reactive management strategies relying on compliance monitoring alone will not be sufficient in protecting the health of the recreational water user.

1.0 The multi-barrier approach

The multi-barrier approach is an integrated system of procedures, actions and tools that collectively reduce the risk of human exposure to recreational water quality hazards. The concept is analogous to the "source to tap" approach used for the management of safe drinking water supplies in Canada (CCME, 2004).

This approach to recreational water management has been recognized by water quality professionals worldwide. The concepts of preventive risk management and the use of multiple barriers were at the heart of the recommendations for improved management of recreational waters proposed by an international panel of experts following a meeting in Stockholm, Sweden, in 1999 sponsored by the World Health Organization (WHO). A report on the outcome of this meeting was later published in the document that has come to be known as the "Annapolis Protocol" (WHO, 1999). Similarly, it is this approach that formed the basis for the management framework outlined in the *Guidelines for Safe Recreational Water Environments* (WHO, 2003a).

The multi-barrier approach achieves success by having numerous barriers in place across all identified areas of management (e.g. source protection, monitoring, hazard control, communication, consultation), rather than focusing all efforts on a single barrier.

Specific benefits include:

- more effective public health protection;
- improved recreational water management (operational plans can be specifically tailored to address an area's individual needs and resources);
- improved public communication (leading to better public understanding of key concepts and the public's role in ensuring recreational water safety); and
- better management of emergencies (potential water quality hazards are understood and plans are in place to address the problems effectively).

The following sections provide additional information on the different elements of the multi-barrier strategy, including situation assessment (Environmental Health and Safety Surveys) and application/implementation of barriers (Compliance Monitoring, Public Awareness and Communication, Public Health Advice, Hazard Control Actions).

2.0 Environmental Health and Safety Survey

An Environmental Health and Safety Survey (EHSS) provides the foundation or "blueprint" for designing and implementing an effective risk management plan for recreational waters. It is a comprehensive search for, and assessment of, existing and potential water quality hazards (biological, chemical and physical) and their associated risks to the health and safety of the public at designated beach areas. The EHSS also represents a general review of all aspects of a beach's operation. The data collected provide beach operators, service providers and responsible authorities with the information necessary to make sound risk management decisions and to develop and maintain an effective beach monitoring program. The EHSS fits under a multi-barrier approach to recreational water management by identifying priority areas for which interventions can be applied to reduce the level of risk for recreational water users.

An EHSS should be conducted on an annual basis, just beefore the start of the swimming season. This survey should:

- catalogue the recreational water area's basic characteristics;
- identify any potential sources of faecal contamination;
- identify any other potential physical, chemical or biological water quality hazards or potential sources of such that may present a risk to recreational water users; and
- evaluate the effectiveness of the monitoring programs and risk management measures currently in place.

The authority with the best knowledge of the day-to-day operation of the beach is the likely candidate to lead this process. The EHSS process can also benefit greatly from intersectoral collaboration. Persons or groups valuable to consult on the process can include:

- the appropriate provincial or territorial management or regulatory authority;
- beach managers;
- public and environmental health departments;
- community members; and/or
- individuals representing local business and industry.

The EHSS process consists of three basic steps: pre-survey preparations, the on-site visit and the assessment report.

2.1 Pre-survey preparations

The pre-survey preparation step involves the collection and review of any and all information available on the beach and adjacent area, including reports of any previous surveys. It can provide valuable information on historical trends, problems and successes, which will help ensure a more thorough and efficient on-site visit. Initial preparations may begin with a review of basic beach information, such as the beach's physical characteristics, the types of activities practised and estimates of beach attendance. The use of topographical maps, aerial photos and geographic information system (GIS) data can provide additional perspective and help in the identification of contamination sources, potential sampling sites and nearby land uses. The examination of historically accumulated data relating to microbiological results, beach postings and disease surveillance will provide information on the area's suitability for recreation and the potential risks for swimmers. Assessment of hydrological, meteorological and other information

on rainfall, currents, tides, prevailing winds and potential discharges (sewage, storm drains, other waste discharges) can help identify their impact (either singly or collectively) on water quality.

2.2 On-site visit

The purpose of the on-site visit is to visually identify and confirm any and all existing or potential water quality hazards. Information may similarly be collected on the existence and adequacy of public facilities, safety provisions and mechanisms for public awareness and communication. For the purposes of this EHSS, a hazard is an object or condition that may endanger human health or safety. For most swimming areas, contact with faecal pollution in the environment represents a significant concern; thus, attention should be paid to the potential sources of faecal contamination, both point sources (discharge or drainage that may contain sewage, stormwater or other faecal wastes) and diffuse sources (e.g., domestic and wild animals and birds, stormwater runoff from the beach and surrounding areas, septic wastes, contamination from swimmers themselves).

Additional existing or potential hazards can include:

- chemical hazards (e.g., industrial discharges, contamination from marinas/watercraft);
- biological hazards (e.g., cyanobacterial blooms, organisms responsible for swimmer's itch); and
- physical hazards (e.g., litter, poor visibility).

Other information collected may be useful in identifying hazards that are less visible. For example, the presence of large amounts of floating debris may be indicative of sewage or stormwater discharges. An example of an EHSS checklist, which includes the type of information to be collected during an on-site visit, is provided in Appendix D.

Additionally, it is advisable to conduct site visits under both dry and wet weather conditions. The effects of rainfall and storm events on water quality should be investigated. Certain contamination events (e.g., runoff, stormwater discharges) may be visible only during rainfall periods. Representative water samples may also be collected and analysed to confirm the presence of contamination and determine its variability and source. Shortened surveys may also be carried out throughout the swimming season at the time of microbiological monitoring to collect more timely information about the recreational water area. Such information has demonstrated value in developing models that may be capable of predicting water quality. Further information on these topics can be found in Section 10.0 (Faecal pollution source tracking) and Appendix B (Microbiological sampling and analysis).

2.3 Assessment report

Once the on-site visit is completed, a risk assessment should be performed to identify priority water quality hazards. Risks, for the purpose of this EHSS, should consider the likelihood of exposure to a given hazard and the associated consequence. Conducting a proper risk assessment therefore requires the consideration of the factors that may contribute to a swimmer's exposure. These may include the proximity of the hazard to the swimming area, effects of the area's physical characteristics (depth, water circulation), potential weather influences, types and patterns of recreational activities practised in the area and impacts of any existing barriers. For example, in the case of the combined sewer overflow, factors contributing to swimmer exposure

could be heavy rains causing a discharge of sewage material, currents or winds driving this material towards the swimming area and the absence of public communication methods advising that contact with the water should be avoided for a period immediately following heavy rainfall. The risk assessment may also be used to identify potential points at which additional barriers may be needed to reduce the degree of human exposure.

The process should culminate with the production of an assessment report, which should be used when developing further beach management or operational plans. In addition to reporting on the survey findings, the report should specify priorities for action, identify barriers that may be implemented and provide recommendations for an appropriate beach monitoring program.

Recommandations for a monitoring program should identify specific sampling locations, times and frequencies, as well as outline the steps to be taken in the event that a warning or other actions are required.

The flowchart in Figure 1 (modified from Codd et al., 2005) is suggested as a possible sequence of events when designing and implementing a multi-barrier strategy for recreational waters. It may be used as a guide for beach operators, service providers or responsible authorities wishing to develop their own operational plans.

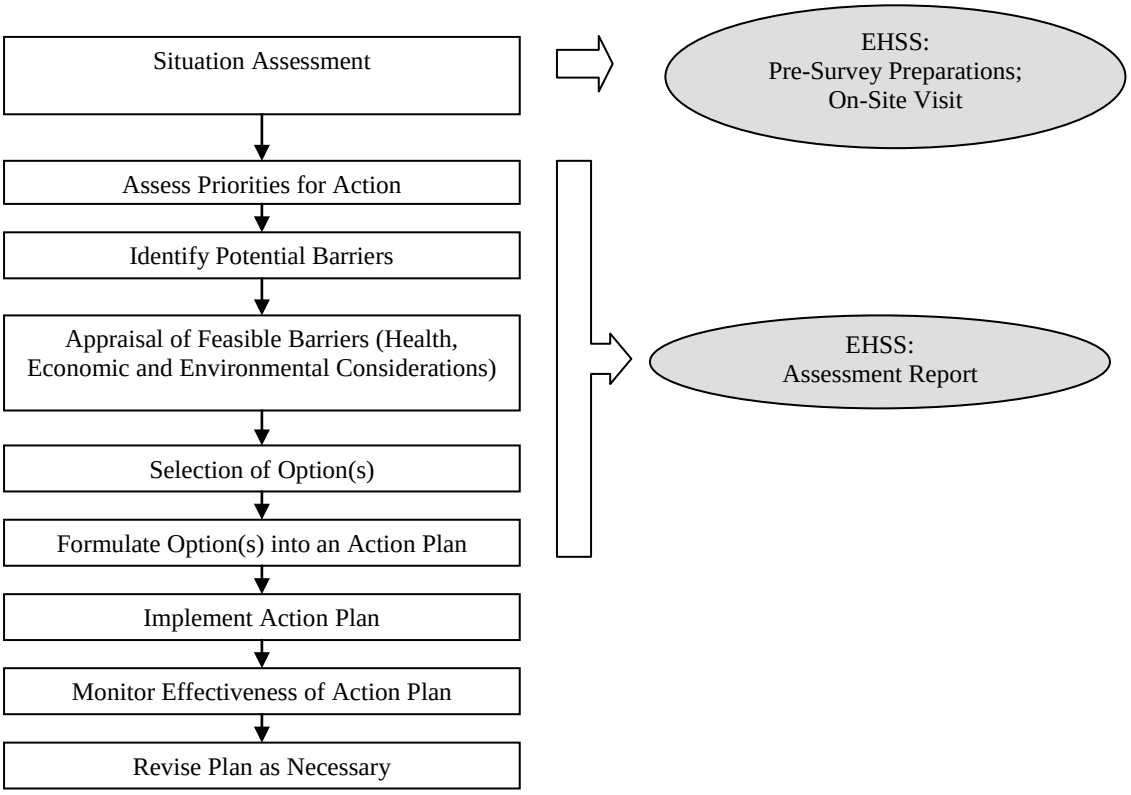

Figure 1. Possible sequence of events for designing and implementing a multi-barrier strategy for recreational waters.

3.0 Barriers for protection of recreational waters

Barriers are procedures or actions that collectively reduce the risk of human exposure to recreational water quality hazards. They can be physical actions, such as beach cleanup and grooming, or they can be processes or tools that improve the effectiveness of the recreational water management program, such as policies and legislation, guidelines and standards, and education and communication strategies.

Four major areas have been described from which barriers can be identified (WHO, 2003a): compliance monitoring; public awareness and communication; public health advice; and hazard control actions.

3.1 Compliance monitoring

Monitoring is a broad concept and can serve many functions. It can be used to:

- determine whether water quality meets the Guidelines
- identify the impacts of water quality events;
- demonstrate long-term water quality trends;
- support EHSS findings or identify gaps;
- verify that barriers (e.g. notifications, corrective actions) are put in place;
- verify that these barriers are operating effectively.

Compliance monitoring is conducted to identify existing water quality hazards and to maintain a record of changes that may occur. Proper monitoring and reporting are essential for assessing and communicating information on the level of safety of recreational waters. Decisions regarding the areas to be monitored, choice of indicators and monitoring program design will be made by the appropriate regulatory and management authorities. The monitoring program should incorporate information derived from the EHSS, taking into account recommendations made regarding priority areas of concern. There should be a documented monitoring plan for all monitored beaches, providing, at a minimum, instructions on:

- the parameters to be analysed;
- the locations at which samples are to be collected; and
- the times and frequencies of sample collection.

Recreational waters should be routinely monitored for the presence of faecal contamination, using the following primary indicators of faecal contamination:

- Fresh water: *E. coli*;
- Marine water: Enterococci.

Guideline values have been established for the geometric mean and single-sample indicator concentrations for these parameters (see Table 1 [Guidelines for Canadian recreational water quality: summary table] and Section 4.0 [Recommended indicators of faecal contamination]).

Other organisms have also been described that may have potential value as indicators by providing supplemental information with respect to the faecal contamination of recreational waters. Such organisms may also be included as part of a recreational water monitoring program,

provided they have been deemed appropriate by the responsible regulatory and management authorities.

Other water quality parameters related to the physical or aesthetic characteristics of a recreational water body and its surrounding environment have links to the health and safety of recreational water users and thus may be included in a monitoring program. Guideline values or aesthetic objectives have been specified for these parameters where they can be established (see Table 1 [Guidelines for Canadian recreational water quality: summary table] and Section 8.0 [Physical, aesthetic and chemical characteristics]).

Tests may also be carried out for specific water quality hazards when there is epidemiological or other evidence of their presence in water. Evidence may include:

- reports of a disease outbreak or illnesses of specific etiology;
- reports of a suspected illness of undetermined cause;
- reports of water-related injuries;
- levels of an indicator strongly suggesting the presence of a specific hazard;
- reports of a specific event such as a sewage or chemical spill; or
- the development of a cyanobacterial bloom.

Procedures may also be established for the monitoring of other barriers that may be utilized for recreational water management. Examples may include:

- public notification / warning signs posted;
- water quality hazard control actions implemented;
- health authorities notified; and
- policies, legislation, protocols and/or guidelines established and in place.

It is recognized that recreational water areas will each have unique characteristics and operational considerations. Decisions regarding the specific design of the recreational water monitoring program should be made by the appropriate authorities (e.g. local, regional, federal, provincial, territorial, as applicable). These should take into consideration the specific needs and conditions of the area, the types of users and recreational activities practised, and any relevant historical information.

3.1.1 Frequency of microbiological sampling

Decisions regarding the frequency of water samples collected for microbiological analysis should be made by the appropriate local or regional authority. Guidance is provided on some of the factors to be considered when selecting sampling frequency. Published texts are available that can provide further information with respect to the design and implementation of recreational water monitoring programs (e.g., Bartram and Rees, 2000).

There are a number of factors that can influence the microbiological quality of a recreational water body at any given time. These can include the type and periodicity of contamination events (both point and non-point sources), the time of day, recent weather conditions, the number of users frequenting the swimming area and the physical characteristics of the area itself. The significant day-to-day (and within-day) variation in indicator organism densities observed for

recreational waters has been well documented (Leecaster and Weisberg, 2001; Boehm et al., 2002; Whitman and Nevers, 2004; U.S. EPA, 2005a).

Results from the U.S. Environmental Protection Agency's (EPA) Environmental Monitoring for Public Access and Community Tracking (EMPACT) Project identified that it is the day-to-day variation that is the principal source of uncertainty when attempting to develop an estimate of the water quality for a recreational water area over a given period (U.S. EPA, 2005a). Correlations were shown to exist only relative to indicator levels measured on the very next day. Indicator levels demonstrated only a negligible correlation with those measured more than 2 days later.

More frequent monitoring (daily as opposed to weekly sampling; weekly as opposed to monthly sampling) will have several advantages. As a result of the significant day-to-day variation in faecal indicator counts that can be observed, even daily monitoring will not necessarily improve the ability of the current day's microbiological results to predict the next day's water quality. However, the additional information provided by increasing the number of samples will allow the responsible authorities to more easily observe water quality trends and to make more informed decisions regarding the area's overall suitability for recreation. Moreover, it will enable authorities to more quickly detect persistent water quality problems that may occur.

The Guidelines advocate the use of both a maximum limit for the geometric mean faecal indicator concentration and a single-sample maximum limit. The use of dual limits allows recreational water operators to better evaluate the water quality both in the short term and over the duration of the swimming season. The single-sample limit will alert management to any immediate water quality issues, whereas the geometric mean limit will alert management to chronic contamination problems. This dual approach represents good monitoring practice as part of an overall commitment to a strategy of risk management for recreational waters.

In order to ensure that human health is adequately protected, waters regularly used for primary contact recreational activities should be monitored at a minimum frequency of once per week during the swimming season. A weekly monitoring strategy is useful to alert managers and responsible authorities to more persistent contamination problems that may have developed and allows them to make the necessary decisions within a reasonable time frame. Weekly surveillance is consistent with recommendations made by the U.S. EPA in its 2002 *Implementation Guidance for Ambient Water Quality Criteria for Bacteria*.

When sampling, consideration should also be given to the collection of samples for the purpose of characterizing event-driven episodes of pollution that may affect recreational waters—for example, immediately following periods of heavy rainfall or at times of greatest swimmer activity.

In areas where high swimmer densities are expected, increased monitoring is recommended. In such situations, the number of samples may be increased to permit the calculation of a weekly or even daily geometric mean (based on a minimum of five samples), if so desired.

Similarly, certain circumstances may permit a reduction in the recommended sampling frequency. These may include the existence of beaches in remote locations or in areas where

primary contact recreational activities are not a regular occurrence or beaches that have historically demonstrated acceptable water quality. Once an understanding of water quality behaviour at a site has been achieved through relatively intensive monitoring and the use of an Environmental Health and Safety Survey, a reduction in sampling frequency may be justifiable and can help ease the burden of monitoring (Bartram and Rees, 2000; WHO, 2003a). Thus, if it can be determined that a recreational water area is of consistently good microbiological quality, does not have any obvious sources of faecal contamination and is not considered to present a significant risk to the health and safety of its users, monitoring may be reduced to a frequency sufficient to verify that the conditions have not deteriorated.

It may also be acceptable to reduce monitoring frequencies for recreational water areas that consistently demonstrate poor water quality results, but only where appropriate management actions are taken to discourage recreational use, and provided that the risks are clearly communicated to the public.

3.1.2 Location of microbiological sampling

Decisions regarding the most appropriate location and depth of water samples collected for microbiological analysis should be made by the appropriate local or regional authority. Guidance is provided below on some of the factors to be considered when selecting sampling locations and depths. Published texts are available that can provide further information with respect to the design and implementation of recreational water monitoring programs (e.g., Bartram and Rees, 2000)

Most bodies of water used for recreational purposes are not completely homogeneous with respect to their microbiological properties. In recreational water evaluations, the purpose of sampling is to obtain aliquots that are as representative as possible of microbiological quality of the area. A single water sample provides a quantitative estimate of the indicator bacteria present at a particular site and time. Whitman and Nevers (2004) observed that there can be significant variation between samples collected at multiple points along the beach, as well as among samples collected within close proximity to each other at nearly the same time. As the total number of samples increases, the more representative the data will be of the overall water quality.

Sites should be chosen to be representative of the water quality encountered throughout the entire swimming area. Consideration should be given to specific conditions that may influence the levels and distribution of indicator organisms and pathogens. The sampling sites should include points of greatest swimmer activity, as well as peripheral points subject to external faecal pollution. Stormwater, sewage or river outlets can give certain sections of a body of water microbiological qualities that are very different from those of the water body at large. The degree of heterogeneity can also be affected by rainfall, wind direction and velocity, currents and tides, or the presence of physical barriers, such as sandbars, natural or artificial wave breaks and piers.

The depth at which samples are collected can have a significant effect on the resulting estimates of water quality (for the purposes of these Guidelines, ‘depth’ refers to the vertical distance in the water extending from the bottom of the water to the surface). Where the water is very shallow, disturbances of the foreshore sand and sediment caused by wave swash and swimmer

activity can result in the resuspension of microorganisms. Where the water is deeper, this effect has lesser influence on water quality measurements. In contrast, deeper waters are relatively more exposed to offshore faecal contamination sources than shallower waters (U.S. EPA, 2005a).

Adult chest depth (approximately 1.2–1.5 m) has historically been the most common sampling depth. Traditionally this has been considered to represent the depth of greatest swimmer activity and the location nearest to the point of head immersion, which would be indicative of the risk associated with accidentally swallowing water. Published epidemiological studies have typically found that only samples collected at this depth show evidence of a mathematical relationship between indicator organism density and swimmer illness.

Sampling at shallower depths (ankle or knee depth—approximately 0.15–0.5 m) may be more representative of water quality encountered by young children playing at the water's edge. It is expected that more frequent swimming advisories would be issued if this monitoring approach were used. Sand and sediment disturbances can result in increased microbiological numbers in shallower waters. Currently there is insufficient evidence to determine whether the expected increase in the number of swimming advisories at this monitoring depth would result in a proportionate reduction in the number of swimmer illnesses

Another strategy for monitoring that has been proposed involves the attempt to strike a balance between the depth at which the majority of the health effects have been proven and the depth at which microbiological counts are thought to be the highest (U.S. EPA, 2005a). According to the recommendations outlined in the U.S. EPA's EMPACT report (U.S. EPA, 2005a), sampling in water of knee to waist depth may offer a reasonable, but still conservative, approach to monitoring.

Alternatively, another approach would be for authorities to sample at multiple depths—for example, at ankle to knee depth, as well as at waist to chest depth. Such a design could be used to produce separate estimates of the water quality in both shallow waters and waters of swimming depth. However, when comparing the results of water samples between shallow and deeper waters, samples collected at a specific depth should be analysed as a singular group in order to improve the precision of the data.

The sample depth (or depths) selected for an individual beach should be determined by the local or regional authority in order to obtain the best information for their particular recreational water area.

3.1.3 Other monitoring tools

Composite sampling

The use of composite sampling techniques presents a possible means for increasing the area covered under a beach monitoring program, while potentially minimizing the costs associated with analysis. Composite sampling involves the collection of multiple samples from across a stretch of beach, combining them into one large composite, and then analysing a subsample of the resulting mixture.

There are obstacles to be overcome with the use of this technique. Increased sampling is required initially to validate whether composite sampling will be feasible at a given area. As well the area must be characterized to identify hot spots (sampling points having continuously poor water quality) that can disrupt the analysis. Some statistical knowledge is also required to analyse the data. Nevertheless, preliminary investigations have suggested that, if properly conducted, composite sampling can be used in making water quality decisions with a comparable degree of accuracy to that achieved by analysing the samples individually and averaging the results (Kinzelman et al., 2006). Further information on composite sampling can be found in Appendix C (Composite sampling for faecal contamination).

Predictive water quality models

An emerging area of research has been the development of predictive models that are able to make same-day predictions about the microbiological quality of the water. Researchers have developed and validated models for specific beaches to make predictions of water *E. coli* levels using data from various water- and weather-related parameters (e.g., rainfall, wave height, wind direction, turbidity, previous day's faecal indicator counts). There are a few models currently used in beach monitoring programs in the United States:

- SwimCast, in use at several Lake Michigan beaches in Lake County, Michigan uses air and water temperature, wind speed and direction, precipitation, relative humidity, wave height, lake stage, sunlight and other water quality parameters to predict whether current *E. coli* levels are suitable for swimming (Olyphant and Pfister, 2005; Lake County Health Department, 2010).
- Project S.A.F.E. (Swimming Advisory Forecast Estimate) in use at five Lake Michigan beaches in northern Indiana uses wind direction, gauge height of the nearby Burns Ditch, rainfall in the previous 48 hours and lake conditions such as chlorophyll and turbidity to make predictions about relative *E. coli* levels and swimmability (Whitman, 2005; U.S. Geological Survey, 2007).
- Nowcasting in use at Huntington and Edgewater beaches on Lake Erie in Ohio makes use of $\log_{10}$ turbidity data, wave height and radar rainfall from grids surrounding the respective beach areas during the previous 24 or 48 hours to estimate if *E. coli* levels are acceptable for swimming at these beaches (Francy, 2007).

Evidence generated to date suggests that a properly developed model can achieve a degree of accuracy comparable to that achieved by traditional approaches that use the previous day's indicator concentrations.

There are a number of challenges associated with model development. A significant level of technical expertise is required to develop the models and to analyse the data, and models may not work in all areas. Nevertheless, they do present a possible means for forecasting the quality of the water, thereby improving the timeliness of decisions of whether to open or close a beach. Beach operators, service providers or responsible authorities looking for an additional tool with which to potentially improve the timeliness of their water quality decisions may wish to investigate this approach.

3.2 Public awareness and communication

In order to participate in safe, enjoyable recreational water activities, the public requires access to information on the quality of the area and its facilities, as well as notification of any existing water quality hazards. Beach operators, service providers and responsible authorities have a responsibility to inform and educate the public and provide adequate warnings about any hazards relevant to their recreational water areas.

Efforts to improve the public's awareness and understanding of water quality can have numerous benefits (Bartram and Rees, 2000; Pendleton et al., 2001). Communication tools can be used to:

- reduce the potential risk of swimmer illness or injury;
- improve the quality of the water;
- correct public misconceptions regarding water quality;
- improve public confidence; and
- increase beach attendance.

Posting of information at recreational water areas

Information on the quality of the water should be communicated to the user through the use of posted signs. Signs should be used to warn users when the water is unsafe for recreational use as well as to communicate when the water is safe for use.

Signs should be posted in locations that are highly visible to the public. The information provided should be easily understood and not open to misinterpretation. Ideally, signs should be standardized to permit comparisons across different locations. Warnings should be timely and should be promptly removed once the issuing authority has determined that the risk no longer exists.

The information provided on warning signs should include, at a minimum:

- a statement identifying the health or safety risk;
- recommended actions to be taken;
- the name of the authority responsible for issuing the warning; and
- contact information for the issuing authority.

Similarly, during periods in which the area is considered suitable for use, corresponding signs should be posted that clearly communicate this information to the public. One important concept to communicate to beach users through education is that even in waters considered of good quality for swimming, there is always some probability that swimmers may experience some adverse health effects.

Examples of informative beach signs are provided in Appendix F.

There are two main situations under which a warning sign may be posted—following the issuing of a swimming advisory and following the issuing of a beach closure. Issuing a swimming advisory or a beach closure should be made by the Medical Officer of Health or other appropriate authority in accordance with the statutes existing in each province or territory. This decision should be based on a thorough assessment of the situation with information provided by recreational water monitoring, the EHSS and existing public health surveillance.

A swimming advisory can be issued if the responsible authority identifies that the water is not suitable for recreational use. Under this situation, users are advised to refrain from whole body contact with the water. Contact with the beach is usually permissible, and access to the facilities is generally not restricted. Examples of scenarios that may trigger jurisdictions to decide to issue a swimming advisory include:

- exceedance of the guideline values for the recommended indicators of faecal contamination;
- exceedance of the guideline values for toxic cyanobacteria and their toxins, or in the event of the development of a cyanobacterial bloom;
- existence of evidence of the risk of swimmer's itch for recreational water users; and
- after periods of significant rainfall, which could trigger an advisory as a pre-emptive action.

A beach closure can be issued if the responsible authority identifies that a beach or body of water poses a serious risk to the health and safety of recreational water users, and that it is further necessary to restrict individuals from coming in contact with the area. Under a closure, the area is considered closed to all recreational activity. Users are advised to avoid contact with the beach and recreational water area, and access to the facilities may be restricted. Examples of situations in which jurisdictions may determine it necessary to warrant the issuing of a beach closure include:

- suspicion that the area is responsible for a waterborne disease outbreak;
- a sewage or chemical spill that is expected to affect the recreational water area;
- other conditions such that the area is judged to pose a significant risk to public health (e.g., persistently poor water quality); and
- detection of a cyanobacterial bloom.

Appropriate signs may also be posted at waters that have been deemed suitable for secondary contact activities (e.g., rowing, sailing, canoe touring, fishing), but not for primary contact uses (e.g., swimming, wading, windsurfing, waterskiing). In these instances, it may be necessary to expand the location of the signs beyond the beach area to improve their visibility. Suggested locations include relevant points of entry and launch areas.

The public can also do their part by: educating themselves on actions they can take to protect themselves and the beach; becoming aware of where the water quality monitoring results are posted; and, consulting this information before going to the beach.

Other tools for public education and communication

In addition to posted signs, other tools can be used for public education and the communication of information, including:

- printed materials (e.g., posters, information sheets, educational bulletins, pamphlets, brochures);
- media sources (e.g., local newspapers, television and radio announcements, Internet websites);
- participation in beach certification or award programs;
- educational events such as volunteer monitoring programs and beach cleanup days; and
- classification or grading systems for beaches.

Classification or grading systems for beaches have received interest as a tool to promote communication and understanding of water quality information. They are also thought to help encourage a sense of shared accountability and responsibility among the beach authorities and beach users. A number of jurisdictions and multinational organizations have adopted grading systems as part of their recreational water management recommendations (WHO, 2003a; MDDEP, 2004; NHMRC, 2008). Both the WHO and the Australian Guidelines (WHO, 2003a; NHMRC, 2008) make use of a grade-based format for faecal indicator density within their framework for assessing faecal pollution in recreational waters. These approaches use a microbiological assessment component along with a sanitary inspection categorization to produce a classification of an area's overall suitability for recreation. Under the Quebec framework for faecal pollution assessment (MDDEP, 2004), water quality grades are assigned to recreational waters based on average yearly faecal indicator monitoring results. This grade is used as a public communication tool and also dictates the sampling frequency requirements for the area.

There are advantages and disadvantages associated with the use of beach grading systems. Responsible authorities should be aware of the limitations of any system when investigating its use as a potential communication tool. For example, grading systems based on faecal indicator monitoring results provide information on only one aspect of recreational water safety. Moreover, the accuracy of any such grading system would be strongly affected by the limitations known to be associated with faecal indicator monitoring. Ideally, a successful beach grading system would involve criteria from a number of categories, capturing monitoring results, communication tools and water quality hazard control actions in place.

3.3 Public health advice

Consultation with public health authorities is another essential component of risk management. In the event of an incident (microbiological, chemical or physical) that represents a risk to public health or safety, health officials can play a key role by providing advice and determining what actions need to be taken. Local public health authorities should be promptly notified of any situation that threatens the health or safety of recreational water users. Similarly, as part of normal operations, local public health officials may be periodically consulted for information and advice on topics pertinent to safe recreational water use.

In assessing the risks associated with recreational water quality hazards, the local health authorities should, wherever possible, establish surveillance for swimmer illness or injuries. This can be established by consulting public health surveillance mechanisms currently in operation or by conducting specific investigations. Information sources include:

- federal, provincial/territorial or regional departments or agencies having surveillance programs or reporting systems;
- clinical reports from hospital emergency departments and local physicians;
- accident or incident reports held by recreational water area operators or service providers;
- formal epidemiological investigations; and
- other potential surveillance mechanisms (e.g., monitoring of over-the-counter medicine sales in pharmacies).

Procedures for the investigation of illness associated with recreational waters should adhere to the recommendations given in the second edition of *Procedures to Investigate Waterborne Illness* (International Association for Food Protection, 2002).

3.4 Hazard control actions

Hazard control actions are physical actions intended to reduce the impact of microbiological, chemical or physical water quality hazards on a particular recreational water site. A discussion on the numerous types of possible hazard control actions is outside the scope of this document. In addition, the types of hazard control actions required and their relative effectiveness will be specific to each beach and each situation. Authorities may wish to consult the following types of resources for information to help them address specific beach-related issues:

- Published texts: Texts are available which provide comprehensive discussion of larger topics like stormwater management, wastewater treatment and coastal water management.
- Searchable databases of journal articles: Keyword searches can direct the user to specific scientific studies, related articles, bibliographical citations and topic review papers.
- Proceedings from international conferences: Review of proceedings can identify accounts of actions evaluated in other communities, providing an indication of results and potential contacts; specific conferences include: Great Lakes Beach Association Conferences (www.great-lakes.net/glba), and the U.S. EPA National Beach Conferences (www.epa.gov).
- Manuals or publications produced by stakeholder organizations (e.g., Griffiths, 1999).

Assessment may help identify smaller-scale actions that may provide good returns from a cost–benefit perspective. However, some issues may be more substantial in nature and thus may not be easily resolved without the application of more sophisticated control methods. Examples of small-scale control actions aimed at reducing faecal contamination can include beach cleanup or grooming procedures or installation of structures (fences, overhead wires) to discourage the presence of birds and other wildlife. Examples of larger-scale actions can include sewage treatment or storm drain waste retention. Any potential control action should be assessed from the perspectives of the effects on the health of the users and on the environment.

Water quality issues can cross over multiple boundaries (e.g., health, environment, agriculture, municipal infrastructure), and require cross-sectoral collaboration. Consultation with responsible authorities, other beach operators or service providers and recreational water quality professionals may help to identify actions that have proven to be successful in other communities.

Part II: Guideline Technical Documentation

4.0 Recommended indicators of faecal contamination

Recreational waters may be contaminated with faecal material from such sources as discharged sewage, stormwater runoff from agricultural or urban areas, wild or domesticated animals, and even through faecal shedding by swimmers themselves. Many epidemiological studies have identified gastrointestinal and upper respiratory illnesses in swimmers as a result of such contamination. Historically, the bacteria in the coliform group and its subgroups (total coliforms, thermotolerant [faecal] coliforms, *E. coli*) and the enterococci—the more faecal-specific portion of the faecal streptococci group—have been used to monitor recreational waters for the presence of faecal contamination. As such, they have also been used to indicate the possible presence of pathogenic microorganisms responsible for these illnesses. Routine testing of recreational waters for pathogenic organisms is impractical and is not recommended for the following reasons:

- Testing for every possible waterborne disease-causing microorganism would be prohibitive in terms of both the financial resources necessary and the time required to perform the analyses. These organisms are difficult to isolate and quantify, and testing requires proper laboratory containment facilities, specialized equipment and highly trained and experienced microbiologists. Detection methods for some pathogens do not exist at all.
- Pathogens are usually present at low levels and are unevenly distributed in recreational waters, even during disease outbreaks.
- The absence of one pathogen does not necessarily ensure that other enteric pathogens are also absent.

Consequently, authorities monitor for non-pathogenic faecal indicator bacteria that are present in high numbers in both human and animal faeces. Elevated numbers of these indicator bacteria in the aquatic environment are indicative of faecal contamination and therefore suggest the possible presence of enteric pathogens.

The ideal faecal indicator organism would meet the following requirements (Cabelli et al., 1983; Elliot and Colwell, 1985):

- found within the intestinal tract of humans and warm-blooded animals;
- present in faecally contaminated waters when enteric pathogens are present, but found in greater numbers than pathogens;
- incapable of growth in the aquatic environment, but capable of surviving longer than pathogens;
- applicable to all types of natural recreational waters (fresh, estuarine and marine waters); and
- absent from non-polluted waters and exclusively associated with animal and human faeces.

Other desirable qualities for the indicator organism include:

- Density of the indicator should be directly correlated with the degree of faecal contamination.
- Density of the indicator should be quantitatively related to swimmer-associated illnesses.
- Detection and enumeration test methods should be rapid, easy to perform, inexpensive, specific and sensitive.

No single microorganism unequivocally meets all of these criteria. *E. coli* and enterococci are currently considered the best indicators of faecal contamination in recreational waters, as they most closely fit the above characteristics. There are limitations associated with the use of indicators in assessing the quality of recreational waters. Judicious use of these guideline values as part of a multi-barrier approach to recreational water management represents a sound approach to protecting swimmers against exposure to faecal pathogens in the recreational water environment.

4.1 Indicator organisms for primary contact recreation

4.1.1 Fresh waters: Escherichia coli (E. coli)

Guideline values
For fresh recreational waters used for primary contact activities, the guideline values are as follows:

Geometric mean concentration (minimum of five samples): ≤ 200 *E. coli*/100 mL Single-sample maximum concentration: ≤ 400 *E. coli*/100 mL

Calculation of the geometric mean concentration should be based on a minimum of five samples, collected at times and sites so as to provide representative information on the water quality likely to be encountered by users. Further action should be initiated if either of these guideline values is exceeded. Minimum action should consist of immediate resampling of the site(s). In addition, a swimming advisory may be issued should the responsible authority identify that the area is not suitable for recreational water use.

It is further advised that recreational water areas routinely used for primary contact recreation be monitored at a minimum of once per week, with increased monitoring recommended for those beaches that are highly frequented or are known to experience high user densities. Similarly, under certain scenarios, a reduction in the recommended sampling frequency may be justified. Further guidance on sampling frequency recommendations and the posting of recreational waters can be found in Part I (Management of Recreational Waters).

Enterococci (Section 4.1.2) is also recognized as a suitable indicator of faecal contamination in fresh recreational waters (Cabelli, 1983; Pruss, 1998; Wade et al., 2003, 2006). If it can be shown that enterococci can adequately demonstrate the presence of faecal contamination in fresh waters, then the enterococci maximum limits for marine waters may be adopted. If there is any doubt, samples should be examined for both sets of indicators for extended periods to determine whether a positive relationship exists.

Guideline rationale

The guideline values have been developed based on epidemiological evidence relating *E. coli* concentrations in fresh recreational waters to the incidence of swimming-associated gastrointestinal illness observed among swimmers. The existing epidemiological data are not sufficient to permit the estimation of the level of risk for individual exposures. Based on the U.S. EPA's regression analysis of epidemiological data (Dufour, 1984), Health Canada has estimated that using the guideline values for the recommended indicators of faecal contamination for fresh and marine waters will correspond to a seasonal gastrointestinal illness rate of approximately 1–2% (10–20 illnesses per 1000 swimmers).

In determining the value for the maximum faecal indicator concentration permitted in a single sample, the U.S. EPA equations pertaining to the calculation of a single-sample limit were reviewed (U.S. EPA, 1986). The data regarding the maximum allowable indicator density at designated beach areas is consistent with applying a factor of 2 times the recommended geometric mean value results. Thus, the single-sample maximum concentration of 400 *E. coli*/100 mL is reaffirmed.

These values represent risk management decisions that have been based on a thorough assessment of the potential risks for the recreational water user. In considering both the potential health risks and the benefits of recreational water use in terms of physical activity and enjoyment, it was concluded that this is a tolerable and reasonable estimate of the risk of illness likely to be experienced by users engaged in a voluntary activity.

An evaluation of the epidemiological information published since the Guidelines were last issued concluded that the current body of evidence supports the existing recommendations regarding the use of *E. coli* as the indicator of faecal contamination in fresh recreational waters. There has not been substantial evidence to suggest that revision of the existing guideline values is necessary at this time.

Description

E. coli most closely fits the requirements of an ideal indicator of faecal contamination for fresh waters. The organisms are found in high numbers in the intestinal tract and faeces of humans and warm-blooded animals. The vast majority of the *E. coli* types are harmless. There are several types (serotypes or strains) that possess virulence factors enabling them to act as human pathogens; however it should be noted that faecal concentrations of the typical non-pathogenic *E. coli* will always be greater than those of the pathogenic strains, even during outbreaks. *E. coli* is considered a more specific indicator of faecal contamination than either total coliforms or thermotolerant (faecal) coliforms and can be rapidly and easily enumerated in recreational waters.. In addition, a strong correlation has been demonstrated between the concentration of *E. coli* in fresh waters and the risk of gastrointestinal illness among swimmers (Dufour, 1984; Wade et al., 2003).

For several decades now, recreational water quality experts in Canada have recognized *E. coli* as the indicator of choice for faecal contamination. The use of *E. coli* as an indicator of recreational water quality was of limited use until the 1980s, when standardized laboratory methods permitting detection of the organism within 24–48 hours became available. Prior to this, the

thermotolerant coliform group was used as the primary indicator of faecal contamination in recreational waters. However, subsequent findings that some thermotolerant coliform species had non-faecal or environmental origins and could be isolated in high numbers from waters receiving waste effluents from such sources as pulp and paper and textile manufacturing facilities (Dufour and Cabelli, 1976; Huntley et al., 1976; Rokosh et al., 1977; Vlassoff, 1977) raised concerns about the reliability of using this group as an indicator of faecal contamination in recreational waters. Despite the availability of methods specific for the detection of *E. coli*, testing laboratories were already set up to test for thermotolerant coliforms, and the requirements to perform surveillance of recreational waters for these microorganisms were embedded in long-standing regulatory and legislative documents. As a result, it has taken many years and considerable time to amend the existing guidelines and standards, updating them to reflect the current state of knowledge that *E. coli* is the preferred indicator of faecal pollution for fresh recreational waters.

In the 1992 edition of the Guidelines, the introduction of *E. coli* as the recommended indicator for freshwater quality represented a new direction for recreational water monitoring, moving away from the 1983 recommendation regarding the use of thermotolerant coliforms. As a result, a provision was made that allowed thermotolerant coliforms to continue to be used if it could be determined that greater than 90% of the thermotolerant coliforms were, in fact, *E. coli*. This was done to permit an adjustment period for jurisdictions in making the changeover to the new recommendations. Significant time has now passed to allow jurisdictions to make the change from thermotolerant coliforms to the more faecal-specific *E. coli*. As a result, this third edition does not recommend the use of thermotolerant coliforms as an indicator of the quality of recreational waters. Instead, it reaffirms that *E. coli* is the preferred indicator for monitoring fresh recreational waters in Canada.

Occurrence in the aquatic environment

Within human and animal faeces, *E. coli* is present at a concentration of approximately 10^9 cells per gram (Edberg et al., 2000) and comprises about 1% of the total biomass in the large intestine (Leclerc et al., 2001; Health Canada, 2012a). Human faecal flora characterization studies have reported that *E. coli* was detected in 94% and 100% of the subjects tested (Finegold et al., 1983; Leclerc et al., 2001). These values were significantly higher than those reported for other members of the coliform group and were matched or exceeded by only enterococci and certain species of anaerobic bacteria (*Bacteroides*, *Eubacterium*).

E. coli comprises about 97% of the coliform organisms in human faeces, with *Klebsiella* spp. comprising 1.5% and *Enterobacter* and *Citrobacter* spp. together comprising another 1.7%. *E. coli* has been shown to represent between 90% and 100% of all coliforms in faeces from eight species of domestic animals, including chickens (Dufour, 1977).

Once shed from a human/animal host, faecal bacteria are not expected to survive for long periods in the aquatic environment (Winfield and Groisman, 2003). Survival of *E. coli* in the recreational water environment is dependent on many factors, including temperature, exposure to sunlight, available nutrients, water conditions such as pH and salinity, and competition from and predation by other microorganisms.

Numerous authors have reported on the ability of beach sand and sediments to prolong the survival of faecal microorganisms (Whitman and Nevers, 2003; Ishii et al., 2006a; Kon et al., 2007a). This environment is thought to provide more favourable conditions of temperature and nutrients than the adjacent waters, as well as to offer protection from certain environmental stressors such as sunlight. Others have reported on the ability of *E. coli* to survive in organic-rich environments not known to be associated with faecal contamination, such as industrial process wastes and wastes from pulp and paper manufacturing (Megraw and Farkas, 1993; Gauthier and Archibald, 2001). Recently, researchers have reported on the ability of *E. coli* and other faecal bacteria to survive within mats of the green algal species *Cladophora* (Whitman et al., 2003; Olapade et al., 2006).

Historically it was believed that *E. coli* could only originate from faecal sources and was incapable of growth in the aquatic environment. However, recent studies are raising questions regarding these assumptions (Kon et al., 2007b; Hartz et al., 2008, Vanden Heuvel et al., 2009). Research in this area is ongoing. These recent findings do not invalidate the use of *E. coli* as the best available indicator for recreational water quality.

E. coli is considered to be a good surrogate of the survival of enteric bacterial pathogens in recreational waters. Several authors have reported similar survival rates for *E. coli* and enteric bacterial pathogens (Rhodes and Kator, 1988; Korhonen and Martikainen, 1991; Chandran and Mohamed Hatha, 2005). The indicator is regarded to be more sensitive to environmental stresses than human enteric viruses and protozoa and thus does not survive as long in the environment as these organisms.

In many parts of Canada, freshwater beaches are routinely monitored for levels of *E. coli* as an indication of faecal contamination. Many Canadian recreational waters are of good microbiological quality; however, certain waters are contaminated throughout part, or all, of the swimming season. An examination of beach monitoring data over a 10-year period (1993–2003) at 10 Lake Huron recreational beaches in Ontario demonstrated that levels of *E. coli* can vary widely at a single location from year to year and between beach locations (Ontario Ministry of the Environment, 2005). *E. coli* values can range from 0/100 mL in isolated areas to several thousand per 100 mL in areas directly impacted by faecal contamination (Payment et al., 1982; Sekla et al., 1987; Williamson, 1988; Ontario Ministry of the Environment, 2005).

Association with pathogens

E. coli is considered a good indicator for enteric bacterial pathogens such as *Salmonella*, *Shigella*, *Campylobacter* and *E. coli* O157:H7 (Health Canada, 2012a). Investigations conducted by the Water Environment Research Foundation (Yanko et al., 2004) examined the relationship between *E. coli* concentrations in surface water samples collected from various watersheds in southern California and the probability of detecting *Salmonella* and Shiga toxin-producing *E. coli* (STEC). The results demonstrated that the probability of detecting *Salmonella* using culture-based methods steadily increased up to a concentration of approximately 1000 *E. coli*/100 mL. At this point, a 100% probability of detection was reported. Similar results were reported for the detection of STEC strains using polymerase chain reaction (PCR)-based methods. Although there was clearly an association between *E. coli* concentrations and the probability of detection of *Salmonella* and STEC strains, the authors reported that no single

sample could be used to provide absolute assurance of the presence or absence of these pathogens.

E. coli is a less effective indicator of enteric pathogenic viruses and protozoa. Numerous studies have reported on the lack of a correlation between *E. coli* concentrations and the presence of enteric viruses and protozoa in surface waters (Griffin et al., 1999; Denis-Mize et al., 2004; Hörman et al., 2004; Dorner et al., 2007).

E. coli are always present in faecal contamination from human and animal sources. Detection indicates faecal contamination of water and thus the possible presence of faecal pathogenic bacteria, viruses and protozoa. The occurrence of faecal pathogens in recreational waters is strongly dependent on the nature of the contamination sources impacting the swimming area. Their presence and numbers in the environment can be sporadic and highly variable. As well, some enteric pathogens can survive longer than the faecal indicators. The absence of *E. coli* should not be interpreted to mean that enteric pathogenic microorganisms are also absent.

The combination of routine *E. coli* monitoring alongside actions, procedures and tools to collectively reduce the risk of swimmer exposure to faecal contamination in the recreational water environment represents the most effective approach to protecting the health of recreational water users.

Guidelines used by other countries/organizations

The guideline formats and values established by other government and multinational organizations worldwide for faecal indicator organisms in fresh waters were reviewed in developing the revised edition of this document.

Table 2. Guideline values for faecal indicator concentrations in fresh recreational waters established by other countries or organizations.

Country/ organization	Freshwater indicator	Format and guideline values	Reference
U.S EPA[a]	*E. coli*	Geometric mean concentration: 126/100 mL Single sample maximum concentration :[b] 235/100 mL	U.S EPA, 2002
	Enterococci	Geometric mean concentration: 33/100 mL Single sample maximum concentration:[b] 62/100 mL	
WHO	Intestinal enterococci[c]	95th percentile/100 mL: A: ≤40 B: 41-200 C: 201-500 D: >500	WHO, 2003a
Australia	Intestinal enterococci[c]	95th percentile/100 mL: A: ≤40 B: 41-200 C: 201-500 D: >500	NHMRC, 2008
European Union	Intestinal enterococci	95th percentile/100 mL: Excellent: 200 /100 mL	EU, 2006

	Good: 400/100 mL
E. coli	90th percentile/100 mL: Sufficient: 330/100 mL 95th percentile/100 mL: Excellent: 500 /100 mL Good 1000/100 mL
	90th percentile/100 mL: Sufficient: 900/100 mL

a – Note that new criteria are currently under development and may be available in 2012.
b – Designated beach area (75% confidence level)
c – Recommends guidelines for coastal waters be used until more freshwater data is available.

Related epidemiological studies

The U.S. EPA's original epidemiological study in fresh recreational waters. measured the concentrations of faecal indicator organisms (thermotolerant (faecal) coliforms, *E. coli*, enterococci) in swimming waters and compared them with the rates of swimming-associated illness reported on the same days on which the samples were collected (Dufour, 1984). Statistically significant rates of gastrointestinal illness were observed among swimmers in waters considered to be more faecally contaminated. For symptoms unrelated to gastrointestinal illness, no statistically significant differences were observed. For the data analysis, the mean seasonal faecal indicator concentration per 100 mL was plotted against the seasonal swimming associated rate of gastrointestinal symptoms per 1000 persons for each indicator. Correlation and regression analysis were then used to determine the correlation coefficients and the slope of the linear regression equation for each indicator. The best correlation coefficient (r) was obtained with *E. coli* (r = 0.80) and an almost equal correlation coefficient was observed with enterococci (r = 0.74). The E. coli data were used to produce a regression equation:

$$\text{seasonal risk of gastrointestinal illness per 1000 persons} = 9.40\ (\log \textit{E. coli} \text{ concentration per 100 mL}) - 11.74$$

Several freshwater epidemiological studies have been conducted since the previous version of the *Guidelines for Canadian Recreational Water Quality* was developed (Lightfoot, 1988; Ferley et al., 1989; Calderon et al., 1991; van Asperen et al., 1998). All have confirmed the existence of a strong relationship between exposure to recreational waters and swimming-associated illness; however, few have been able to demonstrate evidence of a mathematical relationship between faecal indicator counts and illness among swimmers. Van Asperen et al. (1998) reported that the risk of gastroenteritis was significantly higher among triathletes who swam in water with a geometric mean concentration of > 355 *E. coli* colony-forming units (cfu)/100 mL (equivalent to *E. coli*/100 mL). Ferley et al. (1989) proposed that faecal streptococci were a better indicator of gastrointestinal illness at freshwater beaches in France. Calderon et al. (1991) observed that total staphylococci counts were most closely associated with gastrointestinal illness among swimmers at a recreational pond not affected by point source discharges.

Only a few epidemiological studies have been conducted that have investigated the health effects of recreational activities other than swimming, such as whitewater canoeing and rafting (Fewtrell et al., 1992; Lee et al., 1997). The data linking water quality and illness through these activities

have been less strong. Still, conclusions that have been reported are that gastrointestinal illness similarly constitutes the most frequently reported health outcome during these types of activities; and that factors related to the risk of illness include the quality of the water and the frequencies of immersion and water ingestion.

Several reviews of the epidemiological findings have also been published. In 1998, WHO (Pruss, 1998) published a comprehensive review of the epidemiological research that had been conducted over the period from 1953 to 1996. This was the first extensive review of the existing epidemiological literature. Pruss (1998) concluded that gastrointestinal illness was the most frequent health outcome for which dose–response relationships were reported and that the indicators that best correlated with health outcomes were enterococci for marine waters and *E. coli* and enterococci for fresh waters.

The U.S. EPA published two reviews of the existing epidemiological literature for recreational waters. The first, published in its *Implementation Guidance for Ambient Water Quality Criteria for Bacteria* (U.S. EPA, 2002), was a minireview of the epidemiological studies conducted since the previous guidance had been issued, in 1986. The U.S. EPA concluded that the epidemiological methods used to derive its 1986 Water Quality Criteria remained scientifically valid and that no new scientific principles had been established that would require a revision of the current guidelines. Subsequent to this, the U.S. EPA (Wade et al., 2003) conducted a meta-analysis of the existing epidemiological data available in the literature to determine if its current regulatory standards were sufficiently protective against the risk of gastrointestinal illness in recreational waters. The authors demonstrated that in the freshwater studies, *E. coli* was shown to be the most suitable indicator of recreational water illness. It was further reported that when comparing the summary relative risk values with the U.S. EPA guidelines for fresh water, *E. coli* densities above the guideline values were associated with an increased risk of illness, whereas exposures below the guideline values did not show an association.

Wiedenmann et al. (2006) reported on the results of a randomized controlled prospective cohort study conducted at freshwater swimming sites in Germany. Earlier randomized controlled trials had been conducted in coastal waters of the United Kingdom (Kay et al., 1994; Fleisher et al., 1996); however, this study was the first of its type to be conducted in fresh waters. The study design used by the authors was similar to that originally used in the UK trials. The authors observed evidence of a relationship between the observed rates of illness and measured concentrations of *E. coli*, enterococci, *Clostridium perfringens* and somatic coliphages. No-observed-adverse-effect-levels (NOAELs) were reported for several definitions of gastroenteritis, ranging from 78 to 180 *E. coli*/100 mL and from 21 to 24 enterococci/100 mL. The authors proposed possible guidelines by combining all of the data derived from the different definitions of gastrointestinal illness investigated, suggesting values of 100 *E. coli*/100 mL, 25 enterococci/100 mL, 10 somatic coliphages/100 mL and 10 *C. perfringens*/100 mL. Although the authors propose a value of 100 *E.coli*/100 mL, it is important to note that the NOAEL reported for gastrointestinal illness that most closely fits the criteria of “highly credible gastrointestinal illness” (as defined by Cabelli et al , 1983) was 180 *E. coli*/100 mL. In addition, the quartile and quintile breakdown of the data for the UK definition of gastrointestinal illness indicated that the rates of swimmer illness compared to those of the control group were not statistically significant until *E. coli* concentration ranges approached or exceeded 200 *E. coli*/100 mL. Even using the

least stringent definition of gastrointestinal illness resulted in a NOAEL (NL-2[1]: 164 *E. coli*/100 mL) well above 100 *E. coli*/100 mL.

The U.S. EPA and Centers for Disease Control and Prevention (CDC) have also been conducting epidemiological studies at freshwater and marine beaches under the National Epidemiologic and Environmental Assessment of Recreational (NEEAR) Water Study. The studies are intended to support the development of new recreational water quality guidelines (U.S. EPA, 2002; Dufour et al., 2003) as well as to investigate new water quality indicators and rapid methods for water quality monitoring. Data collection for these studies was completed in 2010.

Summary

It has been concluded, based on all of the existing evidence, that *E. coli* remains the most suitable indicator of faecal contamination in fresh recreational waters. In summary:

1. The guideline values have been developed based on the analysis of epidemiological evidence relating *E. coli* concentrations in fresh recreational waters to the incidence of swimming-associated gastrointestinal illness observed among swimmers. The values represent risk management decisions based on the assessment of possible health risks for the recreational water user and the recognition of the significant benefits that recreational water activities provide in terms of health and enjoyment.

2. *E. coli* most closely fits the requirements of an ideal indicator of faecal contamination for fresh waters. *E. coli* are always present in faecal contamination from human and animal sources. Detection suggests faecal contamination of water and thus the possible presence of faecal pathogenic bacteria, viruses and protozoa.

3. There are limitations associated with the use of indicators in assessing the quality of recreational waters. The occurrence of faecal pathogens in recreational waters is dependent on many factors and can be variable and sporadic. The absence of *E. coli* should not be interpreted to mean that enteric pathogenic microorganisms are also absent.

4. Combining routine *E. coli* monitoring alongside actions, procedures and tools to collectively reduce the risk of swimmer exposure to faecal contamination in the recreational water environment represents the most effective approach to protecting the health of recreational water users.

[1] NL-2 – Definition of gastrointestinal illness used in the Netherlands according to Van Asperen et al., 1998.

4.1.2 Marine waters: Enterococci

Guideline values
For marine recreational waters used for primary contact activities, the guideline values are as follows:

Geometric mean concentration (minimum of five samples): ≤ 35 enterococci/100 mL Single-sample maximum concentration: ≤ 70 enterococci/100 mL

Calculation of the geometric mean concentration should be based on a minimum of five samples, collected at appropriate times and sites to provide representative information on the water quality likely to be encountered by users. Further action should be initiated if either of these guideline values is exceeded. Minimum action should consist of immediate resampling of the site (or sites). In addition, a swimming advisory may be issued should the responsible authority identify that the area is not suitable for recreational water use.

It is further advised that recreational water areas routinely used for primary contact recreation be monitored at a minimum of once per week, with increased monitoring recommended for those beaches that are highly frequented or are known to experience high user densities. Similarly, under certain scenarios, a reduction in the recommended sampling frequency may be justified. Further guidance on sampling frequency recommendations and the posting of recreational waters can be found in Part I (Management of Recreational Waters).

E. coli (Section 4.1.1) is also recognized as a useful predictor of the risk of gastrointestinal illness in marine recreational waters (Wade et al., 2003). If it can be shown that *E. coli* can adequately demonstrate the presence of faecal contamination in marine waters, then the *E. coli* maximum limit for fresh waters may be adopted. If there is any doubt, samples should be examined for both sets of indicators for extended periods to determine whether a positive relationship exists.

Guideline rationale
The guideline values have been developed based on epidemiological evidence relating enterococci concentrations in marine recreational waters to the incidence of swimming-associated gastrointestinal illness observed among swimmers. The existing epidemiological data are not sufficient to permit the estimation of the level of risk for individual exposures. Based on the U.S. EPA's regression analysis of epidemiological data (Cabelli, 1983), Health Canada has estimated that using the guideline values for the recommended indicators of faecal contamination for fresh and marine waters will correspond to a seasonal gastrointestinal illness rate of approximately 1–2% (10–20 illnesses per 1000 swimmers). In Canada, owing to geography and climate, a significantly smaller percentage of the population engages in marine recreational water activities compared with those engaging in freshwater recreation.

In determining the value for the maximum faecal indicator concentration permitted in a single sample, the U.S. EPA equations pertaining to the calculation of a single-sample limit were reviewed (U.S. EPA, 1986). The data regarding the maximum allowable indicator density at designated beach areas are consistent with applying a factor of 2 times the recommended geometric mean value results. Subsequently, a single-sample maximum concentration of 70 enterococci/100 mL is reaffirmed.

These values represent risk management decisions that have been based on a thorough assessment of the potential risks for the recreational water user. In considering both the potential health risks and the benefits of recreational water use in terms of physical activity and enjoyment, it was concluded that this is a tolerable and reasonable estimate of the risk of illness likely to be experienced by users engaged in a voluntary activity.

Evaluation of the epidemiological information published since the Guidelines were last issued concluded that the current body of evidence supports the existing recommendations regarding the use of enterococci as the indicator of faecal contamination in marine recreational waters. There has not been substantial evidence to suggest that revision of the existing guideline values is necessary at this time.

Description

Enterococci are members of the genus *Enterococcus.* The genus was created to include the more faecal-specific species of genus *Streptococcus*, formerly considered as group D streptococci. In practice, the terms enterococci, faecal streptococci, *Enterococcus* and intestinal enterococci have been used interchangeably (Bartram and Rees, 2000). Enterococci are characterized by their ability to fulfil the following criteria: growth at temperatures of 10°C and 45°C, resistance to 60°C for 30 minutes, growth in the presence of 6.5% sodium chloride and at pH 9.6, and the ability to reduce 0.1% methylene blue (Bartram and Rees, 2000; APHA et al., 2005). Species of the genus include *E. faecalis*, *E. faecium*, *E. durans*, *E. hirae*, *E. gallinarum* and *E. avium*.

E. faecalis and *E. faecium* occur in significant quantities in both human and animal faeces and, along with *E. durans*, have been reported to be the species most frequently encountered in polluted aquatic environments (Bartram and Rees, 2000). *E. gallinarum* and *E. avium* occur at high concentrations in animal faeces, but are not exclusively associated with animal sources.

Enterococci closely satisfy many characteristics of a suitable indicator of faecal contamination in recreational waters. Many species within the group enterococci are found in high numbers in human and animal faeces. They are not commonly found in unpolluted waters and are generally regarded to be incapable of growth in recreational waters (Ashbolt et al., 2001). Compared with other indicator organisms (e.g., *E. coli*, thermotolerant coliforms), enterococci have demonstrated greater resistance to certain environmental stresses in recreational waters, such as conditions of sunlight and salinity. Enterococci have also demonstrated greater resistance to wastewater treatment practices, including chlorination. A strong correlation has also been demonstrated between the concentration of enterococci in marine waters and the risk of gastrointestinal illness among swimmers (Cabelli, 1983; Kay et al., 1994).

In the past, a ratio of thermotolerant coliforms to faecal streptococci concentrations was used in attempts to indicate the origin of bacterial contamination (Geldreich, 1976; Clausen et al., 1977). A thermotolerant coliform/faecal streptococcus ratio of 4 or higher was said to indicate a human source of contamination, whereas a lower ratio would represent an animal source. However, because of the noted differences in the survival times between these two groups in the environment and the variability of the different methods used for their enumeration, the use of the thermotolerant coliform/faecal streptococcus ratio is now considered inaccurate (Ashbolt et al., 2001; APHA et al., 2005). As a result, the use of this ratio is not recommended. Further information on faecal pollution source tracking can be found in Section 10.0 (Faecal pollution source tracking).

Occurrence in the aquatic environment

Enterococci can be routinely isolated from marine and fresh recreational waters known to be impacted by human and animal faecal pollution sources. These organisms are present in high concentrations in human and animal faeces, with concentrations reported on the order of 10^6–10^7/g (Sinton, 1993; Edberg et al., 2000). Overall, it is thought that enterococci are present at concentrations approximately 1- to 3-fold lower than those of *E. coli* in faeces and municipal wastes (Sinton, 1993; Edberg et al., 2000). Human faecal flora studies reported by Leclerc et al. (2001) demonstrated that *Enterococcus* species could be detected in the faeces of 100% of the subjects tested.

Several publications have reported that prolonged survival of enterococci is possible in marine and freshwater sediments (Davies et al., 1995; Desmarais et al., 2002; Ferguson et al., 2005). These are thought to provide more favourable conditions of temperature and nutrients than the adjacent recreational waters. Others have reported on the ability of enterococci to survive in organic-rich environments not known to be associated with faecal contamination, such as on mats of the green algae species *Cladophora* (Whitman et al., 2003).

In Canada, there have been few published investigations on the distribution of enterococci in the marine environment. Gibson and Smith (1988) conducted a study to investigate the distribution of enterococci at 26 marine beaches in the Vancouver region. The findings of this study demonstrated that 1.6% of the results would have exceeded the enterococci geometric mean concentration guideline value of 35/100 mL. In 1988, the New Brunswick Department of Health and Community Services (1989) monitored eight marine beaches along the Northumberland Strait in New Brunswick. The overall enterococci levels were low, showing a geometric mean concentration of 3.5/100 mL. The results of the study indicated that enterococci were absent in 60% of the samples.

Association with pathogens

Enterococci are considered a good indicator for enteric bacterial pathogens. In a survey of surface waters collected from various watersheds in southern California, enterococci were shown to demonstrate good predictive ability with PCR detection of STEC (Yanko et al., 2004). It was reported that above an enterococci concentration of 100 most probable number (MPN)/100 mL, the probability of detection of STEC was approximately 60–70%.

Enterococci are somewhat less effective as an indicator of the presence of enteric pathogenic viruses and protozoa. A number of researchers have reported on the lack of a relationship between enterococci concentrations and the presence of human viruses in surface waters (Griffin et al., 1999; Schvoerer et al., 2000, 2001; Jiang et al., 2001, Jiang and Chu, 2004)

Enterococci are considered the best available indicator of water quality for marine recreational waters (Pruss, 1998; WHO, 1999; Wade et al., 2003). Detection indicates faecal contamination of water and thus the possible presence of faecal pathogenic bacteria, viruses and protozoa. Human enteric viral and protozoan pathogens of faecal origin can survive for prolonged periods in marine waters. Although the presence of high enterococci counts may indicate the possible presence of viral and protozoan pathogens, the opposite—that the absence of enterococci indicates that these pathogens are also absent—cannot be assured.

The combination of routine monitoring for enterococci alongside actions, procedures and tools to collectively reduce the risk of swimmer exposure to faecal contamination in the recreational water environment represents the most effective approach to protecting the health of recreational water users.

Guidelines used by other countries/organizations

The guideline formats and values established by other government and multinational organizations worldwide for faecal indicator organisms in marine waters were reviewed in developing the revised edition of this document.

Table 3. Guideline values for faecal indicator concentrations in marine recreational waters established by other countries or organizations.

Country/ organization	Marine water indicator	Format and guideline values	Reference
U.S. EPA[a]	Enterococci	Geometric mean concentration: 35/100 mL Single sample maximum concentration:[b] 104/100 mL	U.S EPA, 2002
WHO	Intestinal enterococci	95th percentile/100 mL: A: ≤40 B: 41-200 C: 201-500 D: >500	WHO, 2003a
Australia	Intestinal enterococci	95th percentile/100 mL: A: ≤40 B: 41-200 C: 201-500 D: >500	NHMRC, 2008
European Union	Intestinal enterococci	95th percentile/100 mL: Excellent: 100 /100 mL Good: 200/100 mL 90th percentile/100 mL: Sufficient: 185/100 mL	EU, 2006
	E. coli	95th percentile/100 mL: Excellent: 250 /100 mL Good 500/100 mL	

	90th percentile/100 mL: Sufficient: 500/100 mL

a – Note that new criteria is currently under development and may be available in 2012.
b – Designated beach area (75% confidence level)

Related epidemiological studies

The U.S. EPA's original epidemiological studies in marine recreational waters (Cabelli, 1983) measured the concentrations of faecal indicator organisms - total coliforms, thermotolerant (faecal) coliforms, *E. coli*, enterococci - in swimming waters and compared them with the rates of swimming-associated illness reported on the same days on which the samples were collected. Statistically significant rates of gastrointestinal illness were observed among swimmers in waters considered to be more faecally contaminated. For symptoms unrelated to gastrointestinal illness, no statistically significant differences were observed. For the data analysis, the mean seasonal faecal indicator concentration per 100 mL was plotted against the seasonal swimming associated rate of gastrointestinal symptoms per 1000 persons. Correlation and regression analysis were then used to determine the correlation coefficients and the slope of the linear regression equation for each indicator. The best correlation coefficient (r) was obtained with enterococci (r = 0.75). The following regression equation was produced for the enterococci data :

$$\text{seasonal risk of gastrointestinal illness per 1000 persons} = 0.20 + 12.17(\log \text{enterococci concentration per 100 mL}).$$

Several marine water epidemiological studies have been conducted since the *Guidelines for Canadian Recreational Water Quality* were last developed (Cheung et al., 1990; Alexander et al., 1992; von Schirnding et al., 1992; Corbett et al., 1993; Harrington et al., 1993; Kay et al., 1994; Kueh et al., 1995; Marino et al., 1995, Fleisher et al., 1996; van Dijk et al., 1996; McBride et al., 1998; Haile et al., 1999; Prieto et al., 2001). All of the studies have confirmed the existence of a relationship between exposure to marine recreational waters of varying quality and symptoms of water-related illness among swimmers. The most significant findings came from the results of the randomized controlled program of epidemiological studies conducted at coastal beaches in the United Kingdom (Kay et al., 1994; Fleisher et al., 1996). These studies were designed to address some of the perceived shortcomings of the traditional beach survey design used in many of the earlier studies. The key features of the randomized controlled design were efforts to ensure a more random distribution of subjects in the swimming and non-swimming groups and tighter monitoring of the water quality experienced by the individual swimmers. Of the faecal indicators monitored, only faecal streptococci levels measured at chest depth showed a significant relationship with the incidence of both gastrointestinal illness and respiratory illness among swimmers. The authors further reported the existence of possible thresholds for an increased risk of gastroenteritis at a concentration of 32 faecal streptococci/100 mL and an increased risk of respiratory illness at a concentration of 60 faecal streptococci/100 mL. In other studies, McBride et al. (1998) reported an increasing risk of respiratory illness with increasing enterococci levels among swimmers at New Zealand beaches. Cheung et al. (1990) observed a moderate correlation (r = 0.63) between enterococci levels and rates of highly credible gastrointestinal illness (HCGI) and skin symptoms combined at coastal beaches in Hong Kong, although a stronger correlation was observed with *E. coli* (r = 0.73).

A few epidemiological studies have been conducted that have investigated the health effects of recreational activities other than swimming, such as surfing (Gammie and Wyn-Jones, 1997; Dwight et al., 2004). The data linking water quality and illness through these activities have been less strong. Still, conclusions that have been reported are that gastrointestinal illness similarly constitutes the most frequently reported health outcome during these types of activities and that factors related to the risk of illness include the quality of the water and the frequencies of immersion and water ingestion.

Several reviews of the epidemiological findings have also been published. WHO (Pruss, 1998) published a comprehensive review of all of the recreational water epidemiological studies conducted over the period from 1953 to 1996. From the review, it was concluded that gastrointestinal symptoms were the most frequently reported outcome for which water quality dose–response relationships were reported, and that the indicator organisms that best correlated with the health outcomes were enterococci for marine waters and *E. coli* and enterococci for fresh waters. The U.S. EPA has also published two reviews of the existing epidemiological literature for recreational waters. The first, published in its *Implementation Guidance for Ambient Water Quality Criteria for Bacteria* (U.S. EPA, 2002), was a minireview of the epidemiological studies conducted since the previous guidance had been issued, in 1986. The U.S. EPA concluded that the epidemiological methods used to derive its 1986 Water Quality Criteria remained scientifically valid and that no new scientific principles had been established that would require a revision of the current guidelines. More recently, Wade et al. (2003) conducted a meta-analysis of all of the existing epidemiological data published since 1950 linking microbiological indicators of recreational water quality to gastrointestinal illness in swimmers. The authors concluded that in the marine water studies, enterococci and, to a lesser extent, *E. coli* were the most reliable predictors of gastrointestinal illness. Moreover, the authors observed that the reported risks of gastrointestinal illness at enterococci concentrations below the current U.S. EPA standards were not statistically significant, whereas values above the standards were elevated and statistically significant.

The U.S. EPA and CDC are currently conducting epidemiological studies at freshwater and marine beaches under the NEEAR Water Study. The studies are intended to support the development of new recreational water quality guidelines (U.S. EPA, 2002; Dufour et al., 2003), as well as to investigate new water quality indicators and rapid methods for water quality monitoring. Data collection for these studies was completed in 2010.

Summary

Based on all of the existing evidence, the enterococci group remains the most suitable indicator of faecal contamination in marine recreational waters. In summary:

1. The guideline values have been developed based on the analysis of epidemiological evidence relating enterococci concentrations in marine recreational waters to the incidence of swimming-associated gastrointestinal illness observed among swimmers. The values represent risk management decisions based on the assessment of possible health risks for the recreational water user and the recognition of the tremendous benefits that recreational water activities provide in terms of health and enjoyment.

2. Enterococci most closely fit the requirements of an ideal indicator of faecal contamination for marine recreational waters. Detection suggests faecal contamination of water and thus the possible presence of faecal pathogenic bacteria, viruses and protozoa.

3. There are limitations associated with the use of indicators in assessing the quality of recreational waters. The occurrence of faecal pathogens in recreational waters is dependent on many factors and can be variable and sporadic. The absence of enterococci should not be interpreted to mean that enteric pathogenic microorganisms are also absent.

4. Combining routine enterococci monitoring alongside actions, procedures and tools to collectively reduce the risk of swimmer exposure to faecal contamination in the recreational water environment represents the most effective approach to protecting the health of recreational water users.

4.2 Advice regarding water intended for secondary contact recreational activities

The *Guidelines for Canadian Recreational Water Quality* are intended to be protective for those activities that involve intentional or incidental immersion in natural waters. Due to increased interest from jurisdictions in distinguishing between primary contact activities and secondary contact activities, this current edition of the Guidelines takes an initial step at providing advice for secondary contact activities separately from that of primary contact with respect to faecal indicator concentration.

Recreational activities that have been traditionally considered as secondary contact activities (e.g. canoeing, fishing) involve exposures much different from those associated with primary contact uses. Ingestion of water and, subsequently, the risk of gastrointestinal illness are presumed to be lower during secondary contact recreation. Still, it is expected that there is some degree of risk of acquiring illness through these activities. Inadvertent immersion can result in whole body contact, and splashing can lead to a variety of water exposure scenarios. Illnesses affecting the skin and perhaps the mucous membranes of the eyes and ears may be of relatively greater importance for secondary contact uses (U.S. EPA, 2002). Inhalation may also be an important route of exposure during secondary contact activities in areas where sprays or aerosols are generated.

Limited research has been conducted on the potential risks of acquiring illness during secondary contact activities in recreational waters. In one study which investigated the relationship between water quality and illness acquired during canoeing or rowing, Fewtrell et al. (1994) noted no significant differences between the exposed group and the unexposed group. The bulk of the epidemiological research on recreational water uses and the risk of acquiring illness have been generated for primary contact activities. As a result, insufficient data are available to derive precise health-based faecal indicator limit values intended to protect users engaged in secondary contact recreational activities from exposure to faecal contamination. However, it is recognized that because a lower degree of water exposure can be expected at most times during the majority of secondary contact recreational activities, there may be some water areas where a secondary

contact use designation with separate water quality values is desired and considered reasonable and acceptable to local and regional authorities.

When contemplating the establishment of separate faecal indicator values for water areas used entirely for secondary contact recreational uses, a clear understanding of the types of activities that would be considered to fit under this description is required. WHO, in its *Guidelines for Safe Recreational Water Environments: Volume 1—Coastal and Fresh Waters* (WHO, 2003a), has proposed a scheme for the classification of recreational water activities according to their degree of water exposure. The following descriptions (adapted from WHO, 2003a), may be used as an initial guide when determining whether a specific recreational activity would be considered as either primary or secondary contact:

- *Primary contact*: Recreational activity in which the whole body or the face and trunk are frequently immersed or the face is frequently wetted by spray, and where it is likely that some water will be swallowed. Inadvertent immersion, through being swept into the water by a wave or slipping, would also result in whole body contact. Examples include swimming, surfing, waterskiing, whitewater canoeing/rafting/kayaking, windsurfing or subsurface diving.
- *Secondary contact*: Recreational activity in which only the limbs are regularly wetted and in which greater contact (including swallowing water) is unusual. Examples include rowing, sailing, canoe touring, or fishing.

Even if these classification criteria are used, it remains a significant challenge to discern which activities constitute primary contact and which constitute secondary contact. The classification of certain recreational water activities will be clear, whereas that of others may be less obvious and more open to interpretation. Activities considered as potential candidates under a secondary contact use designation should be evaluated on a case-by-case basis.

Other factors to consider before assigning a secondary contact use designation to a recreational water area include:

- The water area should first be subject to an assessment of existing uses, water quality and the potential for improvement as well as any other relevant factors, such as health or environmental considerations.
- The secondary contact designation should not be applied where an assessment has shown primary contact recreation to be a significant use.
- Where the water area has a shared use (e.g. swimming and canoeing), it is the primary contact values that should apply.
- When an area is posted as suitable only for secondary contact recreational uses, communication material should clearly convey that accidental immersion (through falls, canoe spills, etc.) can lead to whole body exposure; under these circumstance, water ingestion may result in illness.
- Users should be reminded to take the precautions necessary to ensure that these types of exposures are avoided as much as possible; the skill of the person performing the activity may strongly influence the degree of water exposure.
- Further guidance on the posting of information at recreational water areas can be found in Part I (Management of Recreational Waters).

Responsible authorities have a duty to take precautions, protect the health and safety of all recreational water users and maintain the best water quality possible. The existence of the secondary contact values should not be used as a mechanism for downgrading the status of an area in response to poor water quality issues. This is particularly important where an assessment has shown that the primary contact guideline values could be achieved.

Based on an assessment of the available information, where a water area is intended to be used specifically for secondary contact recreation (i.e. where primary contact is not an existing use), the application of a factor of 5 to the geometric mean faecal indicator concentration used to protect primary contact recreation users may be used as an approach to establish faecal indicator limits. The corresponding values for *E. coli* and *enterococci* concentrations would therefore be as follows:

Fresh waters – *E. coli*: (5 × 200/100 mL) = 1000 *E. coli*/100 mL
Marine waters – Enterococci: (5 × 35/100 mL) = 175 enterococci/100 mL

These values represent a risk management decision based on the assessment of the expected exposure scenarios and potential health risks for the recreational water user. They are intended to allow specified water areas to have a secondary contact use designation where this has been considered appropriate by the responsible local or regional authorities, while still providing some level of protection for secondary contact recreational users until epidemiologically based guideline values can be derived. In considering both the potential health risks and the benefits of recreational water use, it was concluded that this is a tolerable and reasonable approach to protect users engaged in a voluntary activity. These values are also consistent with advice provided by other jurisdictions (Saskatchewan Environment, 1997; Alberta Environment, 1999; U.S. EPA, 2002; British Columbia Ministry of Health Services, 2007; MDDEP, 2007). The values will be periodically reviewed or adjusted, however, as new or more significant data become available.

Insufficient information is availableto develop separate values with respect to water areas used solely for secondary-contact for other parameters in the *Guidelines for Canadian Recreational Water Quality*. Area operators, service providers and responsible authorities should remain aware that these parameters may also affect water areas intended only for secondary contact uses. Where separate guidance for secondary contact does not exist, it is advised that the values and associated guidance provided in the Guidelines apply to all recreational waters, regardless of the types of activities practised.

4.3 Other organisms as potential indicators

Many different waterborne pathogens can be encountered in Canadian recreational waters. As discussed in the previous section, recreational water quality is most frequently determined by the presence of faecal indicator bacteria which, by extension, suggests the possible existence of faecally transmitted waterborne pathogens. Currently, *E. coli* (fresh waters) and enterococci (marine waters) remain the best available indicators of recreational water quality, as they, above

all other organisms, have been shown to be the most successful in meeting the desired indicator criteria.

Nevertheless, the current indicators do not unequivocally meet all of the requirements of an ideal indicator, and the limitations of these two organisms as pathogen indicators are well known. It is understood that no single organism would be able to fill all of the roles of what might be considered a perfect indicator of recreational water quality—one that models all of the known pathogens, provides information on the degree and source of faecal contamination and communicates the potential risk of illness for recreational water users. It has been proposed that this task would require multiple indicators, each with unique characteristics that would enable them to satisfy specific roles (Ashbolt et al., 2001).

The term indicator can be further specified to reflect these different functions. Indicators may be considered as faecal indicators (indicative of the presence of faecal contamination, but not necessarily of specific pathogens) or pathogen indicators (indicative of the presence and behaviour of specific pathogens). Moreover, faecal indicators may be further categorized into what can be considered primary indicators—those that provide information on the magnitude or extent of faecal contamination— and secondary indicators—those that provide information on the source of faecal contamination.

The objective of this section is to provide a summary of what is known regarding other microorganisms that have been widely discussed among recognized experts, academics and policymakers as potential indicators for recreational water. The organisms covered are *Bacteroides* spp., *Clostridium perfringens*, F^+ RNA coliphages and bacteriophages infecting *Bacteroides fragilis*. A summary of the characteristics of the recommended indicators and the other potential indicator organisms discussed is presented in Table 4.

Potential indicators

Bacteroides spp.

Bacteroides spp. are Gram-negative, rod-shaped, obligate anaerobic bacteria. The genus is considered to represent the most abundant bacterial genus in human faeces (Fiksdal et al., 1985). The four dominant species—*B. fragilis*, *B. vulgatus*, *B. distasonis* and *B. thetaiotaomicron*—can reach concentrations on the order of 10^{10} cells/g faeces (Kator and Rhodes, 1994), outnumbering *E. coli* concentrations by as much as 100- to 1000-fold (Slanetz and Bartley, 1957; Holdeman et al., 1976; Fiksdal et al., 1985). *Bacteroides* species can occur in much lower densities (10^5- fold to 10^{10}-fold lower) in animals (Allsop and Stickler, 1985; Kator and Rhodes, 1994), although higher densities have been observed in some species, such as domestic pets and gulls (10^7–10^8 cfu/g) (Allsop and Stickler, 1985).

Because of their high numbers in human faeces, *Bacteroides* species have long been considered a candidate indicator of faecal pollution; however, the difficulties associated with culturing anaerobic bacteria discouraged their use in investigations (Kreader, 1995; Bernhard and Field, 2000a). Recent advances in molecular biology have overcome this problem. Researchers have developed PCR assays for the detection of both generic and species-specific (human, bovine) *Bacteroides* genetic markers in faeces. In this case, the presence of the genetic markers is considered to infer the presence of *Bacteroides* cells.

Bacteroides PCR methods have been successfully used for the detection of faecal pollution in contaminated water samples (Kreader, 1998; Bernhard and Field, 2000b; Field et al., 2003). As well, quantitative PCR (QPCR) methods have been developed that have enabled the near real-time enumeration of *Bacteroides* spp. in recreational waters (Fung, 2004; Seurinck et al., 2005; Wade et al., 2006). The U.S. EPA included QPCR monitoring for *Bacteroides* as part of its NEEAR Waters Study (Wade et al., 2006).

Detection of *Bacteroides* genetic markers in recreational waters is a relatively new area of research. Few studies have been conducted to date that provide an analysis of *Bacteroides* markers relative to faecal indicator organisms, faecal pathogens or rates of illness among swimmers. Wade et al. (2006) reported a positive but weak association between *Bacteroides* and gastrointestinal illness among swimmers at one of two freshwater beaches investigated during the NEEAR Waters Study. A problem with the sensitivity of the QPCR method was cited by the authors (Wade et al., 2006).

Noted strengths of *Bacteroides* as a possible primary indicator of faecal contamination include their high occurrence in human faeces and sewage, the inability of the bacterium to grow in the environment and the long environmental persistence of the DNA markers. Limitations include the lower concentrations observed in non-human faecal sources, current data gaps regarding its capabilities as an indicator (primary indicator, pathogen indicator and indicator of swimming-associated illness) and challenges relating to the analytical methods (expensive, technically demanding, issues of sensitivity).

Information generated to date suggests that *Bacteroides* markers may have a stronger role as a secondary indicator of faecal contamination, providing information on the potential sources of faecal material.

Clostridium perfringens

C. perfringens is a Gram-positive, rod-shaped, spore-forming anaerobic bacterium that is consistently found in both human and non-human faeces (Bisson and Cabelli, 1980). *Clostridium* species have the ability to enter into a protective spore form that is able to resist environmental stresses and that can allow the organism to persist in the environment for long periods of time.

C. perfringens has long been considered a valuable indicator of the sanitary quality of water, dating back to the late 19th century (Ashbolt et al., 2001). Much of the interest in the use of *C. perfringens* as an indicator of recreational water quality has stemmed from research conducted in the state of Hawaii. Researchers there had observed that many stream and soil samples not thought to be affected by a known source of faecal pollution contained faecal coliforms and *E. coli* in excess of the current water quality standards (Fujioka and Shizumura, 1985). It was also observed that *C. perfringens* concentrations in streams receiving wastewater discharges were consistently higher than in streams that were not affected (Fujioka and Shizumura, 1985). The researchers subsequently asserted that *C. perfringens* was a more reliable indicator of faecal contamination in Hawaiian waters. It has since been proposed that this situation may be encountered in other tropical locations in the United States (U.S. EPA, 2001b). At present,

Hawaii is the only known jurisdiction that includes monitoring for *C. perfringens* as a recreational water quality indicator (Anon., 1996).

The concentration of *C. perfringens* in human and animal faeces is considerably less than that of *E. coli* or enterococci (Wright, 1982). Published data suggest that *C. perfringens* may be detected in only a small to moderate percentage of human faecal samples (13–35%), with an average faecal concentration of approximately 10^3 cells/g (Ashbolt et al., 2001). Higher concentrations of *C. perfringens* have been reported in sewage (Fujioka and Shizumura, 1985). *C. perfringens* has been detected in the faeces of a wide range of animals, including birds, mammals, reptiles and amphibians (Conboy and Goss, 2003). Significant numbers of the organism have been encountered in the faeces of a few animal species, including dogs (10^8 cells/g), cats (10^7 cells/g) and sheep (10^5 cells/g) (Ashbolt et al., 2001). *C. perfringens* is not exclusively associated with faecal wastes and is a common inhabitant of soils (Toranzos, 1991).

Water quality investigations conducted in coastal waters in Florida have indicated that concentrations of *C. perfringens* do not correlate well with levels of faecal indicator bacteria (Griffin et al., 1999; Lipp et al., 2001) or with the presence of enteric viruses (Griffin et al., 1999). Furthermore, the levels of *C. perfringens* were generally lower than those of enterococci or the faecal coliform group in the water column, but were substantially greater than the levels of either indicator in the underlying sediment (Lipp et al., 2001). In a study of indicator and pathogen presence in selected lakes and rivers in southwestern Finland, Hörman et al. (2004) reported a positive correlation between the presence of *C. perfringens* and the detection of one or more of the pathogens being tested for (*Cryptosporidium*, *Giardia*, *Campylobacter*, noroviruses). However, the absence of *C. perfringens* demonstrated only a weak predictive value for a negative pathogen sample.

Several epidemiological studies have included *C. perfringens* as an indicator when investigating the relationships between water quality and illness in swimmers (Cabelli, 1983; Harrington et al., 1993; Kueh et al., 1995; Lee et al., 1997; Wiedenmann et al., 2006). Cabelli (1983) observed a weak correlation between *C. perfringens* densities and acute gastrointestinal illness in swimmers during the U.S. EPA's original epidemiological studies at marine beaches in the 1970s. Kueh et al. (1995) noted a positive, but not strong, correlation with the incidence of gastroenteritis among swimmers and *C. perfringens* in two marine beaches in Hong Kong. Wiedenmann et al. (2006) did report a relationship between *C. perfringens* and the incidence of gastroenteritis among swimmers during a randomized controlled epidemiological study at German freshwater beaches. A NOAEL of 13 *C. perfringens*/100 mL was reported for several definitions of gastrointestinal illness.

Strengths of *C. perfringens* include its inability to grow in the environment and its capability for surviving longer than faecal waterborne pathogens. Advances in culture methods (Adcock and Saint, 2001) have helped to improve the ease with which *C. perfringens* can be detected—previously an impediment surrounding the use of this organism as a water quality indicator.

Limitations include not being faecal specific, having lower faecal numbers relative to other indicator bacteria; detection being strongly dependent on the source of contamination; presence not necessarily indicative of recent contamination (owing to the long environmental persistence

of the spores); and a lack of epidemiological evidence linking *C. perfringens* concentrations to the potential for acquiring swimming-associated illness.

C. perfringens may be better suited as an indicator of the effectiveness of drinking water treatment processes (Bisson and Cabelli, 1980; Payment and Franco, 1993) or as an indicator of intermittent or cumulative sewage inputs (Sorensen et al., 1989; Hill et al., 1993; Lisle et al., 2004). At present, *C. perfringens* appears to more closely satisfy the role of a pathogen indicator or perhaps a secondary indicator of faecal contamination.

F^+ RNA coliphages

Coliphages are bacteriophages (viruses infecting only bacteria) that specifically infect *E. coli* cells. The reasons cited for investigating coliphages as potential indicators of faecal contamination are that coliphages are thought to more closely resemble enteric viruses in terms of their physical characteristics, environmental persistence and resistance to disinfection, compared with the traditional bacterial indicators of faecal contamination. They are also less costly and easier to enumerate than human viruses. Also, because they theoretically infect only *E. coli* cells, it is thought that their detection should be sufficiently indicative of the presence of faecal contamination.

There are two main types of coliphages: somatic coliphages and male-specific (F^+) coliphages. Somatic coliphages infect *E. coli* cells by attaching to the lipopolysaccharide component of the cells' outer membranes. They have been investigated as potential indicators of swimming water quality (Contreras-Coll et al., 2002; Vantarakis et al., 2005; Wiedenmann et al., 2006); however, they are thought to represent a less specific group than the F^+ coliphages, and knowledge of their sources and behaviour is currently lacking. By comparison, F^+ coliphages have been more extensively studied (Duran et al., 2002).

F^+ coliphages demonstrate a greater specificity than somatic coliphages, infecting *E. coli* cells possessing F-pili—tube-like structures coded for by an F-plasmid and that allow connections to form between cells for the transfer of genetic material (Singleton and Sainsbury, 1997; Scott et al., 2002). It is these F-pili that serve as the site of phage attachment.

F^+ coliphages include F^+ RNA phages and F^+ DNA phages. F^+ RNA phages more closely resemble the human viruses of waterborne significance and have thus been preferentially explored (Sobsey, 2002). Using immunological or genetic methods, the group can be further categorized into four distinct serogroups or genogroups, and subsequent source tracking investigations have identified that the presence of a particular subgroup has some merit in distinguishing the source of faecal contamination (Havelaar et al., 1990; Brion et al., 2002; Schaper et al., 2002b; Cole et al., 2003). In general, Groups II and III have been shown to be highly associated with human faecal contamination (e.g., domestic or municipal sewage), Group IV has been shown to be predominantly linked to animal faecal contamination and animal waste materials, Group I has been isolated from both human and animal faecal material and wastes (Scott et al., 2002; Sobsey, 2002).

F^+ RNA phages are not always present in human faeces and, when detected, are often present in low numbers (Havelaar and Pot-Hogeboom, 1988; Havelaar et al., 1990; Luther and Fujioka,

2004). Researchers have similarly reported low isolation frequencies among septage samples or waters known to be affected by septic system wastes (Calci et al., 1998; Griffin et al., 1999). A low isolation frequency among animal faecal samples has also been reported (Calci et al., 1998; Luther and Fujioka, 2004). Significantly higher numbers of F^+ RNA coliphages have been detected in sewage and wastewater (Contreras-Coll et al., 2002; Lucena et al., 2003).

Sinton et al. (1999) reported that the degree of survival for various indicator organisms in sewage-polluted seawater during sunlight inactivation experiments (simulated summer conditions) was (from greatest to least): somatic coliphages > F^+ RNA phages > enterococci > *E. coli*. The F^+ RNA coliphage groups have been shown to vary markedly in their ability to persist in the environment (Brion et al., 2002; Schaper et al., 2002a; Sobsey, 2002; Long and Sobsey, 2004). In general, it has been observed that Group I is capable of the longest environmental persistence, followed by Groups II and III, with Group IV demonstrating the shortest period of survival (Brion et al., 2002; Schaper et al., 2002a; Long and Sobsey, 2004).

Evidence as to whether F^+ RNA coliphages are a reliable indicator of faecal pollution in natural waters has been conflicting. Havelaar et al. (1993) and Ballester et al. (2005) reported that F^+ RNA coliphage concentrations were more strongly correlated with concentrations of infectious enteroviruses and enteric viruses than either faecal coliforms or enterococci in environmental waters. The authors did report, however, that in a few instances, viruses were isolated in the absence of the coliphages, and that the reverse was also true (Havelaar et al., 1993). Griffin et al. (1999) demonstrated a lack of predictive ability when using coliphages (somatic and F^+ RNA) as indicators of the presence of enteric viruses in canal waters in the Florida Keys. Jiang and Chu (2004) reported no apparent relationship between the detection of adenovirus, enterovirus and hepatitis A virus (HAV) genomes and F-specific coliphage concentrations in a study of human viral contamination in river and coastal waters in southern California.

Grabow (2001) suggested that a direct correlation between the number of coliphages and enteric viruses in water environments cannot be expected, as coliphages are excreted at all times by a certain percentage of the human population, whereas enteric viruses are largely excreted during infection, which can be intermittent and seasonal.

There has been some information collected regarding the relationship between coliphage concentrations and the rate of swimming-associated illness. Lee et al. (1997) reported a significant association between the concentration of F^+ RNA coliphages and the reporting of gastrointestinal symptoms among canoeists and rafters at an artificial freshwater canoe course in the United Kingdom. Other researchers have included coliphages among the panel of prospective indicators monitored during epidemiological investigations (von Schirnding et al., 1992; Marino et al., 1995; McBride et al., 1998; van Asperen et al., 1998); however, no significant correlations were identified.

Strengths of F^+ RNA phages as a possible faecal indicator include the noted similarities to human enteric viruses, strong evidence of being exclusively associated with human and animal faecal material, an apparent inability to replicate in the environment and potential applications in faecal source identification. Limitations include presence being dependent on the source of

contamination, different survival rates of the individual phage groups and a lack of a demonstrated relationship with either enteric virus presence or rates of swimming-associated illness.

F^+ RNA coliphages appear to be more useful as an indicator of sewage, rather than of faecal contamination in general. At present, they appear to be better positioned as potential pathogen indicators or secondary indicators, as opposed to primary indicators of faecal contamination (Chapron et al., 2000).

Bacteriophages of *Bacteroides fragilis*

Bacteriophages of *B. fragilis* have been investigated as a possible indicator of faecal contamination on the theory that the organism might combine some of the desirable properties reported for both the coliphage group and *Bacteroides* spp.—the potential to be present in high numbers in faecal material, while demonstrating survival traits more representative of enteric viruses.

Although concentrations of *B. fragilis* in human faeces are high, *B. fragilis* phages have been shown to be isolated somewhat infrequently in faeces, and in lower numbers (Tartera and Jofre, 1987; Grabow et al., 1995). Published accounts have placed the percentage of human faecal samples from which *B. fragilis* phages have been isolated in the range from 10% to 28% (Tartera and Jofre, 1987; Grabow et al., 1995; Puig et al., 1999; Gantzer et al., 2002). Phage isolation from human and animal faeces has been shown to be largely dependent upon the *B. fragilis* host strain used for recovery (Tartera and Jofre, 1987; Puig et al., 1999). Molecular methods for the detection of *B. fragilis* bacteriophages are under investigation, and these are expected to eliminate many of the difficulties associated with the current host recovery methods (Puig et al., 2000).

B. fragilis phages have been shown to be routinely isolated from sewage; however, the concentrations encountered (< 10–10^5 phages/100 mL) are frequently less than those reported for somatic and F^+ RNA coliphages (Puig et al., 1999; Contreras-Coll et al., 2002; Lucena et al., 2003). Tartera and Jofre (1987) reported that *B. fragilis* phages could be detected in all water and sediment samples from highly polluted rivers (10^1–10^5 plaque-forming units [pfu]/100 mL), but not in samples collected from areas not known to receive sewage pollution. In a more recent investigation of the occurrence and levels of phages of *B. fragilis* in swimming waters throughout Europe, Contreras-Coll et al. (2002) documented a median phage concentration well below 10 pfu/100 mL, with 95% of the samples below 10^2 pfu/100 mL. Lucena et al. (2003) reported similar numbers during a survey of water samples collected from selected rivers in Europe and South America.

No correlation has been shown between the concentration of faecal indicator bacteria and levels of *B. fragilis* phages in recreational waters (Tartera et al., 1989; Contreras-Coll et al., 2002; Lucena et al., 2003). Certain studies have suggested that *B. fragilis* phages are reliable indicators of viral contamination in treated wastewaters (Gantzer et al., 1998) and shellfish (Hernroth et al., 2002; Formiga-Cruz et al., 2003). However, there has been limited information published to date in which the occurrence and concentrations of enteric viruses and phages of *B. fragilis* in recreational waters have been directly compared. In a study of river water samples where

domestic sewage was cited as its primary contamination source, Tartera et al. (1989) reported that *B. fragilis* phages could be consistently isolated from samples in which enteroviruses were detected.

Of the organisms suggested as possible alternative indicators of faecal pollution, *B. fragilis* phages have been perhaps the least well investigated. Potentially useful properties include the absence of significant non-faecal sources, the inability to replicate in the environment and structural similarities to the enteric viruses. Limitations include low faecal concentrations, variable isolation depending on the contamination source and difficulties associated with recovery of the organisms.

Currently, it is speculated that, similar to the F^+ coliphages, *B. fragilis* phages may be more suitable as indicators of sewage contamination—and thus as possible secondary indicators of faecal contamination.

Summary

1. The faecal contamination of recreational waters and the associated risk for water users are a complex issue. *E. coli* and enterococci are considered the best available indicators of faecal contamination in recreational waters; however, no single organism is capable of satisfying all of the roles of an ideal indicator. Multiple indicators may be required for a more complete understanding of this subject.

2. The organisms most widely discussed as other potential recreational water indicators include *Bacteroides* spp., *C. perfringens*, F^+ RNA coliphages and bacteriophages infecting *B. fragilis* (see Table 4).

3. At present, none of the proposed indicators investigated meet a sufficient number of the requirements necessary to be successful as a routine indicator of recreational water quality. None of these organisms has demonstrated a consistent correlation with the presence of waterborne pathogens in recreational waters, nor is there evidence of a strong epidemiological link between the occurrence of these organisms and the incidence of illness among recreational water users.

4. Nevertheless, these organisms do possess certain unique properties that might enable them to fulfil other roles as recreational water indicators. Currently, these organisms appear to be better suited as possible pathogen indicators or as faecal source indicators. Advances in detection and enumeration methods may help improve our understanding of these organisms and the roles they may play in recreational water monitoring programs in the future.

Table 4. Characteristics of recommended and potential indicator organisms

	Indicator organism					
Criteria	*E. coli*	Enterococci	*C. perfringens*	*Bacteroides* spp. (genetic markers)	F^+ RNA coliphages	*B. fragilis* phages
Found within the intestinal tract of humans and warm-blooded animals.	Present in high numbers in human and animal faeces.	Present in high numbers in human and animal faeces.	Low numbers in human and animal faeces (high in certain animal species). Higher in sewage.	Very high numbers in human faeces. Low to high numbers in animal faeces (species dependent).	Low numbers and variable isolation among human and animal faeces. Higher in sewage.	Low numbers and variable isolation among human and animal faeces. Higher in sewage.
Present in faeces-contaminated waters when enteric pathogens present, and in greater numbers.	Good indicator of all sources of faecal contamination. Typically present in faecal material in higher concentrations than pathogens.	Good indicator of all sources of faecal contamination. Typically present in faecal material in higher concentrations than pathogens.	Dependent upon source. Recovery difficult at low levels of contamination.	Dependent upon source. Insufficient data on correlation with pathogens.	Dependent upon source. Recovery difficult at low levels of contamination.	Dependent upon source. Recovery difficult at low levels of contamination.
Incapable of growth in the aquatic environment.	Generally regarded as true. Evidence to suggest select strains may be capable of growth in soil environment if proper conditions are met.	Generally regarded as true. Evidence to suggest select strains may be capable of growth in soil environment if proper conditions are met.	Anaerobic bacteria. Unable to replicate in aquatic environment.	Anaerobic bacteria. Unable to replicate in aquatic environment.	Thought not to be capable of replication in aquatic environment. Possibility for replication in sewage.	Host bacteria are anaerobic. Thought not to be capable of replication in aquatic environment.
Capable of surviving longer than pathogens.	Considered a good indicator of survival of pathogenic enteric bacteria, but not enteric viruses or protozoa.	Considered a good indicator of survival of pathogenic enteric bacteria, but not enteric viruses or protozoa.	Spores show extreme environmental persistence. Capable of surviving longer than waterborne pathogens.	Insufficient data on survival compared with pathogens. DNA markers show long environmental persistence.	Thought to be a good model for the survival of enteric viruses. Phage types show variable environmental persistence.	Thought to be good models for the survival of enteric viruses.
Applicable to all types of water (fresh, estuarine and marine).	Yes. Shorter survival time demonstrated in marine waters.	Yes. Demonstrates similar survival rates in fresh and marine waters.	Yes. Detection demonstrated in fresh and marine waters.	Yes. Detection demonstrated in fresh and marine waters.	Yes. Detection demonstrated in fresh and marine waters.	Yes. Detection demonstrated in fresh and marine waters.

	Indicator organism					
Criteria	*E. coli*	Enterococci	*C. perfringens*	*Bacteroides* spp. (genetic markers)	F^+ RNA coliphages	*B. fragilis* phages
Absent from non-polluted waters and exclusively associated with animal and human faeces.	Generally regarded as true. Some evidence of survival in organic-rich environments not associated with faecal contamination.	Generally regarded as true. Some evidence of survival in organic-rich environments not associated with faecal contamination.	No. Spores capable of persisting in soils and aquatic sediments.	Insufficient data. DNA markers show long persistence and may not be indicative of recent faecal contamination.	Yes. No significant non-faecal sources recognized.	Yes. No significant non-faecal sources recognized.
Density directly correlated with the degree of faecal contamination.	Generally regarded as true.	Generally regarded as true.	No. Dependent upon source.	No. Dependent upon source.	No. Dependent upon source.	No. Dependent upon source.
Density quantitatively related to swimmer-associated illnesses.	Yes. Body of epidemiological evidence demonstrating indicator shows best correlation with health outcomes for fresh waters.	Yes. Body of epidemiological evidence demonstrating indicator shows best correlation with health outcomes for marine waters and good correlation for fresh waters.	No. Strong correlation with illness not demonstrated in epidemiological investigations.	Insufficient data.	No. Strong correlation with illness not demonstrated in epidemiological investigations.	Insufficient data.
Detection and enumeration methods rapid, easy to perform, inexpensive, specific and sensitive.	Yes. Culture-based methods inexpensive, easy to perform, relatively rapid (24 hours), specific and sensitive.	Yes. Culture-based methods inexpensive, easy to perform, relatively rapid (24 hours), specific and sensitive.	Yes. Culture-based methods inexpensive, easy to perform, relatively rapid (24 hours), specific and sensitive.	No. Molecular methods of detection rapid, but technically challenging and expensive. Sensitivity also an issue.	No. Expensive and labour-intensive recovery methods.	No. Complex methodology.
Currently suggested role:	Primary indicator of faecal contamination.	Primary indicator of faecal contamination.	Pathogen indicator; secondary indicator of faecal contamination.	Secondary indicator of faecal contamination.	Pathogen indicator; secondary indicator of faecal contamination.	Pathogen indicator; secondary indicator of faecal contamination.

5.0 Pathogenic microorganisms

Guideline values
No guideline values can be established for waterborne pathogenic microorganisms in recreational waters. Testing for their presence in waters used for recreation should be performed only when there is epidemiological or other evidence to suggest that this is necessary.

Background
There are three main types of pathogenic microorganisms that can be found in recreational waters: bacteria, viruses and protozoa. Many occur as a result of contamination from human or animal wastes, whereas some are free-living microorganisms that exist naturally in the recreational water environment.

The challenges associated with the detection of pathogenic microorganisms in recreational waters are currently too great to recommend this practice as part of a regular monitoring program. Surveillance should be undertaken only during special circumstances, such as during waterborne disease outbreak investigations.

Faecal indicators such as *E. coli* and enterococci are the best available surrogates for predicting the presence of enteric pathogenic microorganisms. The presence of these indicators is expected to indicate the possible presence of these organisms. The absence of the recommended faecal indicators, however, should not be interpreted to mean that all pathogenic microorganisms are also absent. Although it is not possible to completely eliminate the risk of waterborne disease, adopting a multi-barrier approach to recreational water management will help minimize the risk of human exposure to pathogenic microorganisms (bacteria, viruses and protozoa) in recreational waters.

Information is provided on those pathogens recognized as having significance for Canadian recreational waters. This list is not intended to be exhaustive, and responsible authorities may wish to provide information on other organisms to consider regional interests. Additional information on many of these organisms can be found in the technical documents for the *Guidelines for Canadian Drinking Water Quality*.

5.1 Bacterial pathogens

A number of pathogenic bacteria can potentially be found in Canadian recreational waters. Enteric pathogenic bacteria occur in recreational waters as a result of contamination with human or animal faecal wastes. Sources include sewage discharges, combined sewer overflows, stormwater, malfunctioning septic waste systems and infected swimmers. A number are recognized zoonotic pathogens, and thus faecal shedding by animals and stormwater runoff from areas affected by the presence of animals are also important sources. Transmission occurs via the faecal–oral route, through accidental ingestion of contaminated waters. Gastrointestinal symptoms are the most common manifestation of illness following infection with enteric bacterial pathogens. Some pathogens can cause illness with more serious outcomes. *E. coli* and enterococci are the best available indicators for predicting the possible presence of enteric pathogenic bacteria.

Other pathogenic bacteria can be free-living species or can enter natural waters through means other than faecal contamination. Transmission can occur in waters containing sufficient quantities of the organisms, typically by inhalation or via direct contact with body surfaces. The types of illness caused can be varied, ranging from respiratory illnesses to infections of the eyes, ears or skin. As these organisms are not of faecal origin, faecal indicators are not expected to correlate well with the presence of these bacteria. Currently, there is no recognized microbiological indicator for many of these pathogens.

5.1.1 Enteric pathogenic bacteria (Campylobacter, pathogenic E. coli, Salmonella, Shigella)

Campylobacter

Campylobacter species are Gram-negative, motile, non-spore-forming, spiral, curved or S-shaped rods. They are thermophilic (growing optimally at 42°C and incapable of growth below 30°C) and mesoaerophilic (surviving best under partially anaerobic conditions) organisms. The genus *Campylobacter* is composed of 15 species; however, *C. jejuni* and *C. coli* represent the major species of human concern in the water environment.

Campylobacter are predominantly considered to be zoonotic pathogens (Fricker, 2006). The organisms are harboured in the intestinal tract of a wide range of domestic and wild animals, particularly birds. Poultry is regarded as the primary avian source, and the organism has been isolated from virtually all bird species (Fricker, 2006). It is likely that a significant proportion of seagulls also carry these organisms (Moore et al., 2002; Pond, 2005). Cattle, sheep and pigs are also considered to be reservoirs.

The exact mechanisms of *Campylobacter* virulence are incompletely understood. Attachment to and invasion of the human intestinal tract are important factors in causing disease. *C. jejuni* is reported to be capable of producing a cholera-like enterotoxin that is thought to illicit the production of a profuse, watery diarrhoea in ill individuals.

Symptoms of *Campylobacter* enteritis include a profuse, watery diarrhoea (with or without blood and/or faecal leukocytes), cramps, abdominal pain, chills and fever. The average incubation period is 2–3 days but can span from 1 to 8 days (Percival et al., 2004). Illness is typically self-limited, requiring 3–7 days for recovery. Estimates of the number of cells generally required to be ingested to lead to infection have ranged from as many as 10^4 organisms to as few as 500–1000 cells (Percival et al., 2004; Pond, 2005). With pathogens in general, it is theorized that a single organism is sufficient to cause human infection. Epidemiological studies have shown, however, that the dose required is usually greater.

Certain complications (Guillain-Barré syndrome, Reiter's syndrome, appendicitis, carditis and meningitis) have been associated with *Campylobacter* enteritis; however, these are considered rare. Fatalities from *Campylobacter* infection are uncommon and have been predominantly restricted to infants, the elderly or patients afflicted with other underlying illness (Pond, 2005).

Despite the fact that *Campylobacter* spp. can be fairly widely isolated from surface waters, there have been virtually no recorded instances of *Campylobacter*-associated illness as a result of

recreational water activity in North America. The U.S. CDC reported that *Campylobacter* spp. were not implicated as the causative agents in any of the recreational water outbreaks of gastroenteritis to have been documented in the United States for the period 1992–2002 (Craun et al., 2005). Similarly, no outbreaks of campylobacteriosis have been recorded in Canadian recreational waters.

Pathogenic *E. coli*

E. coli are Gram-negative, motile, facultatively anaerobic, non-spore-forming rods that are natural inhabitants of the intestinal tract of humans and animals. The vast majority of the *E. coli* isolates are harmless; however, there are several serotypes or strains that possess virulence factors enabling them to act as human pathogens. Pathogenic enteric strains can be separated into six groupings according to their serological or virulence characteristics: enterohaemorrhagic (EHEC), enterotoxigenic (ETEC), enteroinvasive (EIEC), enteropathogenic (EPEC), enteroaggregative (EAEC) and diffuse adherent (DAEC). Human sewage is the principal source of all of the major pathogenic *E. coli* groups, with the exception of EHEC. Cattle are considered the primary reservoir for EHEC, although human wastes also remain an important source.

Of the groupings, it is the EHEC group—(also commonly referred to as verocytotoxigenic *E. coli* [VTEC] or Shiga-toxin producing *E. coli* [STEC])—that is of greatest importance to recreational waters. *E. coli* O157:H7 is the most significant serotype of this group, and has been identified as the causative agent in a number of recreational water outbreaks (Craun et al., 2005).

An important virulence factor for EHEC is this ability to produce Shiga-like toxins, similar to those produced by *Shigella dysenteriae*. EHEC infection causes haemorrhagic colitis, marked by grossly bloody diarrhoea, severe cramping and abdominal pain with a general lack of fever. The incubation period for the disease ranges from 1 to 8 days (Percival et al., 2004; Pond, 2005), with the duration of infection lasting from 1 to 12 days (Percival et al., 2004). Persons only suffering from diarrhoea typically experience a full recovery (Pond, 2005). An estimated 2–8% of all cases progress to what is known as haemolytic uraemic syndrome, or HUS—a life-threatening condition involving large-scale destruction of red blood cells and kidney failure. Children, the elderly and immunocompromised persons are at increased risk for developing HUS.

The number of EHEC cells needed to be ingested in order to lead to infection is considered to be very low. General estimates regarding the infectivity of *E. coli* O157:H7 suggest that ingestion of fewer than 100 cells may be all that is necessary (Percival et al., 2004), and that as few as 50 or even 5 organisms may be sufficient (Pond, 2005).

According to surveillance data published by the U.S. CDC for the period 1992–2002, EHEC were associated with 25% (16 of 64) of the total number of outbreaks of gastrointestinal illness reported for natural waters (Moore et al., 1993; Kramer et al., 1996; Levy et al., 1998; Barwick et al., 2000; Lee et al., 2002; Yoder et al., 2004). *E. coli* O157:H7 was the serotype implicated in 14 of 16 of those outbreaks. The remaining two outbreaks involved *E. coli* serotypes O121:H19 and O26:NM.

In August 2001, an outbreak of *E. coli* O157:H7-associated illness involving four children was linked to swimming at a public beach in Montreal (Bruneau et al., 2004). This was the first

reported incident of *E. coli* O157:H7 to be associated with recreational water activity in Canada. Weekly water samples collected around the time of the outbreak were shown to be within the recreational water quality limits specified by the Province of Quebec. It was suggested that a high swimmer population and the shallow depth of water encountered in the swimming area contributed to the transmission of the organisms. To date, there have been no reported fatalities resulting from infection with pathogenic *E. coli* acquired through recreational contact with natural waters in either the United States or Canada.

Salmonella

Salmonella are members of the family Enterobacteriaceae. They are Gram-negative, facultatively anaerobic, motile, non-spore-forming rods. The taxonomy of the genus *Salmonella* is quite complex. Currently, there are over 2500 known *Salmonella* serotypes (or serovars) (Lightfoot, 2004). The genus is officially thought to comprise two species: *S. enterica* and *S. bongori* (Percival et al., 2004). *S. enterica* can be further subdivided into six subspecies (*S. enterica* subsp.): *enterica*, *salmae*, *arizonae*, *diarizonae*, *houtenae* and *indica*. The majority of the serotypes encountered in cases of human gastroenteritis belong to the subspecies *S. enterica* subsp. *enterica* (Lightfoot, 2004). Owing to the complexity of the nomenclature, by convention, when referring to *Salmonella*, the serotype is adopted in place of the species name. Thus, *S.* Enteritidis takes the place of *S. enterica* subsp. *enterica* serovar *enteritidis*.

With the exception of the typhoidal species (*S.* Typhi, *S.* Paratyphi), *Salmonella* are considered zoonotic pathogens. Reservoirs for non-typhoidal *Salmonella* species include poultry, swine, birds, cattle, rodents, tortoises and turtles, dogs and cats (Percival et al., 2004). Humans recovering from illness can also provide a source of *Salmonella*, and asymptomatic infections among humans are also possible. In contrast, humans are considered the primary source of *S.* Typhi and *S.* Paratyphi. Occurrence of these isolates in animal hosts or in the natural environment is rare, particularly in Canada.

Gastroenteritis represents by far the most commonly encountered type of *Salmonella*-associated illness. The chief symptoms are mild to severe diarrhoea, nausea and vomiting. Symptoms usually appear between 12 and 48 hours from the time of infection, but this time lag may be reduced in cases where large quantities of cells have been consumed (Percival et al., 2004). Illness is generally mild and self-limiting, lasting 2–5 days on average. Reports on the infectivity of *Salmonella* have suggested that the median dose for the non-typhoidal species may be as low as 1000 cells, and possibly below 10 cells (Hunter, 1997; Pond, 2005).

Enteric fever (typhoid or paratyphoid fever) is a more severe and often fatal form of salmonellosis caused by *S.* Typhi and *S.* Paratyphi. Symptoms of the illness are prolonged fever, diarrhoea and abdominal pain; progression to septicaemia may also occur. Waterborne outbreaks of enteric fever are more prevalent in developing countries where crowded living conditions and poor hygienic practices exist and are often associated with improperly treated drinking water supplies. Cases are rare in North America.

Septicaemia represents the condition in which the infecting bacteria have invaded the bloodstream and is accompanied by visible symptoms, such as high, remittent fever. Fatal damage to the liver, spleen, respiratory or neurological functions may occur when the organism

has spread to these organs or systems. Septicaemia from non-typhoidal species is uncommon (Pond, 2005).

Although *Salmonella* can be fairly widely isolated from surface waters, there have been no recorded outbreaks from *Salmonella* as a result of recreational water activity in North America. U.S. CDC surveillance data for the years 1992–2002 indicated that *Salmonella* was not cited as a causative agent for any of the waterborne outbreaks of gastroenteritis reported over that period. Surveillance in Canada has been somewhat limited; however, similarly, there have been no documented outbreaks from *Salmonella* in Canadian recreational waters.

Shigella

Shigella species are members of the family Enterobacteriaceae and, as such, possess many of the same characteristics as *E. coli*. Both are Gram-negative, facultatively anaerobic, non-spore-forming rods. Unlike the majority of *E. coli* isolates, however, *Shigella* species do possess certain attributes that make them important human pathogens. The genus *Shigella* is composed of four species: *S. sonnei* (1 serotype), *S. flexneri* (6 serotypes), *S. boydii* (15 serotypes) and *S. dysenteriae* (10 known serotypes). Two species, *S. sonnei* and *S. flexneri*, account for the vast majority of *Shigella*-associated illness in North America (CDC, 2005a). Other *Shigella* species are uncommon, but remain important causes of disease in developing countries (CDC, 2005a).

Humans are considered to be the only important reservoir for *Shigella* (Percival et al., 2004). Non-infected individuals are not expected to harbour the organism. Recovering individuals may continue to shed significant numbers of the bacterium for several weeks after symptoms have resolved, and asymptomatic carriage of *Shigella* is also possible. Municipal sewage discharges present an obvious source of *Shigella*; however, faecal shedding by infected swimmers may be the most significant source of the organism in recreational waters (Kramer et al., 1996; Levy et al., 1998).

Shigella causes what had been historically referred to as bacillary dysentery (invasion of the intestinal tract, causing frequent passage of stools containing blood and mucus). Illness (shigellosis) is caused by invasion and colonization of the intestinal tract, which leads to inflammation and the destruction of intestinal epithelial cells. *Shigella* species are also capable of producing a heat-labile enterotoxin; however, its role is not completely understood (Percival et al., 2004).

Shigellosis is characterized by watery or bloody diarrhoea, abdominal pain and fever. Symptoms often present within 1–3 days post-infection, but may be evident in as little as 12 hours. The severity of illness is strongly dependent on the virulence of the individual species or strain. *S. sonnei* infections are thought to be both of shorter duration and somewhat milder than those caused by *S. flexneri* (Percival et al., 2004). In many developing countries, *S. dysenteriae* is known to be responsible for causing severe epidemics. *S. dysenteriae* serotype 1 is also known to be capable of producing what is referred to as Shiga toxin—a unique toxin (separate from the general *Shigella* enterotoxin) that can be extremely damaging to human intestinal and kidney endothelial cells. For *S. sonnei* or *S. flexneri*, an inoculum of 100 cells may be sufficient to produce infection, whereas *S. dysenteriae* may require the ingestion of as few as 10 cells to initiate disease (Pond, 2005).

In North America, most cases of shigellosis are mild and self-limiting. Infection typically does not spread beyond the intestinal tract. Complications such as Reiter's syndrome and HUS (following infection with *S. dysenteriae* serotype 1) have been reported, but are uncommon. Similarly, mortality is rare, but higher incidences may be observed among the elderly and in undernourished children (Pond, 2005).

According to CDC surveillance data, *Shigella* accounted for approximately 22% (14 of 64) of the total number of outbreaks of gastrointestinal illness reported for natural recreational waters in the United States over the period 1992–2002 (Moore et al., 1993; Kramer et al., 1996; Levy et al., 1998; Barwick et al., 2000; Lee et al., 2002; Yoder et al., 2004). *S. sonnei* was implicated as the causative agent in all but one of these incidences. Recreational lakes provided the setting for the majority of the outbreaks reported, with poor water circulation frequently cited as a contributing factor. It was suspected that faecal contamination by other swimmers was the cause in the majority of these instances. Given the high infectivity reported for *Shigella*, it is thought that accidental swallowing of water containing relatively low concentrations of these organisms may be sufficient to cause illness.

Although surveillance has been somewhat limited, to date there have been no reported incidences of *Shigella*-associated illness as a result of recreational water activity in Canadian waters.

5.1.2 Free-living pathogenic bacteria (Aeromonas, Legionella, Pseudomonas, Mycobacterium)

Aeromonas

Aeromonas are Gram-negative, facultatively anaerobic, variably motile, oxidase-positive, rod-shaped to coccoid-like bacteria. They are thought to share many morphological and biochemical characteristics with members of the Enterobacteriaceae family, which includes *E. coli.*

Currently, the genus *Aeromonas* is thought to consist of 17 unique genospecies and 14 unique phenospecies (Moyer, 2006; U.S. EPA, 2006a). The genus may be subdivided into two major groups: the motile mesophilic species, which grow at temperatures between 15 and 38°C and have been associated with human infections, and the non-motile psychrophilic species, which can grow at temperatures below 15°C and are fish pathogens. At present, six species (*A. hydrophila*, *A. caviae*, *A. sobria*, *A. veronii*, *A jandaei*, *A. trota* and *A. schubertii*) are recognized as being pathogenic to humans (U.S. EPA, 2006a).

Aeromonas species are natural inhabitants of the aquatic environment. They are frequently found in fresh, marine and estuarine waters, sediments, and sewage and wastewater effluents. Aeromonads are not considered to be present in significant numbers in the faeces of healthy individuals. However, a certain percentage of individuals may carry the organisms in their intestinal tract without showing outward signs of illness.

Aeromonas are recognized animal pathogens and have been isolated from the intestinal tract of a number of animal species, including fish, reptiles, amphibians, birds and domestic livestock, with and without evidence of illness. Occurrence of the organism in recreational waters is not dependent on faecal pollution; however, the organisms are present in high numbers in sewage

and thus can be detected at significant levels in sewage-contaminated waters. Aeromonads can grow to relatively high densities in eutrophic waters (Moyer, 2006).

Aeromonas is most often associated with serious wound infections in recreational water users. Infection typically requires the existence of some sort of skin trauma, such as an open wound or following some sort of penetrating injury. Wound infections are characterized by pain, swelling, redness and fluid accumulation around the infected area. Cellulitis (severe inflammation) is frequently observed with such infections, and septicaemia is also considered a fairly common outcome (Percival et al., 2004). Other, rarer complications include necrotizing fasciitis, meningitis, pneumonia, peritonitis and endocarditis (Percival et al., 2004).

Several *Aeromonas* species (*A. hydrophila*, *A. veronii* and *A. caviae*) have been associated with human gastrointestinal illness. Gastrointestinal illness in humans exposed to contaminated water supplies has been only occasionally reported. Illness is typically mild and self-limiting, although certain strains are reported to be capable of causing a dysentery- or cholera-like illness, marked by severe abdominal cramps, vomiting, diarrhoea (including bloody stools) and fever.

The mechanisms through which *Aeromonas* is able to cause human illness are not completely understood. The organisms possess a host of virulence factors considered important for infection, colonization and evading the host's immune response. These include both cell-associated mechanisms (pili, flagella, outer membrane proteins, lipopolysaccharides and capsules) and extracellular products (toxins, proteases, haemolysins, adhesions and various hydrolytic enzymes) (U.S. EPA, 2006a).

Marino et al. (1995) reported a positive correlation between *A. hydrophila* concentrations and skin infections at two swimming beaches in Malaga, Spain. Currently, no evidence has been provided linking *Aeromonas* concentrations and the risk of acquiring swimming-associated gastroenteritis.

Despite their widespread occurrence, in North America there have been no reported outbreaks of *Aeromonas*-associated illness as a result of recreational water activities. Superficial infections with *Aeromonas* are thought to be relatively common; however, *Aeromonas* infections are not considered reportable illnesses. Thus, an estimate of the likely incidence of *Aeromonas* infections due to recreational water exposures in Canadian waters is not available.

Legionella

Legionella are Gram-negative, thermotolerant, motile, short, irregularly shaped bacteria that have strict nutrient requirements when grown on laboratory media. Over 40 species of the genus have been recognized. *L. pneumophila* (serotype 1) is the species most frequently associated with human disease (legionellosis). It is suspected that all *Legionella* species may be capable of causing illness, and approximately half of the identified species have been implicated in human disease (Hall, 2006). Other *Legionella* species frequently recovered from the environment include *L. bozemanii*, *L. longbeachae*, *L. dumoffii* and *L. gormanii*.

Legionella are naturally occurring aquatic organisms. They can be isolated from a wide range of freshwater habitats, including soils, lakes, rivers and natural thermal pools at temperatures as

high as 60°C. Marine environments typically do not provide appropriate growth conditions for these organisms. Free-living freshwater protozoa such as *Naegleria* or *Acanthamoeba* are natural hosts for these organisms. Although *Legionella* are thought to be fairly resistant to environmental stresses, survival within protozoan hosts is thought to provide an additional, significant measure of protection.

Legionella is typically encountered in low numbers in the aquatic environment, but can reach higher concentrations in sources associated with human-made water supplies, such as cooling towers, air conditioning condensers, humidifiers, hot water tanks, shower heads and whirlpool spas (Percival et al., 2004). Hot springs or other hydrothermal spas are particularly well suited to the survival of *Legionella* as a result of the elevated water temperatures.

Dose–response experiments with animals have suggested that high doses of *Legionella* (approximately 10^7 cells) are required to initiate infection (O'Brien and Bhopal, 1993). In contrast, published reviews of the organism have suggested that the median dose is as low as a few organisms (Percival et al., 2004; Pond, 2005). Aerosol carriage of protozoa that have been heavily parasitized by *Legionella* may be one means of increasing the number of organisms available to initiate infection (Percival et al., 2004).

Legionella are important agents of respiratory disease in humans. Legionellosis comprises two forms of illness: Legionnaire's disease and Pontiac fever (Pond, 2005). Legionnaire's disease is a more severe and sometimes fatal form of respiratory illness. The disease is defined by the clinical diagnosis of pneumonia, accompanied by microbiological evidence of infection with *L. pneumophila* or other *Legionella* species (Pond, 2005). Other symptoms can include fatigue, fever, headache, muscle and/or abdominal pain, jaundice and mental confusion. The incubation period is between 3 and 6 days, and recovery is slow, lasting weeks up to several months. Fatalities can result, largely due to respiratory failure. The mortality rates from community-acquired infections are estimated to be in the range of 5–20% (Pond, 2005).

Pontiac fever is a relatively mild, influenza-like illness, defined by non-pneumonic respiratory illness with microbiological evidence of *Legionella* infection. The disease has a shorter incubation period (1–2 days), and illness is considered non-fatal, with infected individuals requiring approximately 2–5 days for recovery. It is estimated that Pontiac fever occurs 2–100 times more frequently than Legionnaire's disease (Hall, 2006).

Groups considered most sensitive to *Legionella* infection include the elderly, immunocompromised individuals, persons with heart or lung disease and individuals who are excessive smokers or consume excess amounts of alcohol.

Despite the fact that *Legionella* species are thought to be ubiquitous in environmental waters, no recorded outbreaks of legionellosis have been reported in Canada or the United States as a result of recreational activity in natural waters. Any reported outbreaks of legionellosis associated with human recreational water contact have been restricted to treated water facilities, such as hot tubs and spas (Moore et al., 1993; Kramer et al., 1996; Levy et al., 1998; Barwick et al., 2000; Lee et al., 2002; Yoder et al., 2004).

Mycobacterium

Mycobacterium species are aerobic, non-motile, non-spore-forming, rod- to coccoid-shaped bacteria. The organisms are considered to have a Gram-positive cell structure. However, the mycobacterial cell wall contains high levels of mycolic acid—complex lipids that give the cell surface a waxy, hydrophobic character that resists Gram staining. Positive staining with acid-fast staining techniques is diagnostic for mycobacteria; thus, the organisms are more commonly referred to as "acid-fast."

The pathogenic mycobacteria encountered in recreational waters are environmental species. They are generally referred to as "atypical" or "non-tuberculous" mycobacteria to distinguish them from *M. tuberculosis* (tuberculosis) and *M. leprae* (leprosy). Neither *M. tuberculosis* nor *M. leprae* is found in the environment. Consequently, they are not a concern for recreational waters. At least 16 different waterborne species have demonstrated the capability for causing infection in humans (Pond, 2005). The species most commonly discussed as having relevance for recreational water exposures are the members of the *Mycobacterium avium* complex (*M. avium* and *M. intracellulare*), which are known to cause respiratory illness; and *M. marinum* and *M. kansaii*, which can cause skin infections.

Environmental mycobacteria are considered ubiquitous in natural waters. They can be found in virtually every medium, including soils, wastewater, lakes, rivers, ponds, streams, groundwater and treated water supplies. Few mycobacteria are encountered in marine waters (Pond, 2005; LeChevallier, 2006). The organisms can survive over a wide range of temperatures, extending from below 0°C to above 50°C. Mycobacteria are not known to be present in high numbers in faeces, and sewage is not considered to be a significant source (Falkinham, 2002). *M. avium* complex members are capable of survival and growth within certain species of phagocytic protozoa, specifically members of the genus *Acanthamoeba*.

In recreational water environments, transmission can occur through contact with waters containing sufficient quantities of the organisms. The main routes of infection are via inhalation of mycobacteria contained within aerosols and through direct water contact with abraded skin. There is little evidence of person-to-person transmission. Environmental mycobacteria are primarily considered to be opportunistic pathogens, as illness is more commonly observed in individuals with some underlying condition that predisposes them to infection (abraded or traumatized skin; or having a weakened or compromised immune system). Exposures to environmental mycobacteria have been most strongly linked to swimming pool and hot tub use, resulting in cases of skin and soft tissue infections and hypersensitivity pneumonitis (inflammation of the lungs). Recreational contact with natural waters is not considered to be a significant risk factor for acquiring mycobacterial illness.

Although environmental mycobacteria are considered ubiquitous in most types of water, to date there have been no recorded outbreaks of mycobacterial-associated illness through contact with natural recreational waters in either Canada or the United States. The risk of healthy individuals acquiring a mycobacterial infection as a result of recreational activity in natural waters is considered extremely low.

Pseudomonas aeruginosa
Pseudomonads are Gram-negative, motile, oxidase-positive, non-spore-forming, rod-shaped bacteria. Over 100 species are currently recognized as belonging to the genus *Pseudomonas* (Hunter, 1997). *P. aeruginosa* represents the most significant species of human concern.

P. aeruginosa is widely distributed in the aquatic environment and can be frequently isolated from fresh water, seawater and soils (Hunter, 1997). The organism has minimal growth requirements and is able to proliferate in waters of low nutrient content. *P. aeruginosa* is infrequently isolated from human faeces (Geldreich, 2006). The organism can be recovered from sewage and stormwater (as these contain a mixture of domestic wastes) and from industrial discharges such as food processing and pulp and paper wastes. Swimmers themselves also present another possible source of *P. aeruginosa*.

Transmission of *P. aeruginosa* in recreational waters occurs through direct body contact with waters containing sufficient quantities of the organism. Ingestion is not considered to be a significant route of infection.

P. aeruginosa can be responsible for causing skin rashes and eye and ear infections among recreational water users. Infection rarely occurs among healthy individuals unless some condition exists that might predispose them to infection (having a history of ear infections, or reporting frequent immersions) (Hunter, 1997). Ear infections occur when *P. aeruginosa* is able to enter into and colonize the outer ear canal. Within a few days of swimming, the ear may become itchy and painful, and discharges of pus may be observed. Skin irritations (dermatitis) present as a red, itchy rash, occurring roughly 18–24 hours after water contact. Infection can progress to folliculitis (inflammation of the hair follicles of the skin), which is marked by an increased tenderness of the infected area and the presence of pus-filled blisters or pimples that surround the hair follicles.

Several epidemiological studies have demonstrated the existence of a link between *Pseudomonas* in natural waters and the incidence of eye and skin infections among swimmers (Seyfried and Cook, 1984; Springer and Shapiro, 1985; Ferley et al., 1989; Marino et al., 1995; van Asperen et al., 1995). Reported outbreaks of *Pseudomonas* dermatitis have virtually all been associated with treated water venues such as hot tubs, swimming pools or hotel whirlpool or spa baths (Moore et al., 1993; Kramer et al., 1996; Levy et al., 1998; Barwick et al., 2000; Lee et al., 2002; Yoder et al., 2004). The incidence of *P. aeruginosa* infections from contact with natural recreational waters is not known, as illnesses are usually of mild severity and typically not recorded.

5.1.3 Other pathogenic bacteria (Leptospira, Staphylococcus)

Leptospira
Leptospira are spirochetes—spirally coiled or corkscrew-shaped bacteria. They are Gram-negative staining, aerobic, long, thin and motile organisms. Initially, the genus *Leptospira* consisted of two species, the pathogenic *L. interrogans* and the free-living *L. biflexa* (WHO, 2003b). At present, 12 *Leptospira* species are recognized, and over 200 pathogenic serotypes have been described, in which the more severe forms are attributed to serovars of *L. interrogans*

(Pond, 2005). By convention, the serotype name is often adopted as the species name when referring to specific strains.

Leptospira species can be either pathogenic or free-living. They are encountered worldwide and are predominantly associated with freshwater environments. The pathogenic leptospires are important zoonotic pathogens that are carried in the renal tract (kidney) of infected animal carriers and excreted in the urine. Small rodents, such as rats, mice and voles, are considered the most important source of pathogenic *Leptospira*. The organisms can also be spread by domestic animals, such as cattle, pigs, dogs and cats, sheep, goats and horses (WHO, 2003b; CDC, 2005b). Heavy rainfall is thought to facilitate the spread of the organisms, as runoff from contaminated soils can affect surface waters (Pond, 2005).

Human infection can occur following direct contact with the urine of infected animals or indirectly though contact with contaminated water, soil or mud. Leptospires gain access through cuts or abrasions in the skin or via passage through the mucous membranes of the eyes, nose and mouth. Ingestion of contaminated water and inhalation of leptospires carried in aerosols are also possible routes of infection. It is thought that ingestion of as few as 1–10 organisms can be sufficient to lead to human illness (Pond, 2005). Recreational water activity is perhaps the most significant source of exposure for acquiring illness, although swimming-associated outbreaks are considered extremely rare (Pond, 2005).

Illness following infection with *Leptospira* can range in severity from a mild, influenza-like illness to more severe, and possibly fatal, disease. Early symptoms of illness are fever, chills, headache, muscle pains, vomiting and reddening of the eyes (Public Health Agency of Canada, 2004). Recovery from mild illness is usually complete, but can be lengthy, in some cases requiring months to years (WHO, 2003b). If left untreated, the disease can progress to more serious illness. Severe cases of leptospirosis can be fatal, with death occurring as a result of kidney failure, cardiorespiratory failure or extensive haemorrhaging. The reasons for the differences in the severity of infection are not fully understood; however, it is believed that each pathogenic serovar possesses the capacity to cause either mild or severe disease (WHO, 2003b).

Illness can be difficult to diagnose, as it may be mistaken for other infections or illnesses that produce similar symptoms. Similarly, mild forms of the illness may not always be reported.

Leptospirosis is considered to be a greater concern among developing countries and in tropical climates. There have been three reported outbreaks of leptospirosis in recreational waters in the United States over the period 1991–2002 (Moore et al., 1993; Barwick et al., 2000; Lee et al., 2002).

Reports of increases in the number of observed cases of leptospirosis in developed countries suggest that *Leptospira* may represent an important re-emerging pathogen (CRC, 2004; Meites et al., 2004). Currently, the prevalence of *Leptospira* in Canadian waters is not known. To date, there have been no documented incidences of *Leptospira* infection from recreational water activity in Canadian waters.

Staphylococcus aureus

Members of the genus *Staphylococcus* are Gram-positive, catalase-positive, non-motile cocci. *S. aureus* is considered the major pathogen of the genus and is the species of most significance for recreational water users.

S. aureus is not considered to be a natural inhabitant of environmental waters. The major reservoirs for this organism are the skin, nose, ears and mucous membranes of warm-blooded animals. The presence of *S. aureus* in recreational waters is predominantly due to releases of the organism from the mouths, noses and throats of swimmers and from discharges from existing infections. The organism can be isolated from human faeces; however, occurrence is thought to be variable (Percival et al., 2004). Sewage and stormwater are additional sources of the organism.

Transmission of *S. aureus* in recreational waters occurs via direct contact with waters containing sufficient quantities of the organism. Infection occurs through cuts or scratches on the skin or, to a lesser extent, through contact with the eyes and ears. Person-to-person spread of the organism is also possible. Ingestion is not considered to be a significant route of exposure. The organism produces a wide array of extracellular toxins, exoenzymes and adherence factors that are used during colonization and infection and for evading host immune defences (Percival et al., 2004). Concentrations of a few hundred cells per millilitre may be sufficient to initiate infection in injured or distressed skin (Percival et al., 2004).

S. aureus is predominantly associated with skin infections in recreational water users. Common infections include infected cuts and scratches, boils, pustules, dermatitis, folliculitis and impetigo (WHO, 2006). Infections are most often pus-forming, with symptoms often not becoming apparent until 48 hours after contact. Other illnesses to which the organism has been linked include eye infections, otitis externa and urinary tract infections (WHO, 2006).

Epidemiological investigations have demonstrated evidence of possible connections between the presence of staphylococci in recreational waters and swimmer illness (Calderon et al., 1991; Charoenca and Fujioka, 1995). However, to date, there has been no conclusive evidence relating the frequency of illness to the concentration of *S. aureus* in recreational waters. In certain instances, testing may provide additional information—for example, in assessing the effects of high swimmer densities on water quality and the potential implications for the possible person-to-person transfer of pathogens.

5.2 Viral pathogens

Viruses are submicroscopic organisms, much smaller in size than bacteria. They are simply constructed, consisting of a nucleic acid core composed of either RNA or DNA, and surrounded by an external protein shell called a capsid. The nucleic acid encodes for viral structural proteins and enzymes necessary for replication, whereas the capsid protects the viral unit from environmental stresses. Some viruses (enveloped viruses) may also possess a lipoprotein envelope surrounding the capsid. Non-enveloped viruses lack this external layer. Viruses are obligate intracellular parasites and must infect a host cell in order to replicate. As a result, they are incapable of replicating outside of their host environment.

The pathogenic viruses of concern for recreational waters are enteric viruses—viruses that infect the human gastrointestinal tract and are shed in human faeces. These viruses are considered to have a narrow host range, meaning that in general, enteric viruses that infect animals do not infect humans, and vice versa. Transfer to humans in recreational waters occurs via the faecal–oral route through the accidental ingestion of contaminated waters. Some viruses, like the adenoviruses, have additional routes of infection, such as via inhalation or through contact with mucosal membranes of the eyes. Enteric viruses cause a wide variety of human health effects, which can range in severity from mild to severe. Gastrointestinal symptoms (nausea, vomiting, diarrhoea) are the most commonly encountered symptoms of viral illness. Some virus infections can result in more serious health outcomes, although these are considered to be much rarer.

5.2.1 Enteric viruses

There are over 100 types of viruses that can be excreted in faeces and thus can potentially be transmitted to recreational waters. The viruses most commonly associated with waterborne illness include adenoviruses, astroviruses, enteroviruses (polioviruses, coxsackieviruses and echoviruses), noroviruses, rotaviruses and the Hepatitis A virus.

Enteroviruses

Enteroviruses are a large group of small (20–30 nm), non-enveloped RNA viruses belonging to the family Picornaviridae. Members of this group include polioviruses, coxsackieviruses, echoviruses and several yet unclassified enteroviruses. Many enterovirus infections are asymptomatic. The symptoms and severity of illness vary considerably among the individual virus types and serotypes. The most commonly observed health effects are vomiting, diarrhoea, febrile flu-like symptoms, malaise, respiratory disease, headache and muscle ache (Percival et al., 2004). More serious outcomes have been associated with individual virus groups, including myocarditis (coxsackievirus), aseptic meningitis (coxsackievirus, poliovirus), encephalitis (coxsackievirus, echovirus) and poliomyelitis (poliovirus), although these are not considered to be common.

Noroviruses

The term “norovirus” has been designated as the official name for the group of viruses formerly known as Norwalk viruses, Norwalk-like viruses or “small, round, structured viruses” (SRSV). Noroviruses are small (27–30 nm), non-enveloped RNA viruses. Norovirus infection is considered to be the leading cause of viral gastroenteritis outbreaks (from all sources) in the United States and the United Kingdom (Percival et al., 2004). The primary symptoms of illness are diarrhoea, vomiting, headache and muscle ache. The onset of projectile vomiting is considered a characteristic trait of norovirus infection. Asymptomatic infections with norovirus are rare. In healthy adults, illness is self-limiting, rarely progressing to more serious concerns (e.g., dehydration). Infection is considered more serious among vulnerable groups such as the elderly.

Rotaviruses

Rotaviruses are larger (60–80 nm), non-enveloped RNA viruses. Dose–response studies have suggested that rotaviruses are the most infective of all the enteric viruses (Gerba et al., 1996). Rotavirus infection has been implicated as the number one cause of infantile gastroenteritis worldwide. Although all age groups can be affected, infections among healthy adults are often asymptomatic as a result of the immunity acquired during childhood (Percival et al., 2004). Diarrhoea constitutes the predominant symptom of illness, which can become life-threatening should the resulting dehydration and electrolyte imbalance become severe. Groups considered vulnerable for severe disease- and illness-induced mortality include young children, immunocompromised individuals and the elderly.

Adenoviruses

Adenoviruses are also larger by comparison (70–100 nm) and are non-enveloped DNA viruses. Over 49 individual serotypes have been identified as being capable of causing human illness, with the clinical features and severity of illness varying considerably among the individual types (Percival et al., 2004). The majority of adenovirus serotypes cause respiratory illness, which presents with pharyngitis and cough and cold-like symptoms. Conjunctivitis can also occur as a result of infection of the eye. Gastrointestinal illness, caused exclusively by serotypes 40 and 41, is also a frequently reported outcome. Adenovirus is thought to be second only to rotaviruses as a cause of childhood gastroenteritis (Crabtree et al., 1997). Asymptomatic illness is commonly observed with gastrointestinal infection, as it is believed that the immunity conferred during early childhood is lifelong (Percival et al., 2004).

Hepatitis A virus

The Hepatitis A virus (HAV) is a small (25–28 nm), non-enveloped RNA virus whose major target organ is the liver. The majority of HAV infections are asymptomatic. Illness is most frequently reported among adults. Symptoms include malaise and fever, followed by nausea, vomiting, abdominal pain and, ultimately, jaundice. Infection is typically self-limiting.

Astroviruses

Astroviruses are small (28–30 nm), non-enveloped RNA viruses. Of the viral agents known to cause enteric illness, the significance of astroviruses as a cause of waterborne illness is perhaps the least well characterized (Percival et al., 2004). Illness in infected individuals appears similar to rotaviral illness, although markedly less severe.

Occurrence in the environment

Enteric viruses are shed in high numbers in the faeces of infected individuals and can reach concentrations as high as 10^{10}–10^{12} particles per gram of faeces (Gerba, 2000). Even asymptomatic individuals (those infected, but not exhibiting symptoms of disease) are capable of excreting large numbers of viruses.

The principal route of entry for viruses in recreational waters is via the discharge of sewage-contaminated wastes. Point sources of pollution such as municipal sewage discharges or combined sewer overflows constitute the primary sources of sewage contamination. Non-point sources capable of contributing to the viral loading of environmental waters include storm drains, river discharges (which capture runoff from urban and rural areas) and malfunctioning or

improperly designed septic waste systems. Swimmers themselves, particularly young children, can also present a source of contamination through faecal shedding and the accidental release of faecal material. Animal wastes, although capable of harbouring many bacterial and protozoan pathogens, are considered to be of low risk for the transmission of viruses to humans (Cliver and Moe, 2004; Percival et al., 2004). There have been some examples of animals serving as a reservoir for human viruses (avian influenza virus, West Nile virus), yet to date there has been no documented evidence of human waterborne infection having been caused by animal viruses (Cliver and Moe, 2004).

The total viral load of sewage can be quite constant; however, the types and numbers of individual viruses are strongly influenced by the rates of epidemic and endemic illness within the discharging population. As a result, the viral composition of sewage can vary considerably, often demonstrating strong seasonal trends (Krikelis et al., 1985; Tani et al., 1995; Pina et al., 1998; Lipp et al., 2001). Published estimates of culturable viruses in raw sewage suggest that concentrations can reach over 10 000 infectious units per litre (Reynolds et al., 1998; Payment et al., 2001). The presence of viruses in surface waters is expected to vary regionally and is dependent upon (among other factors) the degree and type of faecal contamination and the rates of environmental inactivation. Detectable levels of culturable enteroviruses in surface waters in general have ranged from 1–10/100 L to 1–200/L for more contaminated waters (Pina et al., 1998; Reynolds et al., 1998; Payment et al., 2000; Lipp et al., 2001).

Studies have reported the detection of enteroviruses, noroviruses, rotaviruses, adenoviruses, HAV and astroviruses in marine and fresh waters used for recreational purposes in the United States, Europe and Canada (Payment, 1984; Puig et al., 1994; Pina et al., 1998; Griffin et al., 1999; Chapron et al., 2000; Payment et al., 2000; Schvoerer et al., 2001; Denis-Mize et al., 2004; Jiang and Chu, 2004; Laverick et al., 2004). The studies report varying figures in terms of virus detection frequencies and concentrations. Differences in the analytical methods used in the respective studies prevent direct comparisons of the results; nevertheless, the information provided by these investigations serves to shed some light on the potential vulnerability of recreational waters to contamination with pathogenic viruses.

Viruses are hardy microorganisms that can survive for prolonged periods once shed into the aquatic environment. Survival is dependent upon a number of biological and environmental factors, including the virus's specific physical characteristics, the presence of natural microbial predators and various water characteristics, such as temperature, pH, salinity, turbidity and ultraviolet (UV) levels. Data on the survival of individual virus types in natural waters have been limited. Viruses are generally regarded to be more resistant to environmental degradation than bacteria, and experimental data suggest that some enteric viruses may demonstrate greater resistance than some enteric protozoa (e.g., *Giardia*) (Johnson et al., 1997).

Epidemiology

Surveillance data on recreational water outbreaks published by the U.S. CDC indicated that for the period 1991–2002, 13% (8 of 64) outbreaks of gastroenteritis reported in natural waters were caused by noroviruses (Moore et al., 1993; Kramer et al., 1996; Levy et al., 1998; Barwick et al., 2000; Lee et al., 2002; Yoder et al., 2004). In general, noroviruses were responsible for between 0 and 2 outbreaks per year, with the total number of cases per outbreak ranging from 11 to

168 individuals. Recreational lakes constituted the setting for the majority of the outbreaks, and a 2002 outbreak involving 44 cases at Lake Michigan State Park in Wisconsin was the first documented recreational water outbreak at a Great Lakes beach (Yoder et al., 2004). No other virus types were implicated in any of the other outbreaks reported from 1991 to 2002.

Outbreaks of acute gastroenteritis in which the causative agent could not be identified also regularly constitute a significant proportion of the total number of recreational water-related outbreaks. It is generally suspected that many of these outbreaks are of viral origin. Pathogenic viruses are notoriously difficult to detect, and the short incubation times, range of symptoms encountered and high frequency of illness observed among children are all consistent with viral infections (Cabelli, 1983; Mena et al., 2003; Percival et al., 2004). According to CDC surveillance data, outbreaks of acute gastrointestinal illness of unknown etiology accounted for 23% (14 of 64) documented outbreaks from 1991 to 2002 (Moore et al., 1993; Kramer et al., 1996; Levy et al., 1998; Barwick et al., 2000; Lee et al., 2002; Yoder et al., 2004). Again, freshwater lakes were identified as the most frequent setting for such outbreaks.

Several epidemiological studies have attempted to characterize the relationship between enteroviruses in swimming water and the incidence of recreational water illness (Lightfoot, 1988; Alexander et al., 1992; Fewtrell et al., 1992; van Dijk et al., 1996; Lee et al., 1997; van Asperen et al., 1998; Haile et al., 1999). In general, no significant relationships could be shown between the concentration of enteric viruses and the incidence of swimmer illness. Haile et al. (1999) did observe an increase in the reporting of a number of adverse health effects (vomiting, fever, sore throat and highly credible gastrointestinal illness) on days on which enteroviruses were detected in swimming waters affected by stormwater discharges.

Relationship with indicators

A number of studies have reported a lack of a relationship between the concentration of faecal indicator bacteria and the presence of enteric viruses in recreational waters (Griffin et al., 1999; Schvoerer et al., 2000, 2001; Jiang et al., 2001; Noble and Fuhrman, 2001; Denis-Mize et al., 2004; Jiang and Chu, 2004; Wetz et al., 2004). Pathogenic viruses have been detected in recreational waters at faecal indicator bacteria concentrations below the existing limits for recreational water quality. The reverse has also been true—that waters with indicator counts well above recreational water limits have yielded negative results for the presence of viruses. The lack of a correlation between faecal indicators and enteric viruses is not unexpected, as faecal indicators are present consistently in human and animal wastes and in relatively constant numbers, whereas viruses are specific to human wastes and shedding may be intermittent and seasonal. Viruses are also more resistant than bacteria to environmental stresses and may persist for longer periods. Other organisms have been proposed as potential surrogate indicators (strains of enterovirus, *C. perfringens*, coliphages and phages of *B. fragilis*); however, investigations conducted to date have not demonstrated conclusive evidence of a link between these organisms and the detection of viruses in contaminated surface waters (Pina et al., 1998; Griffin et al., 1999, Lipp, 2001; Jiang and Chu, 2004).

5.3 Protozoan pathogens

Pathogenic protozoa of importance to recreational waters include both enteric and free-living species. Enteric protozoa are common parasites that infect the intestinal tract of humans and other mammals. They are obligate parasites, meaning that they require the infection of a host to replicate and are incapable of growth outside the host environment. The most important stage of their life cycle involves the production of cysts or oocysts that are shed in large numbers in the faeces. These cysts or oocysts are extremely resistant to environmental stresses and can survive for long periods in the environment. Upon ingestion by a new host, the (oo)cysts undergo excystation in the small intestine to initiate infection. These organisms can enter recreational waters as a result of direct or indirect contact with human or animal faeces. Transmission to humans occurs through the accidental ingestion of contaminated waters. The most common manifestations of illness are gastrointestinal symptoms, specifically diarrhoea. *E. coli* and enterococci are used to indicate faecal contamination and thus the possible presence of these faecal enteric pathogens. Indicator absence does not necessarily indicate that enteric protozoa are also absent.

Free-living protozoa, unlike enteric protozoa, occur naturally in recreational waters and do not require the presence of a host organism to complete their life cycle. Transmission to humans can occur in waters containing sufficient quantities of the organisms through mechanisms such as inhalation or through direct contact with mucous membranes (e.g., those of the eye). The types of illnesses caused by these organisms are varied and include infections of the central nervous system and eye infections. As these organisms are not of faecal origin, faecal indicators are not expected to correlate well with the presence of these protozoans. Currently, there is no recognized microbiological indicator for these pathogens.

5.3.1 Enteric protozoa (Giardia, Cryptosporidium)

The enteric protozoa of most importance to recreational waters are *Giardia* and *Cryptosporidium*.

Giardia

Giardia spp. are small, flagellated protozoan parasites. Species have a two-staged life cycle consisting of a trophozoite (feeding stage) and an environmentally resistant cyst stage. *G. duodenalis* (syn. *lamblia*, *intestinalis*), found in humans and a wide range of other mammals, is the only human-infective species. Other species (*G. muris*, *G. agilis*, *G. microti*, *G. psittaci* and *G. ardea*) have been reported in animals, including rodents, birds and amphibians. Molecular characterization of *G. duodenalis* has demonstrated the existence of genetically distinct genogroups (assemblages) depending on their host range. Some groups have shown occurrence across both human and animal hosts, whereas others have been shown to be host-specific.

Human and animal faeces (especially cattle) are major sources of *G. duodenalis*. Other recognized animal hosts include beavers, muskrats, dogs, sheep and horses. Many of these animals can be infected with *G. duodenalis* originating from human sources (Davies and Hibler, 1979; Hewlett et al., 1982; Erlandsen et al., 1988). Epidemiological and molecular data suggest that it is only these human-source strains that have been significantly associated with

human illness. The pathogenicity of other, animal-specific *G. duodenalis* strains and *Giardia* species is not fully known. As a result, it remains sound practice to consider any *Giardia* cysts found in water as potentially infectious to humans.

Giardia is commonly encountered in sewage and surface waters. In general, concentrations in wastewater are in the range of 5000–50 000 cysts/L, with surface water concentrations typically ranging from < 1 to 100 cysts/100 L (Medema et al., 2003; Pond et al., 2004).

The exact mechanisms through which *Giardia* causes illness are not completely understood. Damage to the intestinal mucosa caused by attachment and detachment of the trophozoites contributes to the impairment of intestinal function. The severity of *Giardia* infection can range from no observable symptoms to severe gastrointestinal illness requiring hospitalization. The most common symptoms of illness include explosive, watery diarrhoea, nausea, intestinal upset, fatigue, low-grade fever and chills.

In theory, a single cyst is sufficient to cause human infection. However, studies have shown that the dose required to cause infection is usually greater. Human (volunteer) feeding studies have suggested that the median dose for infection is around 50 cysts (Hibler et al., 1987), although subjects have shown infection at doses much lower than this (Rendtorff, 1978). The time between ingestion and the excretion of new cysts (prepatent period) ranges between 6 and 16 days. Infection is self-limiting, clearing within 1–3 weeks on average. Some patients may remain as asymptomatic carriers, whereas in other cases individuals may experience recurrent bouts of the disease, a phase persisting for a period of several months to a year. Persistent illness can be treated using a number of antiparasitic drugs.

Cryptosporidium

Cryptosporidium are small, non-motile protozoan parasites. These organisms possess a complex, multi-staged life cycle, of which the most important stage is production of the round, thick-walled oocysts. Sixteen species are currently recognized as belonging to the genus. Two predominant genotypes have been linked to human illness: *C. hominis* (genotype 1), reported only in humans, and *C. parvum* (genotype 2), reported in humans, calves and other ruminants. Other species and genotypes have been encountered, but much less frequently.

Humans and cattle are the most significant sources of *Cryptosporidium*. Sheep, pigs and horses are also considered to be reservoirs (Olson et al., 1997). Rodents are not a significant source of human-infective *Cryptosporidium* (Roach et al., 1993).

Oocysts are commonly found in water affected by human or livestock wastes by mechanisms such as sewage, swimmer contamination and stormwater runoff. It is suggested that waterfowl (ducks, geese) may be capable of picking up oocysts from their habitat and depositing them elsewhere through discharge in their faeces. Typical concentrations in wastewater are on the order of 1000–10 000 oocysts/L, whereas surface water concentrations in general range from < 1 to 5000 oocysts/100 L (Guy et al., 2003).

The precise means through which *Cryptosporidium* causes human illness is not fully understood. Damage caused by infection of red blood cells in the mucosa of the small intestine is known to

contribute to illness. *Cryptosporidium* infection can result in illness of varying severity, ranging from asymptomatic carriage to severe, life-threatening illness in immunocompromised individuals. The primary characteristic of illness is profuse, watery and sometimes mucoid diarrhoea. Other symptoms include nausea, vomiting, abdominal pain, weight loss, anorexia and low-grade fever.

For *Cryptosporidium*, a variety of median infective doses have been reported—although, as is the case with other pathogens, a single organism is theoretically sufficient to initiate infection. Most (volunteer) feeding studies suggest that the median infective dose of *Cryptosporidium* is between 80 and 140 oocysts (DuPont et al., 1995; Chappell et al., 1999, 2006; Okhuysen et al., 2002). The prepatent period (time between ingestion and the excretion of new cysts) is roughly 4–9 days. Most healthy individuals experience a complete recovery, with the disease resolving itself in about 1–2 weeks. Oocysts may continue to be shed in faeces for a short period following recovery. Currently, there is no effective treatment for cryptosporidiosis in adults. The use of the antimicrobial drug nitazoxamide has been approved by the U.S. Food and Drug Administration for treatment in children (Health Canada, 2012b).

Epidemiology

Reported outbreaks of giardiasis and cryptosporidiosis from natural recreational waters have been infrequent. Surveillance data published by the U.S. CDC for the period 1992–2002 indicated that *Giardia* was responsible for 9% (6 of 64) of the total number of outbreaks of gastroenteritis reported for natural waters (Moore et al., 1993; Kramer et al., 1996; Levy et al., 1998; Barwick et al., 2000; Lee et al., 2002; Yoder et al., 2004). Locations included recreational lakes, a recreational river and a pond setting. Although *Giardia* has not been linked to outbreaks in natural recreational waters in Canada, it is likely that cases have occurred that were not detected or went unreported.

Surveillance data over the same period indicated that 6 (9%) of the 64 outbreaks of gastrointestinal illness reported in natural waters were caused by *Cryptosporidium* (Moore et al., 1993; Kramer et al., 1996; Levy et al., 1998; Barwick et al., 2000; Lee et al., 2002; Yoder et al., 2004). Recreational lakes were recorded as the setting for the majority of the outbreaks. A large outbreak at a New Jersey lake in 1994 involving 418 cases was the first recorded U.S. outbreak of cryptosporidiosis related to recreational water use (Kramer et al., 1996). Treated recreational water venues such as water parks and community and motel swimming pools have provided the setting for the majority of outbreaks of cryptosporidiosis. Surveillance in Canada has been limited; to date, however, there have been no reported outbreaks of cryptosporidiosis associated with natural recreational waters. As with *Giardia*, it is expected that cases have occurred and gone undetected or were not reported.

Relationship with indicators

Although the faecal indicators *E. coli* and enterococci are good indicators for enteric bacterial pathogens commonly found in natural recreational waters, they have proven to be less effective indicators of protozoan presence. Studies have demonstrated a lack of correlation between concentrations of *E. coli* and enterococci and the presence of *Giardia* and *Cryptosporidium* in surface waters (Hörman et al., 2004; Dorner et al., 2007; Sunderland et al., 2007). *E. coli* and enterococci are present consistently in human and animal faeces, whereas the presence of

Cryptosporidium and *Giardia* is source dependent. Additionally, shedding of these organisms in the feces of recognized sources can be intermittent and seasonal. *Giardia* cysts and *Cryptosporidium* oocysts are also more resistant to stresses than bacteria and may persist in the environment for longer periods.

Other enteric protozoa of potential concern

Other enteric pathogenic protozoa such as *Entamoeba* and *Toxoplasma* can be shed in human and animal faeces and thus can conceivably be present to contaminate recreational waters. Currently, there have been no reported outbreaks involving these organisms in recreational waters. Recreational water activity is not considered to be a significant risk factor for illness caused by these organisms.

5.3.2 Free-living protozoa

The free-living protozoa recognized as the most important in natural recreational waters are *Naegleria* and *Acanthamoeba*.

Naegleria fowleri

Naegleria are small, thermophilic, free-living freshwater amoebae. The genus *Naegleria* is composed of six species. *N. fowleri* is the primary human pathogen and the species of concern for recreational waters. The organism has a multi-stage life cycle consisting of a motile feeding trophozoite stage, a non-replicating flagellate stage and an environmentally resistant cyst stage.

N. fowleri can be found worldwide in fresh water and soil. The organism has been isolated from both natural and artificial water supplies, including lakes, rivers, hot springs, swimming pools, hydrotherapy baths and tap water. No human or animal reservoirs have been identified. The organism prefers warmer waters and can tolerate temperatures of 40–45°C (Percival et al., 2004). Tropical and subtropical fresh waters and hot springs are particularly well suited for the survival of *N. fowleri*. In colder waters, it is thought that the cyst form may survive in river and lake sediments (Pond, 2005).

N. fowleri causes a disease of the central nervous system called primary amoebic meningitis (PAM), which is almost always fatal. Human infection occurs when water containing the amoeba is forcefully inhaled or splashed into the nasal passages (e.g., during diving, jumping, falling or swimming underwater). Following inhalation, the organism travels through the nasal passages to the brain, causing damage to the cells of the olfactory system and cerebral cortex. The onset of illness is rapid. Symptoms include severe headache, high fever, intracranial pressure, stiff neck, altered mental status and coma, ultimately leading to death. Treatment is possible but requires prompt diagnosis and aggressive antimicrobial therapy. The organism is reported to exhibit sensitivity to amphotericin B.

Naegleria have also been proposed as natural hosts for the bacterial pathogen *Legionella*. Harbouring within *Naegleria* is thought to provide *Legionella* with an environment suitable for replication, in addition to providing protection from environmental stresses.

Cases of PAM are extremely rare; it is estimated that in the United States, one case occurs for approximately every 2.5 million swimmers (Visvesvara and Moura, 2006). Surveillance data

indicated that 29 cases of the disease were reported in the United States over the period 1992–2002, with an average occurrence of 0–6 cases per year (Moore et al., 1993; Kramer et al., 1996; Levy et al., 1998; Barwick et al., 2000; Lee et al., 2002; Yoder et al., 2004). Reported outbreaks have been limited to the southern United States, including Florida, Texas, Oklahoma, California, Georgia and North Carolina. To date, there have been no recorded cases of PAM as a result of recreational water contact in Canadian waters. The bulk of the evidence suggests that amoebic meningoencephalitis is an unlikely health concern in Canada. However, researchers have suggested that increasing lake temperatures brought on by climate change could result in an expanded prevalence of this organism (Rose et al., 2001; Schuster and Visvesvara, 2004). Thus the potential exists for this organism and disease to become an emerging concern for recreational waters in the northern United States and Canada in the future.

Acanthamoeba

Acanthamoeba are small, free-living amoebae. The organisms possess a two-stage life cycle consisting of a trophozoite feeding stage and an environmentally resistant cyst stage. The genus *Acanthamoeba* contains approximately 20 species. *A. culbertsoni, A. polyphaga* and *A. castellanii* are the species most commonly associated with human infections.

Acanthamoeba are considered ubiquitous in the environment. They can be found in fresh, estuarine and marine waters, hot springs, soils and sewage, and human-made water supplies, such as tap water and air conditioning condensers.

Pathogenic species of *Acanthamoeba* are responsible for two distinct clinical illnesses: amoebic keratitis (AK), a painful, vision-threatening disease of the cornea caused by *A. polyphaga* and *A. castellanii*; and granulomatous amoebic encephalitis (GAE), a fatal disease of the central nervous system caused by *A. culbertsoni*. Infections occur via inhalation or through direct contact with the mucous membranes of the eye or through abraded or traumatized skin.

Although pathogenic *Acanthamoeba* species can have a waterborne transmission, recreational water contact is not considered to be a significant risk factor for either of these illnesses. The principal risk for AK is poor hygienic practices in contact lens wearers (use of contaminated solutions, inadequate disinfection practices). Infection can be acquired by wearing contact lenses while swimming in lakes or ponds; however, the risk is considered extremely low. Currently, recreational water contact is not considered a route for acquiring GAE.

Acanthamoeba have also been proposed as natural hosts for certain free-living bacterial pathogens, namely *Legionella* and *Mycobacterium*. Survival within *Acanthamoeba* is thought to provide these organisms with an environment suitable for replication, as well as to provide protection from environmental stresses.

6.0 Cyanobacteria and their toxins

Guideline values
The recommended guideline values for cyanobacteria and their toxins in recreational waters are:

Total cyanobacteria:		100 000 cells/mL
	or	
Total microcystins:		20 µg/L (expressed as microcystin-LR).

Exceedance of these values, or the development of a bloom indicates the potential for exposure to cyanobacterial cells and/or their toxins in amounts which may, in some cases, be sufficient to be harmful to human health. In general, contact with waters where a bloom exists or has very recently collapsed should be avoided.

An appropriate monitoring program is advantageous to reduce the potential risk of user exposure to cyanobacterial blooms and their toxins. It is advised that managed recreational water areas that are suspected or are known to be susceptible to blooms be routinely monitored during the bathing season. Authorities should visually monitor such supplies for cyanobacterial growth. A swimming advisory may be issued at the discretion of the responsible authority.

In the event of bloom development, in order to fully characterize the extent of the risk posed by the cyanobacterial population, it is further advised that authorities conduct sampling during, and after the collapse of, the bloom. Should either of the guideline values be exceeded, a swimming advisory may be issued by the responsible authority. When measuring the toxins it is important that one measures total microcystins. “Total microcystins” includes both the microcystin that is occurring free in the water and the microcystin that is bound to or inside the cyanobacterial cells.

Contact with waters where an advisory has been issued should be avoided until the advisory has been rescinded. Published texts are available that can provide further information with respect to the design and implementation of recreational water monitoring programs (e.g., Chorus and Bartram, 1999)

Cyanobacteria
Cyanobacteria are bacteria that share features of both bacteria and algae. They are similar to algae in size, possess blue-green pigmentation and are capable of photosynthesis; thus, they are often termed blue-green algae (WHO, 2003a). Most planktonic cyanobacteria, including the species found in Canadian lakes, form colonies, which can appear as irregular groupings of cells or as filamentous chains that can be straight, coiled or branched (Chorus and Bartram, 1999; Falconer, 2005). In a typical summer, a lake water sample can contain several species of cyanobacteria, along with numerous other species of algae. Cyanobacterial cells contain small gas bubbles called vacuoles, which allow them to control their buoyancy. The cells use this buoyancy control to move up in the water column to where light is the greatest and down in the

water column to where nutrients are more abundant (Falconer, 2005). In still, stratified surface waters, cyanobacteria effectively use the light and nutrients to proliferate intensively, creating a visible discoloration known as cyanobacterial blooms (Chorus and Bartram, 1999; Falconer, 2005). These blooms can be very dense and can have the appearance of being gelatinous or resemble a collection of fine grass clippings or appear as a homogeneous, soupy mass, as if green paint has been spilled into the water (WHO, 2003a; Falconer, 2005). Surface blooms or scums can occur when the cells develop excess buoyancy and the water is calm enough to let them float to the surface. This excess buoyancy develops when turbulence (e.g., during storms) sends the cellsa too deep, during the hours of darkness, under carbon dioxide-limiting conditions, when the population is at the end of the growth cycle or any combination of these factors. Offshore winds may then drive these scums towards the shore where they can accumulate (Chorus and Bartram, 1999; Falconer, 2005). In this manner, cyanobacterial blooms may increase their density by a factor of 1000 or more in a very short period of time (Chorus et al., 2000).

Cyanobacteria possess a number of special properties that determine their relative importance in phytoplankton communities. Cyanobacteria have lower light intensity requirements and demonstrate a greater affinity for nitrogen and phosphorus than do other algae and phytoplankton and, thus, can outcompete these organisms under conditions where these factors might be limiting (e.g., in turbid waters). The maximum growth rate for cyanobacteria has been reported to occur at temperatures above 25°C (Chorus and Bartram, 1999). The combination of these factors may help explain why blooms typically occur in the late summer months when waters are warmer and the hours of daylight are beginning to grow shorter (Chorus and Bartram, 1999, Falconer, 2005). As well, toxic cyanobacteria may not be grazed by zooplankton to the same extent as other species of algae (Chorus and Bartram, 1999). More recently, it has been suggested that certain mussel species (zebra mussels *Dreissena polymorpha*, quagga mussels *D. bugensis*) selectively reject certain strains of *Microcystis*, possibly leaving toxic species to proliferate under conditions of reduced competition (Brittain et al., 2000; Vanderploeg et al., 2001).

The nutrient enrichment (eutrophication) of surface waters can also have a significant impact on the frequency and severity of cyanobacterial blooms (Chorus and Bartram, 1999, Falconer, 2005). Nitrogen and phosphorus enter the environment as a result of both natural processes and human activities (Chambers et al., 2001). Sources include runoff or erosion from naturally or artificially fertilized soils, storm sewer discharges and discharges of agricultural, industrial or sewage wastes (Jones and Armstrong, 2001). On-site wastewater disposal systems from isolated dwellings (both urban and rural) can also be a significant source of nutrients.

Cyanobacterial toxins

Cyanobacterial blooms are considered a public health concern because direct contact with a bloom can cause allergenic-like reactions; and some cyanobacterial species can produce toxins that may have harmful effects on humans (Chorus and Bartram, 1999). Over 46 species of cyanobacteria are capable of producing toxins (Sivonen and Jones, 1999). The most common toxin-producing genera in fresh water are *Anabaena, Aphanizomenon, Cylindrospermopsis*, *Microcystis*, *Nodularia* and *Planktothrix* (syn. *Oscillatoria*) (Falconer, 2005). Although the conditions leading to the development of a bloom are relatively well known, the factors responsible for the dominance of toxin-producing strains are not completely understood (Chorus

and Bartram, 1999; Falconer, 2005). Consequently, toxin formation from cyanobacteria is even less predictable than the cyanobacterial blooms themselves. Lakes that have never had a problem can suddenly develop blooms that may contain toxins. Conversely, lakes that have shown toxic blooms in the past may never show it again. It has been suggested that worldwide, an average of 60%of the cyanobacterial bloom samples investigated have been positive for cyanobacterial toxins (range, 10–90%) (Chorus et al., 2000; WHO, 2003a). As a result, any bloom encountered should be treated as potentially toxic. Cyanobacterial toxins for the most part are associated with the cyanobacterial cells—either bound within membranes or occurring freely within the cells. Toxin release to the surrounding waters can occur as the cells die or are damaged and leak their contents (Chorus and Bartram, 1999). The bulk of the toxicity, if present, generally lasts as long as the bloom; however, some toxin may still persist for a short period after the bloom is gone (Chorus and Bartram, 1999; Falconer, 2005). Subsequently, contact with waters in which a bloom has developed should be avoided until it can be unequivocally determined that there is not a risk of contact with cyanobacterial toxins. The time of toxin persistence can depend on factors such as the concentration before the bloom's disappearance, and the efficiency of degradation by natural microbial populations in the water (Falconer, 2005).

Some toxic benthic species (e.g., *Lyngbya* spp.) can grow to form dense, bottom-covering mats of cyanobacterial material (Chorus and Bartram, 1999; WHO 2003a). These mats typically occur in clear, shallow waters where sunlight can penetrate to the bottom. The mats can occasionally be dislodged and washed ashore, where they may be scavenged by animals. Human health risks from these mats are considerably less than those from scums produced by other cyanobacterial species; however, they can still present a risk for domestic pets and livestock (WHO 2003a).

There are other known toxic marine algal species that are capable of forming blooms (e.g., *Alexandrium* spp. and the phenomenon known as "red tide") and producing toxins (i.e., those toxins responsible for shellfish poisonings) (Chorus and Bartram, 1999). However, as the focus of this section is the human health risks from exposure to toxic cyanobacteria through recreational water activities, they will not be discussed here.

Confirmation of toxins within a bloom cannot be accomplished by visual inspection; samples must be sent to a laboratory for analysis. Warning signs may be observed, such as the presence of dead waterfowl or other wildlife along the shoreline or reports of domestic animal poisonings (specifically cattle and dogs) (Chorus and Bartram, 1999). Still, toxic blooms can occur without any noticeable effect on the local animal populations.

Increased awareness of cyanobacterial blooms, coupled with occasional escalations in bloom detection, has prompted a growing concern over the possible development of blooms in recreational waters and the resulting health implications for recreational water users. Both drinking water and recreational water are potential sources of exposure. In general, because of circumstances such as the seasonality and localized nature of blooms, their unappealing aesthetic properties and the way drinking water supplies and monitored recreational water areas are managed, the likelihood of exposure to cyanobacterial toxins in sufficient amounts to constitute a chronic or acute health risk is considered to be relatively low. Under circumstances where a recreational area is experiencing prolonged and persistent blooms and where intensive

recreational activities are continuing, the risks of acute exposure may be greater (Funari and Testai, 2008).

Direct epidemiological evidence of adverse health effects associated with recreational exposure to cyanobacteria is limited. Information on human symptoms of illness have come largely from anecdotal and case reports, many of which have gaps in the information on cyanobacterial conditions at the time of exposure (e.g., species present, cell and/or toxin concentrations). The symptoms most frequently described in these reports have been hay fever-like symptoms, gastrointestinal symptoms and skin irritations (Stewart et al., 2006a). Additional information has been provided from toxicological studies that used animal models, as well as from occasional accounts of animal poisonings.

There are three main routes for human exposure to cyanobacteria and their toxins in recreational waters: ingestion, inhalation and direct body contact (Chorus and Bartram, 1999). Cases of illness have been reported that have provided anecdotal evidence of toxicity to recreational water users in bloom-impaired waters through the accidental swallowing of water (Chorus and Bartram, 1999; Stewart et al., 2006a). There has also been experimental evidence to suggest that inhalation of contaminated aerosols may be equally important as a potential route of exposure (Fitzgeorge et al., 1994; Chorus and Bartram, 1999). This route would be relevant for activities in areas where aerosols are generated, such as waterskiing. Activities involving sudden or repeated immersion of the head (such as diving, windsurfing or kayaking) may also lead to ingestion or inhalation exposure via water forced into the mouth and/or nasal passages. Direct contact with cyanobacterial populations has also been known to cause irritative effects of varying severity, although the exact mechanisms for this are not fully understood. Some allergic reactions have been reported in sensitive individuals. It has been suggested that the irritations are due to unknown cyanobacterial components, separate from the toxins (Chorus and Bartram, 1999). Bathing suits and wet suits may also function to exacerbate the potential for skin irritations by trapping the cells and then disrupting their contents as a result of the friction created between the suit material and the user's skin (Chorus and Bartram, 1999).

There are several known cyanobacterial toxins that can pose concerns for recreational water users. These include microcystins, nodularins, anatoxins, cylindrospermopsin, dermatotoxins and irritant toxins (Chorus and Bartram, 1999). Microcystins and nodularins are cyclic peptides that affect the liver (hepatotoxins), anatoxins are alkaloids that target the nervous system (neurotoxins) and cylindrospermopsin is an alkaloid that affects a wide range of organs (general cytotoxin) (Chorus and Bartram, 1999). The dermatotoxins (alkaloids) and irritant toxins (lipopolysaccharides) are toxins that cause irritations of exposed tissues (Chorus and Bartram, 1999).

Microcystins

Microcystins are hepatotoxins that disrupt the functioning of enzymes called protein phosphatases, which are important metabolic switches in human and animal cells (WHO, 2003a). Their primary target is the liver, with the main route of entry into cells occurring through a membrane transport mechanism known as the bile acid carrier (Chorus and Bartram, 1999).

Microcystins are produced by most species of *Microcystis* and some species of *Anabaena*—two notable scum producers (WHO, 2003a). Other cyanobacteria capable of producing the toxin are *Oscillatoria* (syn. *Planktothrix*), *Nostoc* and *Anabaenopsis*. Over 70 microcystin variants have been isolated from bloom samples (Sivonen et al., 1992). Variants are named according to the variable amino acid position encountered in their structure. Microcystin-LR, the most commonly encountered variant, is named for possessing the amino acids leucine (L) and arginine (R) in the variable position (Chorus and Bartram, 1999). Microcystins and, more specifically, microcystin-LR represent the most frequently encountered cyanotoxins in cyanobacterial blooms in temperate surface waters in North America, based on available monitoring data. Therefore, they are the cyanotoxins of most relevance for recreational waters in Canada.

Symptoms reported from incidences of human recreational water exposures to waters contaminated by blooms of *Microcystis* and *Anabaena* have included headaches, nausea, vomiting, diarrhoea, abdominal pain, muscle aches, fever, mouth ulcers, blistering of the lips, sore throat, skin rashes and ear and eye irritations (Chorus and Bartram, 1999). Cases of animal poisonings (cattle, sheep and canines) resulting from contact with *Microcystis* blooms have provided evidence of liver toxicity and have included fatalities, substantiating the concern for human health effects from exposure. To date, no human fatalities have been reported as a result of exposure to microcystins through recreational water activities.

Toxicological studies using animal models have been used to provide further evidence of possible human health effects (Chorus et al., 2000). Microcystin-LR has been shown to be toxic following acute exposures in rodents. The oral (by gavage) LD_{50} is 5000 µg/kg body weight (bw) in mice and higher in rats (Fawell et al., 1999). Studies of acute and short-term exposures have demonstrated evidence of liver enzyme changes and tissue damage (inflammation, haemorrhaging, lesions) in mice as a result of exposure to microcystin-LR through oral administration (Fawell et al., 1999). Chronic exposures to low levels of microcystins have been shown to lead to progressive liver injury in experimental studies in pigs and mice (Chorus and Bartram, 1999). Microcystins are thought to be capable of promoting tumour development by interfering with the normal mechanisms of cell division. Evidence has been provided on the ability of microcystin-LR to promote the growth of certain types of tumours in mice subjected to prolonged exposure to the toxin through oral administration (Falconer, 2005). Presently there is insufficient evidence for the carcinogenicity or genotoxicity of microcystin-LR (Funari and Testai, 2008; Gaudin et al., 2009). However, the toxin has been listed as “possibly carcinogenic to humans” by the International Agency for Research on Cancer as a result of its tumour-promoting potential (IARC, 2010).

Both toxic and non-toxic species exist for all of the predominant microcystin-producing genera, and it is thought that different species or strains can vary in their toxic potential (Chorus and Bartram, 1999; Carillo et al., 2003; Dittmann and Börner, 2005). As a result a single bloom may consist of a mixture of non-toxic and variably toxic strains. It is generally felt that variations in toxicity within a bloom are due to the rise and fall of subpopulations of strains having different toxic potential. Environmental factors (e.g., conditions of winds, sunlight and temperature) are also thought to contribute to this phenomenon, but less significantly (Chorus and Bartram, 2003). Recently, the genes responsible for microcystin production (*mcy* genes) have been identified and sequenced (Falconer, 2005). Numerous studies have subsequently confirmed the detection of the

mcy gene cluster as a tool to discriminate between toxic and non-toxic strains of *Microcystis*, *Anabaena* and *Planktothrix* in both laboratory samples and field isolates (Dittmann and Börner, 2005).

Nodularins

Nodularins are hepatotoxins found in blooms caused by strains of the brackish-water species *Nodularia spumigenia*. The toxins are closely related to microcystins in both structure and function and thus act similarly by inhibiting protein phosphatase activity in liver cells (Chorus and Bartram, 1999).

Data derived from experimental studies, although limited, have suggested that nodularin exhibits toxicity similar to that of microcystin-LR. Results obtained from chronic toxicity studies using animal models have suggested that nodularins may be a more potent tumour promoter than the microcystins (Chorus and Bartram, 1999).

Nodularia blooms have been encountered in brackish lakes in Australia and New Zealand, as well as in the Baltic Sea. In general, blooms in fresh water are considered extremely rare (Chorus and Bartram, 1999). To date, there have been no recorded occurrences of *Nodularia* blooms in North American waters. As a result, they are not considered to be a significant public health threat in Canadian recreational waters.

Anatoxins

The anatoxins (anatoxin-a, anatoxin-a(S), homoanatoxin-a) are neurotoxins that can be found in blooms produced by *Anabaena* (anatoxin-a, anatoxin-a(S)), *Oscillatoria* (anatoxin-a, homoanatoxin-a) and *Aphanizomenon* (anatoxin-a). Anatoxins interfere with the activity of the nerve transmitter acetylcholine, which affects the functioning of the nervous system by disrupting communication between neurons and muscle cells (Chorus and Bartram, 1999). Acute toxicity is characterized by paralysis of both the skeletal and respiratory muscles, resulting in tremors, convulsions and, ultimately, death due to respiratory failure (Rogers et al., 2005).

Information on health effects associated with anatoxins has been gleaned from occasional accounts of animal poisonings and toxicological investigations. Anatoxin-a has been the most widely studied member of this group. Limited data are available for homoanatoxin-a and anatoxin-a(S) (Chorus and Bartram, 1999). However, it is generally regarded that although the toxins have somewhat different mechanisms of action, each is capable of causing fatalities at elevated doses. The oral LD_{50} of anatoxin-a is reported to be greater than 5000 µg/kg bw (Fitzgeorge et al., 1994). Anatoxin-a was shown to be toxic in experimental mice upon acute exposure to high levels of the toxin (15 000 µg/kg bw). Death was reported to occur within minutes (Chorus and Bartram, 1999). Conversely, in a number of experimental studies, recovery after exposure to sublethal doses of anatoxin-a was complete, with no signs of clinical toxicity. To date, there has been only one reported fatality resulting from exposure to cyanobacterial neurotoxins in natural waters. Exposure occurred through ingestion of contaminated water during accidental immersion at a location where swimming was not permitted (Falconer, 2005).

Blooms of anatoxin-producing species are not routinely reported in North American waters and are considered to occur far less frequently than those of the microcystin-producing

cyanobacteria. In addition, the anatoxins are relatively unstable and, as such, are not considered to be as widespread as microcystins in water supplies (Chorus and Bartram, 1999). As a result, anatoxins are currently considered to be of lesser concern than microcystins for Canadian recreational waters.

Cylindrospermopsin

Cylindrospermopsin is a relatively new cyanobacterial toxin named after the species from which it was first isolated: *Cylindrospermopsis raciborskii*. It is a member of the alkaloid family of toxins; however, the mode of action is considerably different from that of the anatoxins. Similarly, cylindrospermopsin does exhibit hepatotoxic activity, but operates by a mechanism much different from that of the other microcystins (Chorus and Bartram, 1999). The toxin has been isolated from certain species of a few other genera of cyanobacteria, including *Anabaena* and *Aphanizomenon* (Falconer and Humpage, 2006)

Cylindrospermopsin is considered to be a general cytotoxin, which acts by inhibiting protein synthesis. Cellular damage is caused by blocking the functioning of key proteins and enzymes. The liver and kidneys are considered the main targets for the toxin; however, crude extracts of *Cylindrospermopsis* given to mice have also demonstrated evidence of injury to other organs, such as the lung, spleen, thymus and heart (Chorus and Bartram, 1999). Data, derived from a study in which mice orally dosed with cylindrospermopsin demonstrated a positive but not statistically significant increase in the number of tumours, have provided evidence that the toxin may also possess carcinogenic properties (Falconer and Humpage, 2001).

The first recorded incident of human cylindrospermopsin poisoning occurred in 1979. Initially termed “Palm Island Mystery Disease,” this outbreak of hepatoenteritis among residents of a tropical island off the coast of Queensland, Australia, was later attributed to a bloom of *C. raciborskii* in the drinking water reservoir. Patients had reported symptoms that included vomiting, malaise, headache and constipation, later followed by bloody diarrhoea. Blood and urine analysis revealed evidence of liver and kidney damage; however, all patients were reported to have recovered following treatment. Follow-up research led to the identification of the toxin cylindrospermopsin produced by this cyanobacterial species. At present, there have been no human fatalities associated with cylindrospermopsin, and there have been no other recorded poisonings since the Palm Island outbreak (Chorus and Bartram, 1999).

Cylindrospermopsis is more frequently encountered in the warmer waters found in tropical and subtropical locations of the world. Blooms have been routinely encountered in freshwater lakes and drinking water reservoirs in Australia, South and Central America, and the state of Florida. To date, the organism has been only occasionally encountered in temperate fresh waters. In North America, populations have been detected in several northern U.S. states (Michigan, Ohio, Minnesota, Illinois and Indiana), as well as in the province of Manitoba.

There are several notable differences between populations of *Cylindrospermopsis* and those of cyanobacteria having microcystin-producing potential such as *Microcystis* or *Anabaena*. *Cylindrospermopsis* does not form slicks or scums of the type produced by these organisms. In *Cylindrospermopsis* blooms, the regions of highest cell concentrations are located beneath the surface (Falconer, 2005). As well, in these blooms, a significant portion of the toxin is released

into the surrounding water, in contrast to the situation in microcystin-producing species in which the toxin is predominantly retained within the cells and released only upon cell rupture or death (Falconer, 2005). Blooms of *Cylindrospermopsis* are considered infrequent in Canadian waters. Those caused by microcystin-producing species are far more prevalent. Nonetheless, the increasing frequency at which this organism is being detected in temperate fresh waters has resulted in the identification of *Cylindrospermopsis* blooms as a potentially emerging concern for recreational waters in Canada and the United States.

Saxitoxins

Saxitoxins are neurotoxins that belong to a larger family of toxins referred to as Paralytic Shellfish Poisoning (PSP) toxins. These toxins, originally isolated from shellfish having fed on toxic species of marine dinoflagellates, have also been isolated from several genera of freshwater cyanobacteria, including *Aphanizomenon*, *Anabaena*, *Cylindrospermopsis*, *Lyngbya* and *Planktothrix* (*Oscillatoria*) (Chorus and Bartram, 1999; Aráoz et al., 2010). Saxitoxins act by blocking ion channels in nerve and muscle cells, which prevents the transmission of electrical impulses. This can lead to neuromuscular paralysis, which can ultimately be fatal owing to the resulting respiratory failure.

Most of the toxicological information on saxitoxins has been obtained from studies using toxins produced by the marine dinoflagellates (e.g., *Alexandrium* spp.). Nevertheless, saxitoxin types derived from different sources remain identical in structure and toxicological profile (Funari and Testai, 2008). Saxitoxin is considered one of the most toxic of the PSP toxins (Chorus and Bartram, 1999). The oral LD_{50} reported for saxitoxin is 263 µg/kg bw (Mons et al., 1998; Funari and Testai, 2008). Animal deaths have been linked to contact with cyanobacterial blooms containing saxitoxin (Negri et al., 1995). There have been no saxiotoxin-related illnesses reported for humans through drinking water or recreational water exposure (Chorus and Bartram, 1999; Aráoz et al., 2009).

Saxitoxin-containing blooms are considered widespread in Australia, and toxic blooms have also been detected in waters in Brazil and in the southern and northern United States (Chorus and Bartram, 1999; dos Anjos et al., 2006). Presently, saxitoxins are not considered to be as significant a concern as microcystins in Canadian recreational waters. However, the detection of saxitoxins in blooms in temperate fresh waters suggests that this issue should continue to be monitored.

Dermatotoxins and other irritant toxins

Certain marine cyanobacteria such as *Lyngbya*, *Oscillatoria* and *Schizothrix* can produce toxins called aplysiatoxins and lyngbyatoxins, which have been reported to cause severe dermatitis among swimmers who come in contact with the cyanobacterial filaments. Aplysiatoxins are considered potent tumour promoters and are also thought to demonstrate other properties that may be linked to carcinogenesis (Chorus and Bartram, 1999). As these are primarily marine species, they are not thought to be of concern for freshwater lakes and rivers.

Although not well understood, it is also thought that the lipopolysaccharide component of the cyanobacterial cell wall can elicit an irritant or allergenic response in humans. Lipopolysaccharides have been known to exhibit pyrogenic (fever-inducing) and toxic

properties. It is generally regarded, though, that lipopolysaccharides from cyanobacteria are considerably less toxic than those of other Gram-negative bacteria, such as *Salmonella* (Chorus and Bartram, 1999). Nonetheless, it is thought that they may be at least partially responsible for some of the non-specific irritative effects associated with human exposure to cyanobacterial blooms.

Compound of interest: B-methylamino-L-alanine (BMAA)

An emerging topic of interest involves the unusual amino acid B-methylamino-L-alanine (BMAA), its links to cyanobacteria and the research findings concerning its potential neurotoxic capabilities. The present evidence does not suggest that BMAA is a recreational water quality hazard of human health concern. Information is provided on the current state of evidence on this issue.

BMAA was first identified during exploratory studies into the high rate of amyotrophic lateral sclerosis/parkinsonism–dementia complex (ALS-PDC, a neurodegenerative disease with symptoms similar to Parkinson's disease and Alzheimer's disease) observed among the Chamorro people of Guam. Findings reported by the researchers during investigations into this issue included that BMAA could be detected in brain tissues of ALS-PDC patients in Guam and Canada (Cox et al., 2003), and that BMAA could be found in virtually all groups of cyanobacteria including the notable freshwater genera: *Anabaena*, *Aphanizomenon*, *Microcystis*, *Nodularia* and *Oscillatoria* (Cox et al., 2005; Metcalf et al., 2008). The further observation that BMAA could be detected in flying foxes (i.e., bats, which constitute a portion of the Chamorro diet) led the researchers to hypothesize that BMAA could be subject to magnification up the food chain.

These findings and the proposed implications are currently the subject of debate (CRC 2005). Some researchers have provided conflicting evidence regarding BMAA detection in brain tissue of ALS-PDC patients (Montine et al., 2005), while others have questioned the data relating to neurotoxicity and food chain magnification (Duncan and Marini, 2006).

Research into this issue is considered preliminary, requiring further investigation. It is proposed that much more work is needed before a cause and effect relationship between BMAA and neurological disease can be established or discounted (CRC, 2005). Similarly, there is insufficient evidence at this time to suggest that water supplies or dietary sources could constitute a significant source of BMAA exposure. Developments on this topic will continue to be monitored.

Guideline rationale

The recommended guideline values for total cyanobacteria and total microcystins in Canadian recreational waters are based upon the approach used in the derivation of the maximum acceptable concentration (MAC) for microcystin-LR in the *Guidelines for Canadian Drinking Water Quality* (Health Canada, 2002). The use of single guideline values for total cyanobacterial cell density and total microcystin concentration is the preferred approach to setting guidelines for Canadian recreational waters at this time. They are intended to protect against both the risk of exposure to microcystins through inadvertent ingestion of water as well as from other harmful effects that may be possible after exposure to high densities of cyanobacterial material. In this

situation, values were derived based on children as they may be more intensely affected because they spend more time in water than adults and are more likely to accidentally swallow contaminated water. The guideline value for total microcystins (expressed as microcystin-LR) is intended to be protective against exposure to other microcystin variants that may be present.

For microcystin-LR, the tolerable daily intake (TDI) for recreational water exposures is derived as follows:

$$\text{TDI} = \frac{40\ \mu\text{g/kg bw per day}}{100} = 0.4\ \mu\text{g/kg bw per day},$$

where:

- 40 µg/kg bw per day is the NOAEL for liver changes derived from the 13-week mouse study conducted by Fawell et al. (1999); and
- 100 is the uncertainty factor (×10 for intraspecies variation, ×10 for interspecies variation).

An additional uncertainty factor for less-than-lifetime study was not considered necessary, as the types of exposures being considered are short term and episodic in nature.

The guideline value for total microcystins is calculated from the TDI for microcystin-LR as follows:

$$\text{Guideline value} = \frac{0.4\ \mu\text{g/kg bw per day} \times 13\ \text{kg bw}}{0.25\ \text{L/day}}$$

$$\approx 20\ \mu\text{g/L for total microcystins}$$

where:

- 0.4 µg/kg bw per day is the TDI, as derived above;
- 13 kg bw is the average weight of a child aged 7 months to 4 years (Health Canada, 1994); and
- 0.25 L/day (250 mL/day) is the estimated amount of water accidentally ingested per day during recreational water activities by a child.

The value used for the amount of water accidentally ingested per day is probably a conservative estimate. Risk assessment models developed for enteric pathogens in recreational waters have typically assumed a value of 100 mL for the amount of water likely to be swallowed through recreational water activities (Haas, 1983; Gerba et al., 1996; Mena et al., 2003). Yet the basis for this appears to be more historical than empirical. WHO (2003a) suggests that a child may be capable of consuming 250 mL of water during extensive playing. During an empirical investigation of the volume of water swallowed during swimming activities, Evans et al. (2006) reported average values of 24 mL for adults (95% range, 2–84 mL) and 47 mL for children (95% range, 3–142 mL). The authors further observed that some swimmers swallowed up to 280 mL/hour.

In addition to being capable of initiating toxic effects through ingestion, microcystins have been suspected to be associated with other irritative effects that can occur through recreational water

contact, such as injuries to the tissues of the mouth and lips. As a result, it is possible that a more conservative estimate is appropriate as a worst-case scenario to account for the total amount of water ingested, and not just the portion accidentally swallowed. The value of 250 mL represents a risk management decision, derived based on the assessment of the available information regarding the likely risk of ingestion and of the potential risks for the recreational water user.

The guideline value for cyanobacterial cell density (expressed as total cyanobacteria) represents a general indication of the potential for bloom development and is intended to be protective against exposure to high densities of cyanobacterial material. It similarly may be used to provide protection against exposure to blooms of other cyanobacteria with toxic potential, and not just the microcystin-producing species. It is calculated based on the microcystin guideline value using the recognized reference value for the average toxin quota for *Microcystis* cells and reflects the highest likely water quality hazard scenario of a toxic bloom containing high levels of microcystin (WHO, 2003a; NHMRC, 2008):

$$\text{Guideline value} = \frac{20\ \mu g/L \times 10^{-3}\ L/mL}{2 \times 10^{-7}\ \mu g/cell}$$

$$= 100\ 000\ \text{cells/mL for total cyanobacteria}$$

where:

- 20 µg/L is the guideline value for total microcystins in recreational waters, as derived in the previous section;
- 2×10^{-7} µg/cell is the toxin cell quota for total microcystins per cell (WHO, 2003a; NHMRC, 2008); and
- 10^{-3} L/mL is the factor converting litres to millilitres.

Currently, there is insufficient evidence to derive recreational water guidelines for other cyanobacterial toxins, such as the anatoxins or cylindrospermopsin. Based on the assessment, it was similarly determined that there is not sufficient data upon which to base a guideline whose intent is to provide protection from the risk of allergenic or other irritative effects caused by unknown cyanobacterial substances. Studies conducted by Pilotto et al. (1997, 2004) on the effects of human skin contact with cyanobacteria were reviewed. Both studies reported that skin contact with cyanobacteria across a wide range of cell densities resulted in reactions of only mild severity in only a very small proportion of subjects. No dose–response relationships or thresholds for irritation could be observed. It was judged that these findings did not provide sufficient argument for the need for a separate guideline.

This approach and guideline value for microcystin are consistent with criteria derived by the province of Québec for microcystin-LR toxic equivalents in recreational waters: 16 µg/L (INSPQ, 2004). Further, the guideline value for total cyanobacteria is consistent with the WHO guideline of 100 000 cyanobacterial cells/mL for the moderate probability of adverse health effects. In its supporting documentation, WHO (2003a) further proposes that at this

cyanobacterial density, a concentration of 20 µg microcystin/L is likely if the bloom consists of *Microcystis* with an average toxin content of 2×10^{-7} µg/cell.

The Canadian approach is also similar in principle to the Australian Level 1 Guidelines (NHMRC, 2008); however, a different animal model and derivation process have been used in the Canadian calculations. The Australian approach was based on data considered to be the most suitable for deriving guidelines for toxic cyanobacteria in Australian recreational waters.

Occurrence in the environment

In a 1995 survey of 16 recreational water bodies in southern Manitoba, Jones et al. (1998) reported that microcystin-LR could be found at 44% of the sites, with concentrations occurring in the 0.1–0.6-µg/L range. The authors further reported that cyanobacterial cell density was not a good predictor of microcystin-LR and that there was no correlation between toxin concentration and the other environmental variables monitored in the study.

Kotak et al. (1996) reported on a 3-year investigation that measured microcystin concentrations encountered in phytoplankton and surface water samples collected from four freshwater lakes in central Alberta. Phytoplankton concentrations of microcystin were found to be highest in the two most hypertrophic lakes (Driedmeat and Little Beaver lakes). The highest microcystin concentrations were observed in August and September, and were reported to correspond to periods where *Microcystis* cell counts were at their highest (>200 000 cells/mL). Levels reported during these months at one of the lakes (Driedmeat) ranged from 1.2 to 6.1 µg/L. The highest microcystin concentration observed during the course of the study was 11 µg/L in a sample collected from Little Beaver Lake in mid-August.

Giani et al. (2005) reported total microcystin concentrations of 0.008–1.91 µg/L (mean, 0.140 µg/L) in water samples collected during a survey of 22 freshwater lakes in southern Québec. None of the lakes were reported to be affected by cyanobacterial blooms at the time of sampling. The authors reported that all lakes contained detectable levels of cyanobacterial genera having toxic potential (including *Microcystis*, *Anabaena* and *Oscillatoria*), although these were typically encountered at concentrations considered far below levels of concern.

Rinta-Kanto et al. (2005) reported on the characterization of two separate bloom events that had developed in the western basin of Lake Erie: the first in August 2003 and the second in August 2004. Satellite imaging was used by the researchers to detect the presence of the blooms and was similarly applied to identify potential sites for analysis. Bloom samples were collected and analysed for total microcystin concentrations (cell-bound fraction). At the same time, estimates of the presence of total and toxic *Microcystis* concentrations were determined by QPCR analysis, with targets specific for a *Microcystis* 16S rDNA gene fragment and the microcystin synthase gene *mcy*D. Microcystin concentrations encountered in the 2003 bloom ranged from <0.3 to 15.4 µg/L, and estimates of the corresponding *Microcystis* concentration ranged from below the quantifiable limits of detection to a peak value of 3.9×10^8 *Microcystis* equivalents/L. In the bloom encountered the following year, microcystin concentrations ranged from 0.1 to 2.6 µg/L, and estimates of the total *Microcystis* concentration spanned from approximately 5×10^3 to 3×10^6 *Microcystis* equivalents/L.

Related epidemiology

No human cyanotoxin-related deaths have been associated with exposure via recreational water. To date, there have been only two reported instances of human fatalities as a result of exposure to cyanobacterial toxins. The most notable are those associated with the human dialysis tragedy that occurred in Caruaru, Brazil, in 1996 (Jochimsen et al., 1998). In total, 56 patients at a haemodialysis clinic died as a result of liver failure from exposure to microcystins in the water used for dialysis. The presence of toxic cyanobacteria in the city's drinking water reservoir and insufficient water treatment were cited as major factors contributing to the deaths. More recently, in 2002, a Wisconsin teenager died following exposure to a toxic scum in a pond on a public golf course where swimming was not permitted (Falconer, 2005). Blood and stool samples collected revealed the presence of both the cyanobacterial species *Anabaena flos-aquae* and the neurotoxin anatoxin-a.

There have been few reported accounts of human illness as a result of contact with toxic cyanobacterial populations in recreational waters worldwide. The predominant health effects encountered from such exposures may be gastrointestinal or flu-like in nature and thus may often go unreported or are attributed to other causes (Falconer, 2005). As well, most of the cases of illness are investigated retrospectively; thus, information regarding the exact conditions of exposure (number and type of organisms, identification and/or concentration of toxins) is rarely available.

In Saskatchewan in 1959, despite warnings and reports of livestock deaths, people swam in a lake infested with cyanobacteria. Thirteen individuals in total became ill, reporting symptoms of nausea, muscular pain, headache and diarrhoea (Dillenberg and Dehnel, 1960). In England in 1989, 10 of 20 soldiers became ill after canoe training and swimming in water affected by a *Microcystis* bloom. Swimming skills and the amount of water ingested were reported to have been related to the degree of illness (Turner et al., 1990). In 1991, in Australia, two teenage girls developed gastroenteritis and muscle pains after swimming in the Darling River near Wilcannia during a cyanobacterial bloom containing *Anabaena* (NHMRC, 2008). In the same year, two cases of conjunctivitis and one of dermal irritation were attributed to swimming during an *Anabaena circinalis* bloom in Lake Cargelligo, Australia (NHMRC, 2008).

Pilotto et al. (1997) reported on the results of a prospective epidemiological study designed to investigate the health effects encountered in swimmers after contact with cyanobacteria in recreational water. Symptoms were shown to increase significantly with the duration of water contact and increasing cyanobacterial cell density, but did not correlate with cyanotoxin concentrations. Participants in this study who were exposed to 5000 cells/mL for more than 1 hour had a higher rate of symptom occurrence than did unexposed participants. However, after controlling for previous exposure and prior illnesses, the data suggested that only a small number of people were affected, and only with mild irritation. Stewart et al. (2006b) conducted a prospective epidemiological study to examine the incidence of acute symptoms in recreational water users exposed to different amounts of cyanobacteria based on measurements of surface area per volume (SA/V). The authors observed that symptoms were more likely to be reported among persons exposed to high (>12 mm^2/mL) versus low (<2.4 mm^2/mL) SA/V levels, and that respiratory symptoms were recorded with the greatest frequency. Cyanotoxins were detected only occasionally and at low SA/V levels during the study.

Managing health risks

A multi-barrier approach is considered the best strategy to reduce the risk of exposure to cyanobacteria or their toxins in recreational waters. This approach combines the use of the recommended indicators of water quality alongside actions both to reduce the extent of the water quality hazard and to restrict swimmer exposures during periods or in areas perceived to be of increased risk.

The occurrence of cyanobacterial blooms in recreational waters is extremely difficult to predict. Bloom development is influenced by a variety of physical, chemical and biological factors. As a result of the interplay of these factors, there may be large year-to-year fluctuations in the levels of cyanobacteria and their toxins (Health Canada, 2002). Managed recreational water areas that are suspected or are known to be susceptible to blooms should be routinely monitored during the bathing season. Authorities should visually monitor such supplies for cyanobacterial growth. Toxicity within a bloom can vary considerably, particularly within large blooms. Further, a water body in which a bloom has developed may still contain toxins for a short period after the bloom has dissipated. In order to fully characterize the extent of the risk posed by the cyanobacterial population, authorities should conduct sampling during, and after the collapse of, the bloom. Collection of multiple samples may be required to identify any regional or localized differences in cyanobacterial cell density and toxin concentration. It is advised that both total cyanobacterial cell densities and total microcystin concentrations be monitored as part of a risk management strategy for cyanobacteria and their toxins in Canadian recreational waters. As discussed, monitoring of microcystin levels is necessary to determine the potential health risk posed by a cyanobacterial population, while cell count determinations are useful for providing a general indication of the potential for bloom development and, thus, the possible presence of cyanobacterial toxins.

Waters shown to exceed the established guideline values or those in which a bloom has developed may result in human exposure to cyanobacterial material or cyanotoxins in amounts sufficient to be harmful to human health. A swimming advisory may be issued at the discretion of the responsible authority. Contact with waters where an advisory has been issued should be avoided until the advisory has been rescinded. Further information on the posting of recreational waters can be found in Part I (Management of Recreational Waters).

Other barriers for risk reduction can include the provision of educational materials outlining steps the public may take to reduce their personal risk in the event of a bloom. Guidance provided in materials for public communication may include the following recommendations:

- Recreational water users should avoid areas with visible cyanobacterial blooms and/or scums, as all blooms have the potential to be toxic. Direct contact with bloom material or accidental ingestion of contaminated waters can be harmful to recreational water users. Inhalation may also be an important route of exposure to cyanobacterial toxins during activities in bloom-affected areas where aerosols might be generated.

- Users should shower or wash themselves as well as any item that may have accidentally come into contact with cyanobacterial material as soon as is practical upon exiting the water.

- Any user experiencing adverse health effects from recreational water activity should consult a medical professional and, if necessary, alert the appropriate authorities.

- Users should not let pets swim in or drink from areas where the water has taken on an abnormal discolouration consistent with that of a bloom, or where accumulations of cyanobacterial material are visible. Should pets come in contact with bloom-affected waters, they should be rinsed off immediately to remove all traces of cyanobacterial material that may accidentally be ingested.

The use of algicides is not recommended as a measure to control cyanobacterial populations. The addition of copper sulphate or other algicides to mature toxic blooms may have the effect of destroying the cyanobacterial cells; however, this action may also cause the release of significant amounts of cyanotoxin into the surrounding waters if present within the cells. Jones and Orr (1994) reported that microcystin-LR could be detected up to 21 days after algicide treatment of a toxic *Microcystis aeruginosa* bloom that had developed in a recreational lake. Environmental concerns have also been cited as additional reasons for not pursuing this approach, as the algicides can be detrimental to the healthy functioning of the aquatic ecosystem.

Longer-term risk management actions that may also be considered as barriers to reduce the impact of toxic cyanobacteria in surface waters may involve the identification of the major nutrient inputs, as well as the development of strategies to reduce nitrogen and phosphorus loading through effective control of agricultural, municipal sewage and residential waste disposal practices. Tracking total phosphorus levels in surface waters has been suggested as a proactive step for recognizing water bodies that have the potential for bloom development (Chorus and Bartram, 1999).

Summary

1. The development of cyanobacterial blooms in waters that are used for recreational purposes is dependent on many factors that can be difficult to predict. Blooms can develop very rapidly under the appropriate conditions, and lakes that have never had a problem can suddenly become toxic.

2. Serious swimmer illnesses have been reported following exposure to toxic cyanobacterial blooms in recreational waters. A water body containing a cyanobacterial bloom may still contain toxins after visible signs of the bloom have disappeared. In general, contact with waters where a bloom exists or has very recently collapsed should be avoided. Contact with waters where an advisory has been issued should be avoided until the advisory has been rescinded.

3. To date, there have been very few reported cases of cyanobacterial toxin-related illness due to recreational water activity in Canadian or international waters. The combination of visual inspection, water quality monitoring and public notification and education alongside actions and procedures for reducing nutrient inputs represents the most effective approach to protecting the health of recreational water users.

7.0 Other biological hazards

Guideline values
No guideline values can be specified for any of the organisms identified in this section. In general, areas used for recreational water purposes should remain as free as is practical from these organisms. Furthermore, recreational water activities should not be pursued in waters where the responsible authority deems their presence poses a risk to the health and safety of the users.

This section provides guidance on other organisms that may affect the recreational value of natural waters by rendering them unsafe, aesthetically objectionable or otherwise unusable, which interferes with the health, physical comfort or enjoyment on the part of the user. These organisms are free-living species that are considered to occur naturally in recreational waters. Guidance is provided, as both the authorities responsible for managing recreational waters and the general public should remain informed of both the possible risks posed by these organisms and the steps that can be taken to reduce potential exposures. This list is not intended to be exhaustive; thus, the responsible authorities may wish to provide information on other organisms of regional or local significance.

7.1 Schistosomes causing swimmer's itch

Description
Swimmer's itch (cercarial dermatitis) is caused by human reaction to dermal penetration by parasitic flatworms or "schistosomes" that infect certain waterfowl and aquatic rodent species (Manitoba Water Stewardship, 2007). These schistosomes belong to the family Schistosomatidae, and species implicated as causes of swimmer's itch include members of the genera *Austrobilharzia* and *Trichobilharzia* (Levesque et al., 2002; CDC, 2004a). Cercarial dermatitis should be clearly distinguished from human schistosomiasis, a far more serious human infection that is caused by species of the genus *Schistosoma* and is typically restricted to tropical regions of the world (WHO, 2003a).

The schistosomes that cause swimmer's itch have a two-host life cycle, consisting of a primary host (waterfowl or aquatic rodents) and an intermediate host (certain species of aquatic snails). The adult parasite lives in the bloodstream of infected geese, swans, ducks, gulls, muskrats and beavers and produces eggs, which are passed in the faeces of these animals. These eggs hatch in the water, releasing small free-swimming larvae called miracidia. These larvae then seek out a suitable snail host to infect in order to continue their life cycle. Within the snail, the parasite further develops into a different type of larva called cercariae, which are released into the water when conditions are appropriate. The free-swimming cercariae once again seek out a suitable bird or animal host to begin their life cycle all over again. Humans are an accidental or dead-end host for these organisms. If the cercariae accidentally encounter humans in the water, they may penetrate the outer layer of the skin, but quickly die, as they cannot develop any further. The presence of the cercariae beneath the skin causes an allergic reaction to develop (i.e., cercarial dermatitis), which accounts for the symptoms observed in infected swimmers.

Health effects

The effects of swimmer's itch may be felt shortly after swimming, in some cases in as little as a few minutes. Swimmers usually first experience a tingling, itching or burning sensation. Small, reddish pimples typically may appear within 12 hours after infection, and these can progress to larger secondary blisters or rashes, which can be accompanied by an even stronger itching sensation. The infection is self-limiting, typically lasting from 2 to 5 days; however, symptoms can persist for as long as 2 weeks. Swimmer's itch is not contagious and cannot spread from person to person. However, because swimmer's itch is caused by an allergic reaction, individuals can develop an increased sensitivity to subsequent infections. The symptoms become more intense and develop much more rapidly in these instances (British Columbia Ministry of Health, 2005). Sensitivity can vary considerably between different individuals; some may strongly show the effects of infection, whereas others may not show any signs of illness.

Although affected individuals are advised to seek medical treatment from a health professional, treatments that may be effective in combatting itching include the application of cold compresses; the use of anti-itch medications such as corticosteroid creams or calamine lotion; and the taking of oral antihistamines or bathing in baths containing Epsom salts, baking soda or colloidal oatmeal (British Columbia Ministry of Health, 2005; Manitoba Water Stewardship, 2007). It is recommended that affected individuals refrain from scratching, as it increases the potential for secondary bacterial infection (CDC, 2004b).

Occurrence in the environment

These schistosomes can be encountered in fresh waters and at coastal beaches throughout Canada and the northern United States. Indeed, swimmer's itch has been reported in virtually every Canadian province. Accidental introductions of host snail species are reported to have spread this parasite from the Atlantic coast of North America to British Columbia coastal waters (Leighton et al., 2004). Bird species (ducks and gulls) that play host to the parasite may also be expanding in geographic range (Verbrugge et al., 2004). Reports of incidents appear to be increasing in the United States and Canada, possibly reflecting increasing use of recreational water bodies.

The presence of the organisms in natural waters is dependent upon a number of factors, both biological and environmental. As a result, it is very difficult to predict when and where swimmer's itch might become a problem. Propagation of the organisms requires that both the primary and intermediate host be present in sufficient numbers. As well, the species involved do not all follow the same timetable in terms of infection of, and release from, their primary and secondary hosts. Thus, the organisms can be encountered at different times during a recreational water season (Michigan Department of Environmental Quality, 2005).

The cercariae are encountered in areas where the snail beds are the densest. These are typically shallow waters, particularly those with large numbers of aquatic plants. Water temperature is also thought to have a significant effect on the release of mature cercariae by infected snails. Cercarial production and concentrations are thought to increase in warmer waters (Verbrugge et al., 2004), which may partially explain why infections are encountered more frequently during the summer months. The organisms can be carried significant distances by wind and wave

action. Persistent onshore winds may drive them to accumulate at the shoreline, whereas sheltered bays may act to retain the organisms within a localized area.

Related epidemiology

Much of the information on swimmer's itch infection has come from case reports of human illness. For most recreational waters in Canada, the risk of contracting swimmer's itch through recreational activity is considered to be quite low. However, many cases go unreported, as the symptoms are typically benign and thus users may not seek out medical attention.

Levesque et al. (2002) conducted an investigation of an outbreak of cercarial dermatitis that occurred on Lac Beauport, a recreational lake in the Quebec City region in the summer of 1999. A case reporting form was sent to 450 families likely to have activities that would bring them into contact with the lake's water. Snails were characterized, and the prevalence of their infestation by schistosomes was investigated. In total, 63 episodes consistent with cercarial dermatitis were reported, with the symptoms affecting mainly children under 10 years of age. Sixty-nine percent of the cases occurred from swimming at the same beach. This location was the one where the only population of snails in the lake was identified. The people most affected were those who bathed in shallow water along the shoreline. Mallard ducks were observed to be present in high numbers during the 1999 summer season. Concentrations of faecal coliforms, faecal streptococci and other bacteriological water quality indicators at the beaches were low. Based on all of the available evidence, the authors confirmed that the cases were indeed due to schistosomes. Shoreline residents were informed that they should not feed waterfowl, and snail populations were reduced by removing organic wastes found within the main snail habitat. The control mechanisms were thought to have been effective, as there were no reported instances of cercarial dermatitis at this location during the following season.

Another study by Leighton et al. (2004) looked at case reports and the biological factors contributing to two outbreaks of dermatitis at Crescent Beach near Surrey, B.C. Thirty-six cases of dermatitis were reported in the summer of 2001, and 44 more cases were reported in the summer of 2002. The clinical presentation was consistent with schistosome dermatitis or swimmer's itch. The causative agent was identified as the schistosome cercaria, *Austrobilharzia variglandis*, carried by the introduced host snail, *Ilyanassa obsoleta*. Many of the cases of swimmer's itch appear to have occurred after exposure to the parasite in the shallow tide pools of the upper beach where there are large aggregations of the snails and frequent human use of the beach for wading. A survey of schistosome infections in snails demonstrated that infected snails were present in most areas of the beach. The reasons for the sudden outbreak were unclear. Both the snail host and the schistosome species had been known to be present at this location for several years. Factors attributed to the sudden outbreak included increased beach use by recreational users, seasonal environmental factors (temperature, weather), the age of the snail population and the size of the host population.

Verbrugge et al. (2004) conducted a prospective epidemiological study to assess the incidence and severity of swimmer's itch among recreational water users at Douglas Lake, Michigan, in July 2000. In total, 301 subjects were included in the analysis. Data were collected on 1300 water exposure days for the 301 swimmers, and 89 episodes of swimmer's itch were recorded (corresponding to an incidence of 6.8% per water exposure day). A total of 52 people

(17.3%) experienced swimmer's itch, with 58% of these having only one episode, 25% having two episodes and 17% having three or more episodes. There was a highly significant association of swimmer's itch with shallow water use. Risk was also shown to increase with the number of days of exposure reported. Higher incidences were observed in the southern and eastern zones of the lake, and it was reasoned this was at least partially due to persistent onshore winds and the existence of sheltered bays in these areas.

Managing health risks

The schistosomes responsible for causing swimmer's itch are considered to occur naturally in Canadian surface waters. They are not related to faecal pollution; as a result, their presence is not indicated during standard water quality testing for the recommended indicators of faecal contamination. The factors that are necessary for swimmer's itch to become a problem in recreational areas are subject to constant change. Certain areas may report a problem where none appeared to exist previously. Similarly, areas in which swimmer's itch has been reported will not necessarily always remain a problem.

A management strategy combining both actions to control the extent of the water quality hazard and steps to limit exposure during periods or in areas perceived to be of increased risk is recommended to reduce the risk of human exposure to these schistosomes in recreational waters.

Warning signs that clearly notify the public of the risk of exposure should be posted at recreational water areas where cases of swimmer's itch have been reported. Additionally, a swimming advisory may be issued at the discretion of the responsible authority. Further details on the posting of information at recreational water areas can be found in Part I (Management of Recreational Waters).

Another approach to risk reduction can include the provision of educational materials outlining steps the public may take to reduce their personal risk of exposure as well as to potentially reduce the severity of the symptoms of infection. Guidance provided in materials for public communication may include the following recommendations:

- Recreational water users should avoid areas where swimmer's itch has been reported or where signs have been posted warning users of a potential risk of infection.

- At locations where swimmer's itch may be suspected, users should particularly avoid areas where schistosomes are more likely to be encountered, such as shallow waters with large quantities of aquatic plants.

- Users should towel down briskly upon leaving the water. Showering as soon as is practical after recreational water activity is also recommended to help minimize the risk of infection.

- Any user experiencing adverse health effects from recreational water activity should consult a medical professional and, if necessary, alert the appropriate authorities.

- Users are also reminded not to feed any waterfowl that may be present, as these animals can harbour the organisms responsible for causing swimmer's itch.

7.2 Aquatic vascular plants and algae

Description

Aquatic vascular plants (macrophytes) and algae can affect recreational water uses. It is difficult to estimate the magnitude of the adverse effects of these organisms in terms of their degree of interference with recreational pursuits or the potential risks to health posed to recreational water users.

The presence of these organisms can present a safety risk to recreational water users. Swimmers may become entangled in the fronds of aquatic plants. Dense growths can obstruct the view of the bottom and underwater hazards and may impair the ability of safety personnel to see persons in distress. Algal matrices attached to rocks and other substrata (i.e., periphyton) can cause slippery conditions that may lead to unintentional immersions or injuries.

Excessive growth of the organisms can also create aesthetic problems for recreational water areas. Macrophytes can assume high population densities and make nearshore and shallow regions unsuitable for any purpose (Priyadarshi, 2005). Dislodged rafts or mats of plants and algal material can be washed ashore and left to rot, fouling beaches. In addition to being unsightly, these masses can further detract from user enjoyment by producing offensive odours and restricting access to shorelines. They may also pose a public health risk, as they can attract undesirable animals to the area and provide breeding grounds for a variety of species of insects and bacteria (Whitman et al., 2003). The most notorious organism in this respect has been the green algal species *Cladophora* (Priyadarshi, 2005). There have been countless accounts of beaches and shorelines fouled by rotting, stinking masses of this alga. Recent research has indicated that *Cladophora* mats can provide a secondary habitat for bacteria that could potentially influence water quality in affected swimming areas (Whitman et al., 2003; Ishii et al., 2006b).

Increased plant growth (both macrophytes and algae) can be caused by the presence of excess nutrients. Various nutrient sources, including agricultural practices, domestic sewage and industrial effluent, all increase the amount of phosphorus and nitrogen in aquatic systems. The result of this increase in nutrients is called cultural eutrophication. Impaired water quality as a result of high algal populations and eutrophicating conditions can reduce recreational opportunities (Chambers et al., 2001). Canadian Water Quality Guidelines have been developed for both phosphorus and nitrogen species to protect the aquatic environment from nutrients and their effects on aquatic organisms (CCME, 1999).

Managing health risks

Excessive numbers of aquatic plants and algae should be absent from areas intended for use as swimming beaches. Similarly, recreational activities should not be pursued in areas where these organisms are present in quantities such that the responsible authorities deem that their presence poses a potential health or safety risk to recreational water users. It is recommended that an Environmental Health and Safety Survey be conducted at the start of each swimming season to identify potential safety hazards that may be encountered at a given recreational water area. Subsequently, one barrier to risk may involve the posting of signs warning users of potential

visibility or entanglement risks that may be posed by these organisms. Further information on the posting of warning signs can be found in Part I (Management of Recreational Waters).

Many aquatic plants and algae also provide an important habitat for fish and other aquatic biota. Management actions that try to remove these oplant rganisms from natural waters are discouraged, as removal may be harmful to the aquatic environment and is generally not effective from both a practical (plants quickly repopulate) and economical perspective. The application of pesticides to combat these organisms is similarly not recommended, as their use may create a health hazard for recreational water users if used incorrectly, and they are also detrimental to the healthy functioning of the aquatic ecosystem.

Improved beach cleanup procedures to remove masses of plants and algal material that may have washed up on shorelines represents another barrier that can be effective in reducing potential risks to recreational users. Longer-term actions that may also be considered as barriers to reduce the impact of these organisms can involve the identification of the major nutrient inputs within the watershed and the subsequent implementation of strategies for their control.

7.3 Additional organisms

Numerous other organisms can interfere with the safe and enjoyable use of recreational waters in Canada. For example, at some coastal beaches, jellyfish can cause painful and possibly serious stings for recreational water users who come in contact with them. Similarly, leech “bites” can occur in leech-infested areas, and sea urchins and mussel shells can cause painful injuries when stepped on by users of recreational waters. As these organisms are often of local or regional significance, it is recommended that, where necessary, the responsible authority provide the appropriate guidance to recreational water users on these subjects. This may include providing information on the potential risks posed by these organisms as well as steps for reducing the public’s individual risk of exposure.

Summary

1. Recreational water activities should not be pursued in areas where any of the organisms described in this section are known to be present in quantities such that they are deemed to pose a potential health or safety risk to recreational water users. An Environmental Health and Safety Survey is an important tool for identifying water quality hazards that may be encountered within the recreational water area.

2. Effective barriers to protect users from safety risks that may be encountered during recreational activities can include the provision of warning signs or educational materials informing users of the potential risks posed by a given water quality hazard, steps to reduce their risk of contact and possible actions to be taken in the event that exposure does occur.

8.0 Physical, aesthetic and chemical characteristics

This section describes the major physical, aesthetic and chemical characteristics of water that may affect recreational water bodies. Information is provided as to the potential effects each may have on the safe, enjoyable use of recreational waters. Guideline values or aesthetic objectives

have been provided where possible. It is intended that the values and associated guidance apply to all recreational waters, regardless of the types of activities practised. Responsible authorities may wish to establish separate guideline values or aesthetic objectives for waters not intended for primary contact use at their discretion.

Methods for determining physical, aesthetic and chemical characteristics of recreational waters can be found in *Standard Methods for the Examination of Water and Wastewater* (APHA et al., 2005). Sampling for these parameters is at the discretion of the responsible authority, however it is suggested that valuable times may include:

- when conducting the EHSS;
- at the start of, and at regular intervals during the swimming season,
- during area assessments conducted in response to recreational water issues, as appropriate.

8.1 Physical characteristics

8.1.1 pH

Guideline values
Both alkaline and acidic waters may cause eye irritation. To be protective against the risk of eye irritation, the pH of recreational waters should be in the range of 5.0 to 9.0.

Description
Mood (1968) concluded that exposure to water is foreign to the eye and may, under certain circumstances, be very irritating. He assumed that the ideal, non-irritating solution would have the same physicochemical properties as tears, including a pH of 7.4, although there is some evidence to suggest that ophthalmological solutions slightly more alkaline are actually preferred (Raber and Breslin, 1978).

Mood (1968) reported that tears have the capacity to rapidly neutralize an unbuffered solution from a pH as low as 3.5 or as high as 10.5. The neutralizing capacity of the tears would be exceeded by highly buffered waters. However, Mood (1968) concluded that unbuffered waters are not found in nature under normal conditions; hence, he suggested that the pH range for water with low buffering capacity should be between 5.0 and 9.0. Dillon et al. (1978) reported that most lakes in south-central Ontario have 10–200 µeq/L of acid-neutralizing capacity (ANC), and many of these lakes have depressed pH.

Studies completed by Basu et al. (1984) used water from two inland lakes in Ontario: Clearwater Lake (pH approximately 4.5), with an ANC of −40 µeq/L (Ontario Ministry of the Environment, 1980), and Red Chalk Lake (pH approximately 6.5), with an ANC of 70 µeq/L. The eyes of both test rabbits and human volunteers were exposed to these waters, and no significant differences were observed in their reactions (Basu et al., 1984). Basu et al. (1984) exposed one eye to the low-pH water and the other to the higher-pH water. The human eyes were exposed for 5-minute periods, and no ocular effects were noted. The rabbit eyes were exposed for periods of 15 minutes and checked for ocular reactions in terms of conjunctival congestion, corneal epithelial staining with fluorescein, epithelial cell and leukocyte content of tears, change in tear molarity and the penetration of fluorescein into the anterior chamber. Basu et al. (1984) concluded that the

exposure of healthy eyes to lake water having a pH as low as 4.5 is not harmful to the external ocular tissues.

8.1.2 Temperature

Guideline value
Precise guideline values for the temperature of waters to be used for swimming cannot be established. Tolerance to water temperatures can vary considerably from individual to individual. Users should not engage in recreational activities at temperature-time combinations sufficient to cause an appreciable increase or decrease in their core body temperature.

Description
Cold water exposures
Water is very efficient at conducting heat away from the body. When in water, unlike air, the surface area available for heat exchange reaches close to 100% (Transport Canada, 2003). Water has 25 times the thermal conductivity of air and cools the body 4–5 times faster than air of the same temperature (Tipton and Golden, 2006).

The definition of cold water must be considered with respect to normal body temperature, duration of exposure and degree of protection by insulation (Canadian Red Cross, 2006). When heat loss exceeds heat production, the temperature of the body will drop below the normal value of 37°C (Tipton and Golden, 2006). Water, even at comfortable levels, can result in the transfer of heat away from the body. Fatalities from cold water immersion have been reported in subtropical waters in which exposure was prolonged. Thermal neutrality in water is reported to occur at 35°C. Below this value, the human body is expected to lose more heat than it is capable of producing. Sudden, unprotected immersion in water of ≤ 15°C is considered to be a potentially life-threatening situation (Canadian Red Cross, 2006).

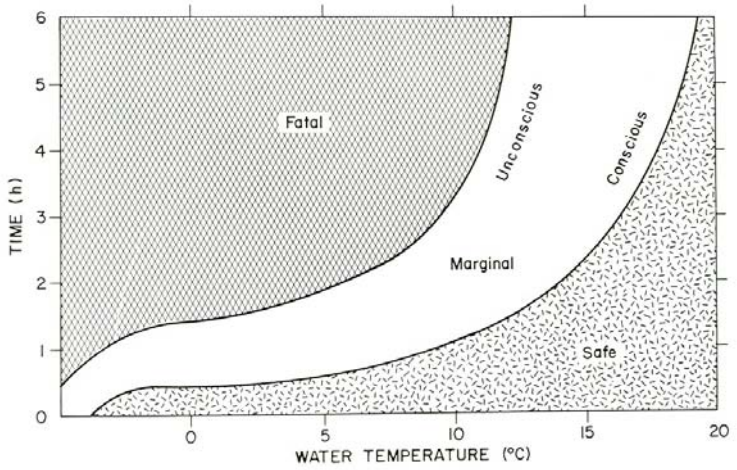

Figure 2. Relationship between water temperature and survival time in cold water (adapted from the Royal Life Saving Society of Canada, 1976).

Experts have identified four sequential stages that can occur following immersion in cold water: 1) gasping and cold shock, 2) swimming failure, 3) hypothermia and 4) post-rescue collapse (Transport Canada, 2003; Canadian Red Cross, 2006). It is believed that most deaths in cold

water occur from drowning by submersion of the airways during the first two stages of cold water immersion (Canadian Red Cross, 2006).

The rate of body cooling and the incidence of survival in cold water can vary considerably from individual to individual. This variability can be related to age, sex, body size, ratio of body mass to surface area, percentage body fat and overall physical fitness. The ratio of body mass to surface area is greater in large, heavy individuals, and their temperatures change more slowly than those of small children (Kreider, 1964). Other factors influencing cooling can include the degree of protective clothing and physical behaviour and body posture in the water. The use of drugs or alcohol can also exacerbate the effects of cold water immersion, by impairing alertness and motor skill use and by interfering with the body's temperature regulation mechanisms (Canadian Red Cross, 2006).

Immersion in cold water may occur through intentional or unintentional activities. Persons using recreational waters should remain aware of the risks involved and take appropriate precautions against cold water exposure. The Canadian Red Cross (2008) and Transport Canada (2006) have produced publications that provide information on survival in cold waters. Proper protective garments such as a wetsuit or survival suit should be worn during recreational water activities where cold water exposure is anticipated. Similarly, precautions should be taken against accidental immersions, including use of safety lines and wearing of proper personal flotation devices.

Warm water exposures

By comparison, relatively little information is available regarding the physiological effects of human exposure to warm waters. Early communication on this topic suggested that, physiologically, neither adults nor children would experience thermal stress under modest metabolic heat production, as long as the water temperature was lower than the normal skin temperature of 33°C (Newburgh, 1949). Water ranging in temperature from 20°C to 30°C is considered comfortable for most swimmers (WHO, 2003a, 2006).

In Canada, under most circumstances, ambient temperatures observed during the summer months do not reach levels sufficient to elevate recreational water temperatures above normal human body temperature. Natural hot springs—thermal springs that can reach temperatures in excess of 37°C—are a notable exception. Individuals using these types of facilities should monitor their exposures carefully so as to avoid overheating. For the majority of recreational water areas, the heat effects observed during summertime water activities are largely attributed to sun exposure. Numerous health authorities have provided guidance on avoiding heat exposure during outdoor activities; thus, this information can be viewed as extending to recreational water exposures as well.

Heat stress or heat exhaustion can occur following vigorous exercise in warm environments. Signs of heat exhaustion can include excessive sweating, elevated temperature or pulse rate, headache and dizziness or weakness. The Canadian Red Cross has similarly developed publications on the prevention of heat-related illness or injury (Canadian Red Cross, 2011). Precautionary measures to minimize the effects of heat exposure during recreational water activities are similar to those for reducing sun exposure. These include wearing lightweight

clothing and broad-brimmed hats, seeking out cool or shady areas, avoiding activity during midday periods when the sun is most intense, ensuring an adequate supply of drinking water and replenishing any salt losses.

8.2 Aesthetic characteristics

Good aesthetic quality is an important consideration in ensuring the maximum use and enjoyment of recreational waters. A recreational water area should be considered aesthetically acceptable to its users. Waters used for recreation should be free from substances (either attributable to human activities or due to natural processes) that impair its aesthetic appreciation. These can include:

- substances producing objectionable colour, odour, taste or turbidity;
- floating debris, oil, scum and other matter;
- materials that will settle to form objectionable deposits; and
- substances and conditions or combinations thereof in concentrations that produce undesirable aquatic life.

The term aesthetic has been defined as "concerned with beauty or the appreciation of beauty" (Canadian Oxford Dictionary, 2004). Thus, not only should a recreational water area be free from objectionable factors, but various other aesthetic components of the aquatic ecosystem and surrounding land should be present. Recreational waters should also be considered free from substances in amounts that would interfere with the existence of life forms of aesthetic value.

This section discusses parameters that may affect the aesthetic quality of a recreational water area. For the purposes of this document, it is the effects that these factors may have on aesthetic perception that are of primary significance. However, it should be noted that with these parameters there also exists certain implications for human health and safety. For example, waters in which the visibility has become significantly impaired can present a safety risk for recreational water users.

Aesthetic objectives for turbidity, clarity and colour have been proposed; however, it is recognized that the natural levels of these parameters in Canadian waters can vary considerably. Thus, it is recommended that, when evaluating these parameters as part of an Environmental Health and Safety Survey, the associated values also be interpreted as not being significantly increased over that which would be considered natural background.

8.2.1 Turbidity

Aesthetic objective
An aesthetic objective of 50 nephelometric turbidity units (NTU) is suggested for recreational waters.

Description

Standard Methods for the Examination of Water and Wastewater (APHA et al., 2005) defines turbidity as an "expression of the optical property that causes light to be scattered and absorbed rather than transmitted with no change in direction or flux level through the sample." The current method of choice for turbidity measurements in Canada is the nephelometric method, and the unit of turbidity measured using this method is the nephelometric turbidity unit, or NTU (Health Canada, 2012c).

Turbidity in water is caused by suspended and colloidal matter, such as clay, silt, finely divided organic and inorganic matter, plankton and other microscopic organisms (APHA et al., 2005). This parameter is important for aesthetic, safety and, to a lesser degree, health reasons. High turbidity is aesthetically displeasing and can be a safety concern when it reduces visibility through the water. Because filtration equipment and modern water treatment processes are not feasible at natural swimming areas, safety concerns associated with turbid or unclear water are dependent upon the intrinsic quality of the water itself. Lifeguards and other persons near the water must be able to see and distinguish people in distress. In addition, swimmers should be able to see quite well while under water.

Health considerations associated with turbidity are related primarily to the ability of particles to adsorb microorganisms and chemical contaminants. This can have a number of important effects on water quality:

- Suspended particles can provide a measure of protection for microorganisms (bacteria, viruses, protozoa) that have been adsorbed to their surface by shielding them from the effects of environmental stresses such as UV radiation and predation by higher microorganisms.

- The presence of turbidity may interfere with the quantification of faecal indicator organisms. In the enumeration of bacteria, it is assumed that each colony represents one cell; however, a single colony could emanate from a particle containing many bacterial cells adsorbed on its surface. Fewer cells than were actually present would then be recorded. This phenomenon would also lead to an underestimation of bacterial numbers with the MPN technique (Health Canada, 2012c).

- Particulate matter may also contain chemical contaminants such as heavy metals and biocides (Health Canada, 2012c).

Surface water levels can vary from 1 to more than 1000 NTU (Health Canada, 2012c). Runoff water quality measurements indicated levels of 4.8–130 NTU during the first hour of an urban rainfall occurrence (U.S. EPA, 1978). In the quiescent zone of a swimming beach or other recreational water area, it is suggested that turbidity measurements in the vicinity of 50 NTU would be sufficient to satisfy most recreational uses, including swimming.

8.2.2 Clarity (light penetration)

Aesthetic bbjective
Water should be sufficiently clear that a Secchi disc is visible at a minimum depth of 1.2 m.

Description
The Canadian Oxford Dictionary (2004) defines "clarity" as "the state or quality of being clear." Clarity is associated with the distance light can penetrate into a body of water and is often interpreted as "how far can one see into the water?" The clarity of a water body can be simply evaluated by using a Secchi disc, which is a device used to approximately measure visibility depths in water. The upper surface of a circular metal plate, 20 cm in diameter, is divided into four quadrants and painted so that the two quadrants directly opposite each other are black and the intervening ones are white. When suspended to various depths of water by means of a graduated line, its point of disappearance indicates the limit of visibility. It is then raised until it reappears, and the average of the two depths is taken as the Secchi disc transparency.

The principal factors affecting the depth of light penetration in natural waters include suspended microscopic and macroscopic organisms, suspended mineral particles, stains that impart a colour, detergent foams, dense mats of floating and suspended debris, or a combination of these factors.

It is important that water at swimming areas be clear enough for users to estimate depth, to see subsurface objects easily and to detect the submerged bodies of swimmers or divers who may be in distress. Aside from the safety factor, clear water fosters enjoyment of the aquatic environment.

For primary contact recreation waters, it has been suggested that clarity be such that a Secchi disc is visible at a minimum depth of 1.2 m (Environment Canada, 1972). In "learn to swim" areas, the clarity should be such that a Secchi disc on the bottom is visible. In diving areas, the clarity shall equal the minimum required safety standards, depending on the height of the diving platform or board (National Technical Advisory Committee, 1968).

8.2.3 Colour

Aesthetic objective
No numerical value can be established for colour in recreational waters. Colour should not be so intense as to impede visibility in areas used for swimming.

Description
The observed colour of water is the result of light backscattered upward from a water body after it has passed through the water to various depths and undergone selective absorption (CCME, 1999). There are two measures of colour in water: true and apparent. The term colour is most often used to mean true colour, the colour of water from which turbidity has been removed. To measure true colour, the water has to be filtered or centrifuged to remove the sources of apparent colour. The standard method for measuring colour is the platinum-cobalt method (APHA et al., 2005). Colour is measured by visual comparison with coloured solutions of known concentrations. Under this method, one Pt-Co unit is equivalent to that produced by 1 mg

platinum/L in the form of the chloroplatinate ion. The ratio of cobalt to platinum given matches the true colour of water.

The correct units for true colour are colour units (CU), with one colour unit being equivalent to 1 Pt-Co unit (APHA et al., 2005). True colour can range from less than 5 CU (colour units) in very clear waters to 1200 CU in dark peaty waters (Kullberg, 1992). Natural minerals give true colour to water; for example, calcium carbonate in limestone regions gives a greenish colour, whereas ferric hydroxide gives a red colour. Organic substances, tannin, lignin and humic acids from decaying vegetation also give true colour to water (Department of National Health and Welfare, 1979).

Apparent colour includes not only colour due to substances in solution but also that due to suspended matter (APHA et al., 2005). Measurements for apparent colour are determined on the original sample without filtration or centrifugation (APHA et al., 2005). Apparent colour is usually the result of the presence of coloured particulates, the interplay of light on suspended particles and such factors as bottom or sky reflection. An abundance of cyanobacteria can impart a dark greenish hue to water, whereas diatoms or dinoflagellates may give a yellowish or yellow-brown colour. There are algae that impart a red colour, and, rarely, zooplankton, particularly microcrustaceans, may tint the water red. Polluted waters may have strong apparent colour. Industrial discharges (particularly those from the pulp and paper and textile industries) can be highly coloured and thus may affect water coloration. Factors increasing the turbidity of natural waters may similarly affect apparent colour.

Colour in lakes may not be uniform from surface to bottom; also, the colour may change periodically. Increases in surface runoff contribute great quantities of inorganic and organic substances. Summer or early autumn production of phytoplankton blooms causes lakes to become a “soupy green,” which disappears later in the season. Exposure to light causes bleaching of certain colours in natural waters, and this effect will vary according to transparency. Colour may also be dependent on factors that affect the solubility and stability of the dissolved and particulate fractions of water, such as temperature and pH.

Generally, a rich, highly productive lake may appear yellow, grey-blue or brown as a result of quantities of organic matter, and less productive lakes tend towards blue or green caused by differential light absorption and scattering of different wavelengths (Ruttner, 1963; Reid and Wood, 1976).

The causes of colour in marine waters are not thoroughly understood, but dissolved substances are one of the contributory factors. The blue of the sea is a result of the scattering of light by water molecules, as in inland waters. Suspended detritus and living organisms give colours ranging from brown through red and green. Estuarine waters are not as brilliantly coloured as the open sea; the darker colours result from the high turbidity usually found in such situations (Reid and Wood, 1976).

The main effects of water colour on recreational activities are aesthetic and safety related. Very dark water restricts visibility both for swimmers and for people concerned with their safety. In recreational waters, it is desirable that the natural colour of the water not be altered by any

human activities. A maximum of 100 CU was proposed by Environment Canada (1972), while a guideline of not more than 30 CU above natural value is used by Alberta Environment (1999). Supporting evidence for these values has not been given. Thus, in the absence of strong supporting evidence, it is recommended that they be considered as guidance values.

8.2.4 Oil and grease

Aesthetic objective
No numerical value can be established for oil and grease in recreational waters. Oil, grease or petrochemicals should not be present in concentrations that:

- can be detected as a visible film, sheen or discoloration on the surface;
- can be detected by odour; or
- can form deposits on shorelines and bottom sediments that are detectable by sight or odour (International Joint Commission, 1987).

Description
Standard Methods for the Examination of Water and Wastewater (APHA *et a*l., 2005) defines "oil and grease" as "any material recovered as a substance soluble in the solvent." The category of oil and grease includes many different substances of mineral, animal, vegetable or synthetic origin—all of which can have vastly different physical, chemical and toxicological properties. Consequently, it is very difficult to establish a numerical criterion for oil and grease.

Contamination of recreational waters with oily substances may have natural origins or may be a result of human activities. Some oils are of natural origin, such as seepage from natural underwater oil deposits or from the decomposition of some materials. For example, natural biological populations release lipid compounds, which can form natural slicks.

Human-made contamination is of greatest concern. It can come from a number of sources, such as the discharge of industrial wastes, road runoff, residual hydrocarbon deposits from motorboat engine exhaust emissions, the discharge of fuel tank contents of ships, either accidentally or deliberately, and shipwrecks. Marinas and boat launches can also be important sources of oil and grease contamination for recreational waters.

Very small quantities of oily substances make water aesthetically unattractive. Oils can form films, and some volatile components may create odours or impart a taste to water (WHO, 2003a). Oil and grease may foul equipment, shorelines or the bodies of swimmers. The possibility exists that recreational users might still use the water in cases of low contamination. The risk of toxicity from exposure to oily substances through ingestion, skin absorption or inhalation of vapours during recreational water activity is regarded to be low. Oils and greases of animal or vegetable origin are generally considered to be non-toxic to humans. Similarly, it is recognized that petroleum compounds become organoleptically objectionable at concentrations far below the levels required for chronic human toxicity. Thus, the consumption of oil-polluted water is unlikely to be a significant source of exposure for humans (Train, 1979).

8.2.5 Litter

Aesthetic objective
No numerical value can be established for litter at recreational water areas. Recreational water areas should be free from floating debris as well as materials that will settle to form objectionable deposits.

Description
There are a variety of types of litter that can be found in recreational waters or deposited on beaches. Some examples include discarded food and food packaging, cardboard and paper products, plastic containers, styrofoam materials, rubber goods, aluminium cans, broken glass, discarded clothing, cigarette butts, medical wastes and dead animals. Large accumulations of seaweed and algae are not technically litter, but they are likely to pose an aesthetic problem (both visually and due to potential odour).

In addition to being aesthetically undesirable, the presence of litter can also present a health and safety risk to recreational water users. Some materials can be injurious to recreational water users who come in direct contact with them. Discarded litter also has the potential to attract wildlife, which can contribute to the faecal contamination of recreational waters. Indeed, litter counts have been considered as a possible indicator of the potential of acquiring gastrointestinal illness though recreational water activities. Similarly, flying and/or biting insects may also be associated with litter. These are considered at the very least a nuisance and could potentially pose a health threat in the form of zoonotic disease (NHMRC, 2008).

8.3 Chemical characteristics

Guideline values
There is insufficient information to support the establishment of guideline values for specific chemical parameters in recreational waters. Risks associated with specific chemical water quality hazards will be dependent on the particular circumstances of the area in question and should be assessed on a case-by-case basis.

In general, potential risks from exposure to chemical parameters will be much smaller than the risks from the microbiological hazards potentially present in recreational waters (WHO, 2003a). With chemical concentrations typically found in water, most recreational water users will not be exposed to sufficient concentrations necessary to elicit either an acute or chronic illness response.

Description
Chemical contaminants can enter recreational waters or be deposited on beaches from both natural and anthropogenic sources (WHO, 2003a). These include point sources, such as industrial outfalls or natural springs, and non-point (diffuse) sources, such as runoff from urban or agricultural areas.

Inorganic chemicals

National surveys of the water quality of lakes and rivers used for recreational activities indicate that concentrations of inorganic chemicals are low (NAQUADAT, 1988; Government of Canada, 1991). Analyses for heavy metals indicated that they are present in concentrations considerably below those recommended as guidelines for drinking water quality (Government of Canada, 1991). Ingestion would be considered the primary pathway of exposure for inorganic chemical contaminants; however, skin absorption is recognized as a route of uptake for certain heavy metal forms. Owing to the low concentrations encountered in most natural waters and the types and degrees of exposure involved during typical recreational water activities, exposure to inorganic chemical contaminants is not considered to represent a significant health risk for recreational water users at recognized swimming areas.

Organic chemicals

There are many sources of contamination by organic chemicals, including industrial manufacture and use and domestic use of such items as paints, fuels, dyes, glues, pesticides and cleaning supplies (NAQUADAT, 1988; Health Canada, 1997).

National surveys have analysed the level of contamination of recreational waters by organic chemicals. The concentrations of organic chemicals that have been detected in waters that could be used for recreational purposes were lower than the recommended drinking water guidelines (Government of Canada, 1991; Marvin et al., 2004) and thus should not pose a significant threat to human health.

It has been suggested that for some organic chemicals, skin absorption can be as important as ingestion in contributing to exposure (Brown et al., 1984; Moody and Chu, 1995). However, it is generally concluded that given the low concentration of organic contaminants encountered in most natural waters and the typical exposure scenarios encountered during recreational water activities, it is not likely that dermal exposure presents a significant risk (Moody and Chu, 1995; Hussain et al., 1998). Nonetheless, precautionary measures such as restricting swimming to public beaches and showering with soap and water following recreational activity will further ensure that any risk is minimized.

Managing health risks

The risk of human exposure to chemical contaminants in Canadian waters through recreational activities is considered low. Nevertheless, scenarios do exist that may contribute to the presence of a chemical water quality hazard at a particular recreational water body. As a result, it is important for beach operators and service providers to have a mechanism in place to ensure that potential chemical hazards and their risks are recognized. An Environmental Health and Safety Survey is an important tool for helping recreational water area operators identify and assess potential sources of chemical contamination that are relevant to their beach area.

Risks associated with specific chemical water quality hazards will be dependent on the particular circumstances of the area under consideration. Thus, in all instances, the risk of human exposure to chemical contaminants in recreational waters must be assessed on a case-by-case basis, taking local factors into account. In general, some key elements that should be included in any approach to assessing chemical water quality hazards in recreational waters are as follows:

- historical understanding of the area to identify past activities that may result in contaminated water and/or sediments;
- inspection of the recreational water area to identify any obvious sources of chemical contamination, such as outfalls or discharges;
- conducting of additional steps as necessary to support a quantitative health risk assessment, including chemical analysis of representative samples (using methods deemed acceptable by the regulating agencies) and review of the available toxicological information on the chemical contaminant(s) in question;
- consideration of the pattern and type of recreational activity to determine whether significant pathways of human exposure exist (e.g., through ingestion, inhalation or skin absorption); and
- consideration of the effects of the water body dimensions (area, depth) and other hydrodynamic and meteorological characteristics (tides, currents, prevailing winds) on the impact of the chemical water quality hazard in question.

A multi-barrier approach represents the most effective way of protecting recreational water users from the risk of exposure to chemical contamination at recreational water areas. This approach uses an Environmental Health and Safety Survey to highlight potential chemical water quality hazards as well as to identify barriers that may be implemented to both reduce the risk of chemical contamination and restrict swimmer exposures during periods or in areas perceived to be of increased risk.

9.0 Faecal contamination and beach sand

This section provides information on the issue of faecal contamination and faecal indicator bacteria in beach sand, including the effects on recreational waters and steps that can be taken to reduce swimmer exposures to contamination in the sand environment.

Currently, there is no conclusive evidence of a relationship between contact with beach sand and illness among recreational water users, and no guideline values can be established for concentrations of the recommended indicators for recreational waters for faecal contamination in beach sand. Routine monitoring of sand samples for the presence of faecal indicators would not be considered practical and is thus not recommended. Certain circumstances may warrant testing of sand and sediment samples, such as during investigations of potential waterborne disease outbreaks or when conducting an Environmental Health and Safety Survey.

Further research is needed to more fully characterize the relationships between faecal indicator bacteria and the possible presence of faecal pathogens in beach sand, as well as the potential implications to human health. Combining actions, procedures and tools to collectively reduce the risk of swimmer exposure to faecal contamination in beach sand and recreational waters represents the most effective approach to protecting the health of recreational water users.

Description

Microorganisms are a recognized natural component of beach sand, with numerous species of bacteria, viruses, parasites and fungi all having been isolated from the sand environment (WHO, 2003a). The results of recently published studies demonstrating that faecal indicator bacteria can be isolated in high numbers from foreshore and nearshore beach sand and sediments (Alm et al., 2003; Whitman and Nevers, 2003; Edge and Hill, 2007) have prompted renewed interest in the question whether sand can serve as a vehicle for the transmission of pathogenic microorganisms to beach users. Understandably, this issue is one of increasing concern to beach operators, public health officials and the beach-going public. Beach users often spend more time on the beach than in the water, and children routinely play in the sand at the water's edge.

Occurrence of faecal indicator bacteria in beach sand

Enteric bacteria are not expected to survive well once displaced from their primary habitat (the intestinal tract of humans and animals) to a secondary habitat such as the aquatic environment (Winfield and Groisman, 2003; Anderson et al., 2005). Their survival in this environment is influenced by a complex array of biological and environmental factors. Selective pressures that can have a negative effect on the survival of microorganisms in recreational waters include sunlight; osmotic stress; large variations in temperature, pH and salinity; low nutrient availability; and competition and predation by other microorganisms (Winfield and Groisman, 2003).

Beach sand and similar environments (foreshore and nearshore soils and sediments) can provide more favourable conditions for survival of microorganisms than the adjacent waters. Whitman and Nevers (2003) suggested that beach sand provides a suboptimal, yet viable, environment for the survival of enteric bacteria by providing protection from sunlight, buffered temperatures, a degree of cover from predation, large surface areas for biofilm development and a replenishing supply of moisture and organic nutrients from wave swash.

Numerous authors have observed the presence of faecal indicator bacteria in beach sands and sediments, often at concentrations several orders of magnitude higher than those of the adjacent swimming waters. At a Lake Michigan beach, Whitman and Nevers (2003) detected *E. coli* counts in foreshore and submerged sand (1000–10 000 mean cfu/100 mL) that exceeded those of the swimming water (100–1000 mean cfu/100 mL). Similar findings were reported by Williamson et al. (2004) in a survey of *E. coli* densities in swimming water and sand pore water at Lake Winnipeg beaches. Alm et al. (2003) reported that, on average, concentrations of *E. coli* and enterococci were 3–17 times and 4–38 times greater, respectively, than in the water column at beaches located on Lake Huron and the St. Clair River. Edge and Hill (2007) found *E. coli* concentrations as high as 114 000 cfu/g dry sand at a Lake Ontario beach.

Similar studies have shown that faecal indicator bacteria can be isolated from other habitats within a beach watershed, such as backshore sand (Byappanahalli et al., 2006), subtropical and temperate stream sediments (Byappanahalli et al., 2003; Jamieson et al., 2003, 2004; Ferguson et al., 2005; Ishii et al., 2006a), temperate forest soils (Byappanahalli et al., 2006) and within mats of the green algal species *Cladophora* (Whitman et al., 2003; Ishii et al., 2006b).

A concept that is receiving increasing attention within the scientific community is that enteric bacteria may be capable of multiplying in the sand and similar environments in tropical and subtropical climates provided that the proper conditions for growth are met (Davies et al., 1995; Byappanhalli and Fujioka, 1998; Solo-Gabrielle et al., 2000; Desmarais et al., 2002; Anderson et al., 2005). These would include (among other factors) suitable conditions of temperature, moisture and available nutrients, as well as reduced competition from other microflora. Warmer temperatures and higher soil nutrient concentrations have been cited as factors that may favour the multiplication of indicator bacteria in tropical soils (Hardina and Fujioka, 1991; Whitman and Nevers, 2003). Whitman and Nevers (2003) proposed that certain conditions that exist at tropical beaches may be encountered at temperate beaches in the United States during the summer months.

During an investigation of sand and water quality at a Lake Michigan beach, Whitman and Nevers (2003) observed that in freshly replaced beach sand *E. coli* was able to recolonize to earlier levels within a period of 2 weeks. The authors were not able to confirm whether the effect was due to multiplication in an unexploited environment or the result of deposition by external sources. Kinzelman et al. (2004a) detected similar levels of diversity among *E. coli* strains collected from water, foreshore sands and submerged sands at a Racine, Wisconsin, beach. It was suggested that accumulation, and not replication, was the major contributor to *E. coli* observed in the sand.

A few researchers have suggested that some portion of the sand- or soilborne strains of *E. coli* may represent a genetically distinct group, separate from the majority of isolates that dominate in the host sources (Winfield and Groisman, 2003; Byappanahalli et al., 2006; Edge and Hill, 2007). Moreover, it is proposed that these strains may have adapted for prolonged survival and possibly growth in the soil environment (Winfield and Groisman, 2003; Byappanahalli et al., 2006). Byappanahalli et al. (2006) reported finding evidence of a genetically diverse *E. coli* population among forest soils located within a Lake Michigan watershed. Ishii et al. (2006a) reported finding evidence of "naturalized" *E. coli* populations in northern temperate soils located within Lake Superior watersheds. The authors further reported that these strains were capable of multiplying in non-amended, non-sterile soils at temperatures at or above 30°C.

Effects of beach sand on microbiological water quality

There has been a wealth of information in support of the notion that beach sand can present a significant non-point source of faecal contamination for swimming waters (Alm et al., 2003; Whitman and Nevers, 2003; Williamson et al., 2004). Faecal indicator bacteria in sand can originate from a variety of faecal pollution sources. Gulls in particular are thought to be a significant source of faecal contamination for beaches (Levesque et al., 1993; Fogarty et al., 2003; Williamson et al., 2004). Canada geese populations can also present a source of faecal contamination at areas adjacent to surface waters (Alderisio and DeLuca, 1999).

Mechanisms of transfer of sandborne contamination to the water environment include wave swash, rain-mediated runoff and direct transfer from swimmers. Resuspension of nearshore sediments can occur through a number of mechanisms, including wave action (including those artificially generated by commercial and recreational boating), storms and swimmer activities. Boehm et al. (2004) proposed that the wave-driven and tidally driven recirculation of water

through the beach aquifier may also present a mechanism for transfer of microorganisms and nutrients from the sand environment to swimming waters.

Pathogenic microorganisms in beach sand

Few studies have been published regarding the occurrence and survival of enteric pathogens in beach sand and sediments. Bolton et al. (1999) reported that culturable *Campylobacter* and *Salmonella* species were detected in 45% and 6% of sand samples, respectively, in beach sand samples collected from various UK coastal beaches. Obiri-Danso and Jones (1999) reported that *Campylobacter* could be detected year-round in low numbers (< 0.5 log cfu/g dry weight) in river sediments at two freshwater swimming sites in the northwest of England. *Salmonella* was not detected at either site. In a follow-up study at coastal beaches within the same watershed, Obiri-Danso and Jones (2000) detected *Campylobacter* (*C. lari*, urease-positive thermophilic campylobacters) in sediment samples collected during the winter months only. *C. jejuni* and *C. coli* could not be isolated, and *Salmonella* was not detected at any point during the analysis. The authors concluded that the sediments were not acting as a reservoir for these pathogens in this system. NWRI (2006) found *Campylobacter* species commonly occurring in sand pore water at two bird-contaminated beaches in Hamilton Harbour, Lake Ontario. *C. jejuni* were more common than *C. coli* or *C. lari*, although all species appeared to be in low numbers in the pore water.

Others have reported the detection of *S. aureus* and *P. aeruginosa* in beach sand (Papadakis et al., 1997; Esiobu et al., 2004). *P. aeruginosa* is a relatively hardy species of bacteria that is widely distributed in the aquatic environment and is known to cause skin rashes and eye and ear infections in swimmers. Humans are the primary source of *S. aureus* in recreational waters; thus, its presence in beach sand is thought to be directly related to swimmer activities. The organism is known to be associated with skin infections in swimmers (rashes, infected cuts and scratches). Very few data are available regarding the presence of other potential recreational waterborne pathogens (viruses, protozoan parasites) in beach sand (WHO, 2003a).

Related epidemiology

Despite the findings that faecal indicator bacteria and potentially pathogenic microorganisms can be detected in beach sand, there has been little evidence published to indicate a link to illness among beach users. Marino et al. (1995) reported that there was no evidence of a relationship between the incidence of skin symptoms and sand concentrations of any of the indicator organisms monitored (*E. coli*, faecal streptococci, *Candida albicans*, dermatophytic fungi) during a prospective epidemiological study at two beaches in Malaga, Spain. Heaney et al. (2009) studied associations between sand exposure and illness at marine and freshwater beaches under the U.S. EPA's NEEAR study. The rates of illness varied from beach to beach, however, overall the authors observed that digging in sand was associated with a modest increase in reported gastrointestinal illness (adjusted incidence ratio 1.13; 95% confidence interval [CI] 1.02-125). Individuals who reported being buried in sand showed a slightly stronger incidence of gastrointestinal illness (1.23; CI,1.05-1.43) and diarrhea (1.24; CI,1.01-1.52). No associations were demonstrated for sand contact and non-enteric illness (Heaney et al., 2009). Epidemiological studies in which microbiological sampling has been performed at very shallow depths have similarly failed to show a correlation between water quality at this depth and swimmer illness (Calderon et al., 1991; Fleisher et al., 1996; McBride et al., 1998; Haile et al.,

1999). Microbiological counts in shallower waters can be expected to exceed those at greater depths, owing to disturbances to foreshore and nearshore sands and sediments.

Managing health risks

Management actions to reduce the extent of faecal contamination affecting the beach area, as well as steps to restrict swimmer exposure to recreational waters during periods of or in areas perceived to be at increased risk are part of an effective strategy to protect against the risk of human exposure to pathogens arising from faecal contamination in the foreshore and nearshore sand environment.

The Environmental Health and Safety Survey is an important tool for helping recreational water operators identify potential onshore sources of faecal contamination that are relevant to their beach area. Further information on the EHSS process can be found in Part I (Management of Recreational Waters).

For beach managers and operators, barriers to reduce the extent of faecal contamination can include the physical removal of litter that may attract animals to the area and the installation of physical barriers designed to discourage wildlife. Examples of such barriers can include animal-proof refuse containers, fences and gull nets. Jurisdictional regulations restricting access for pets on public beaches present another potential control mechanism.

Physical manipulations of the sand environment have also been proposed as a potential action to help minimize faecal contamination and reduce its transport to swimming waters. Kinzelman et al. (2004b) reported that deep mechanical grooming without levelling was effective in reducing sand levels of *E. coli* at a Racine, Wisconsin, beach, particularly in wet sand. Targeted beach grading to increase the steepness of the slope of a beach has been suggested as another action that can improve water quality (City of Racine Health Department, 2006). A more steeply sloped beach reduces the area vulnerable to wave swash and permits more rapid sand drying through improved drainage (Clean Beaches Council, 2005; City of Racine Health Department, 2006).

Pre-emptive beach postings or swimming advisories restricting recreational water activities for short periods immediately after rainfall events present another potential barrier. These act by limiting swimmer exposure to faecal contamination that may have been washed from the sand environment to the swimming area.

Beach users can also do their part to contribute to these strategies by ensuring that their litter is properly disposed of, refraining from feeding animals on or near the beach, and complying with any existing beach regulations or codes of conduct. They may also contribute by becoming informed of steps that can be taken to reduce their personal exposure. Beach users are reminded to adopt proper hygiene practices such as avoiding mouth contact with items that have been in contact with sand, washing their hands prior to eating and showering as soon as is practical after visits to the beach. The use of clean beach towels can also help reduce the degree of sand contact (WHO, 2003a).

Larger-scale management options for beaches will require a comprehensive review of the contamination inputs and watershed characteristics and the identification of specific options to

minimize or control the sources of faecal contamination and to reduce the transfer of pollution to the swimming area.

Summary

1. Beach sand and related environments may provide a more favourable environment for microorganisms of faecal origin, which may permit them to survive for longer periods than in the adjacent water. Physical factors such as wave action, storm surges, tidal activity and high swimmer load can result in the transference of microorganisms from foreshore and nearshore sand and sediments to swimming waters.

2. Currently, there is no conclusive evidence to indicate a link between microorganisms in beach sand and illness among beach users. Further research is needed to determine the relationships between faecal indicator bacteria and the possible presence of faecal pathogens in beach sand, as well as the potential implications to human health.

3. Barriers that collectively reduce risk of exposure for beach users could include public education campaigns, improved beach sanitation practices, appropriate sand grooming practices and actions designed to discourage the activities of animals (birds and other wildlife) within the beach area.

10.0 Faecal pollution source tracking

Description
A wide variety of faecal sources are capable of contaminating recreational waters. Faecal pollution can come from sewage treatment plant effluents, stormwater and combined sewer overflows, faulty or improperly designed septic tanks, improper farming practices, intensive livestock and poultry operations, local wildlife (e.g., geese and gulls) and even recreational water users themselves. Understanding the specific sources of faecal contamination can help to assess public health risks and target appropriate risk management barriers. These, in turn, can reduce beach postings and prevent potential waterborne disease outbreaks.

Faecal source tracking can be performed by a variety of chemical and microbiological methods.

<u>Chemical methods</u>
Numerous chemical compounds have been investigated as potential markers for human sources of faecal pollution. Chemical methods of analysis are based on the detection of chemical compounds known to be present in faecal material as a result of human activities—either through consumption and/or metabolism and the subsequent excretion in faeces, or via disposal as sewage wastes. Caffeine, detergents, laundry brighteners, fragrance materials, faecal sterols and faecal stanols have been proposed as markers of faecal pollution from sewage treatment plants (Glassmeyer et al., 2005). Chemical tracers such as dyes have also been used to confirm suspected point sources of contamination, such as wastewater outfalls. A noted advantage of using chemical markers is that the time required for analysis is shorter than that for many microbial methods. There are still a number of information gaps regarding the use of chemical

markers in terms of presence and fate in the environment and the sensitivity of the detection methods. More research is currently needed to clarify some of the uncertainties surrounding their use as potential faecal source indicators.

Microbial methods

Microbial source tracking (MST) is an emerging field. Methods are based on comparing the similarity of microorganisms in water samples with those from known faecal sources to make inferences about the source of faecal pollution. In recent years, a growing number of microbiological methods have been developed (Scott et al., 2002; Simpson et al., 2002; Meays et al., 2004; Edge and Schaefer, 2006). Consequently, more attention has been focused on microbial methods than on chemical methods.

Microbial methods can be divided into library-dependent and library-independent methods. To date, library-dependent methods have been more widely used in microbial source studies, although library-independent methods are increasingly under investigation. Library-dependent methods are based upon choosing a faecal indicator microorganism (e.g., *E. coli* or enterococci) and establishing a reference library of characteristics of individual isolates obtained from known faecal pollution sources. For example, a library could be a database of antibiotic resistance profiles or DNA fingerprints of *E. coli* isolates obtained from animal faeces and municipal wastewater effluent (Wiggins, 1996; Dombek et al., 2000; Carson et al., 2001). The similarity of the profiles or fingerprints of *E. coli* isolates obtained from recreational waters ("unknowns") can then be compared with the profiles or fingerprints in the library ("knowns") to make statistical inferences about the source of the waterborne *E. coli* isolates.

Library-independent methods are based upon detecting host-specific markers to indicate the presence of faecal contamination from a specific human or animal host in the water. Most library-independent methods rely on PCR to detect the host-specific markers in water samples. Host-specific markers may be such things as toxin genes (Khatib et al., 2002, 2003), genes for virulence factors (Scott et al., 2005) or highly conserved DNA sequences (Bernhard and Field, 2000a). Some of the most promising results to date for developing host-specific markers for faecal pollution source tracking involve 16S rDNA markers within the genus *Bacteroides*. These anaerobic bacteria comprise a very large portion of the faecal flora in warm-blooded animals. Bernhard and Field (2000b) developed *Bacteroides* 16S rDNA PCR assays specific to ruminants and humans, and such assays have been applied successfully to MST studies in recreational waters (Boehm et al., 2003; Bower et al., 2005; Noble et al., 2006).

An important aspect of MST has been discriminating between human and animal faecal contamination. This is because faecal pollution from a human source (e.g., sewage) may present human health risks different from those associated with faecal pollution from animal sources. Viruses that infect humans are more likely to occur in human faecal wastes. However, wildlife species can carry pathogens such as *Campylobacter*, *Cryptosporidium* and *Giardia*, which can pose a risk to human health.

Studies in the late 1990s had raised considerable expectations for microbial source tools to resolve problems in faecal pollution source identification. Recent publications (Griffith et al., 2003; Stoeckel et al., 2004) have pointed out some of the limitations associated with the use of

microbial methods. Library-based methods require very large reference libraries and can have high misclassification rates. Non-library-based methods currently suffer from gaps in the information on the host specificity of the markers. Additional research is required to understand more fully the advantages and limitations of MST methods.

Current state of the science

Current thinking is that while many methods exist in the current MST toolbox, there is no universally accepted best method. While some methods have achieved a level of maturity where they could be considered for standardization, others are still experimental or research-grade tools (Edge and Schaefer, 2006). It is further felt that with any MST study, it is advisable to have multiple lines of evidence before making inferences about sources of faecal contamination.

Some recent MST studies have been perceived to have placed a "wet blanket" on the field by illustrating some of the significant challenges associated with these types of undertakings. These include the high costs of analysis and the difficulties encountered when using these technologies in watersheds that have numerous faecal sources. Nevertheless, there have been successful applications of MST in field studies. MST methods have been used to identify unexpected faecal pollution sources, to verify information from other lines of evidence, to resolve local beach closure problems involving limited contamination sources and to break down large source tracking problems into more manageable studies. Edge and Hill (2007) demonstrated the application of two library dependent MST methods for determining that bird droppings, rather than municipal wastewater, were the primary source of faecal pollution responsible for beach postings in Hamilton Harbour.

The field of MST is still evolving, and the MST toolbox is getting better. Novel molecular tools such as DNA microarrays (Hamelin et al., 2006; Soule et al., 2006) and protozoan genotyping methods (Jiang et al., 2005; Ruecker et al. 2007) could lead to the identification of new host-specific DNA markers. Other tools based on DNA markers for host cells in faeces may also prove useful for faecal pollution source tracking in the future (Martellini et al., 2005).

Application of microbial source tracking

MST studies can be expensive and time-consuming. As well, the current state of the science is such that the methods may not be able to completely resolve all of the sources contributing to the faecal contamination of the watershed and the recreational water area.

A good understanding and formulation of the nature of the faecal contamination problem are required before considering the need for an MST study. The Environmental Health and Safety Survey is a useful tool and an important first step for helping recreational water area operators, service providers and local authorities identify the sources of contamination that are relevant to their recreational water area.

At present, it is not possible to recommend a standard faecal source identification approach and method that would be applicable to any situation in recreational waters. The decision of which MST method to use will be influenced by factors such as the potential complexity and number of faecal pollution sources, the geographic and temporal considerations for the study area, and the available funds, equipment and expertise to conduct the study. For individuals wishing to pursue

this approach, some guidance for the selection of an appropriate MST approach can be found in publications from the U.S. EPA (2005b) and the U.S. Geological Survey (Stoeckel, 2005).

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Appendix A: List of acronyms and abbreviations

AK	amoebic keratitis
ANC	acid-neutralizing capacity
bw	body weight
CDC	Centers for Disease Control and Prevention (U.S.)
cfu	colony-forming unit
DAEC	diffuse adherent *E. coli*
DNA	deoxyribonucleic acid
EAEC	enteroaggregative *E. coli*
EHEC	enterohaemorrhagic *E. coli*
EHSS	Environmental Health and Safety Survey
EIEC	enteroinvasive *E. coli*
EMPACT	Environmental Monitoring for Public Access and Community Tracking
EPA	Environmental Protection Agency (U.S.)
EPEC	enteropathogenic *E. coli*
eq	equivalents
ETEC	enterotoxigenic *E. coli*
EU	European Union
GAE	granulomatous amoebic encephalitis
GIS	geographic information system
HAV	hepatitis A virus
HCGI	highly credible gastrointestinal illness
HUS	haemolytic uraemic syndrome
LD_{50}	median lethal dose
MAC	maximum acceptable concentration
MPN	most probable number
MST	microbial source tracking
NEEAR	National Epidemiologic and Environmental Assessment of Recreational
NOAEL	no-observed-adverse-effect level
NTU	nephelometric turbidity unit
PAM	primary amoebic meningitis
PCR	polymerase chain reaction
pfu	plaque-forming unit
QPCR	quantitative polymerase chain reaction
rDNA	ribosomal deoxyribonucleic acid
RNA	ribonucleic acid
S.A.F.E.	Swimming Advisory Forecast Estimate
SRSV	small, round, structured viruses
STEC	Shiga toxin-producing *E. coli*
TDI	tolerable daily intake
UK	United Kingdom
USGS	United States Geological Survey
UV	ultraviolet
VTEC	verocytotoxigenic *E. coli*
WHO	World Health Organization

Appendix B: Microbiological sampling and analysis

B.1 Sampling procedures for water

Samples for microbiological examination should be collected in clean, sterilized, environmentally sensitive bottles with screw cap closures. Borosilicate glass or autoclavable plastic bottles capable of withstanding repeated sterilization at 121°C or 170°C are recommended. Bottles capable of holding volumes of 200–500 mL should be adequate for most analyses; however, certain circumstances may require the collection of greater volumes of water (e.g., 1 L, 10 L).

When sampling by hand, the sterilized bottle should be opened with the base firmly held in one hand and with the opening facing downward. Exceptional care should be taken at all times during sampling to avoid accidental contact with the mouth of the bottle or the bottle cap. The bottle mouth is plunged downward into the water 15–30 cm below the surface (for both deep and shallow waters). The bottle is then turned towards the current (if there is one), tilted slightly upwards to displace the air and then gently pushed forward away from the hand, body, boat or other sampling platform. Samples collected from a boat or other platforms should be collected from the upstream side of these objects.

When collecting is done with a sampling pole, the bottle should be fit into the holder in the recommended manner. With the cover removed, the sample should be collected upstream and away from the collector by simulating the scooping method used during the collection of a hand-collected sample.

The volume collected should be sufficient to enable all of the required testing to be performed. Before recapping the bottle, a small amount of sample should be discarded, leaving an airspace to allow for proper mixing prior to analysis. Once capped, the bottle should be properly labelled and placed in an insulated cooler that contains pre-frozen cooling packs or ice. Additional data, such as the time of sample collection, temperature of the water and similar observations, should be recorded at this time as part of the sample record.

Responsible authorities may also wish to include within their monitoring program a requirement to collect supplemental data on various water quality and meteorological parameters at the areas being monitored. Several researchers have reported on the usefulness of such information in developing mathematical models for predicting recreational water quality (Nevers and Whitman, 2005; Olyphant and Whitman, 2005). Potentially useful measurements may include the following:

- amount of rainfall;
- degree of sunlight and cloud cover;
- temperature (air, water);
- tidal stage and water level;
- wave height;
- number of swimmers;
- wind direction and speed;
- bird populations (gulls, ducks, geese); and
- turbidity.

An example of a sample collection and reporting form is provided in Appendix E.

B.2 Sampling procedures for sand and sediments

When epidemiological or other evidence indicates that swimming beaches could be the source of waterborne diseases among swimmers, sand and sediment sampling and analysis for suspected pathogens may be warranted. Many investigations have demonstrated that faecal indicator bacteria and faecal pathogens can persist for extended periods in sand and sediments.

At present, there is no single preferred procedure for the collection and analysis of sand and sediment samples. A variety of methods have been proposed for the collection of sand samples, including the use of sterilized scoops, spatulas, corers, probes and other sterilized collection vessels. Authorities are advised to consult the scientific literature to determine which methods may be the most suitable for their needs. Sediment samples may be collected with sterile, 250- to 500-mL wide-mouth jars, and the same precautions that were used with water sampling should be followed to ensure aseptic collection. In shallow waters, the jars are pushed along the bottom, collecting the material at the sediment–water interface. When it has been half filled, the container is retrieved, the excess water is poured off and the container is sealed. In deeper waters, sediment samplers used for collecting benthic invertebrates, such as the Ponar, Petersen or Ekman grabs, can also be used (APHA et al., 2005). Regardless of the method used, the use of sterilized equipment and aseptic technique is of utmost importance in minimizing the risk of accidental contamination of the sample. As with water samples, once collected, sand and sediment samples should be properly labelled and placed in an insulated cooler containing pre-frozen cooler packs or ice.

B.3 Sample transport, preservation and storage

After collection, samples should be held at temperatures below 10°C and in the absence of light until the time of analysis. Insulated coolers containing pre-frozen cooling packs or ice may be used during sample transport to the laboratory. To prevent the possibility of contamination, samples should be packed in such a manner so as to prevent contact between the bottle lids and any free water (from thawing/melting of the cooling material or otherwise) that may be retained within the cooler. As well, the samples should never be frozen. Thus, they should be protected from direct contact with the ice or cooling packs as necessary to prevent freezing.

Storing samples under refrigerated temperatures will have only a limited effect in terms of preserving the distribution of microbiological populations in a recreational water sample. As a result, microbiological analysis of water samples should be initiated as soon as possible to avoid unpredictable changes in the microbial population (APHA et al., 2005). Samples should be analysed within 24 hours from the time of collection (Bartram and Rees, 2000); however, analysis within 8 hours is recognized as the preferred time interval (Bartram and Rees, 2000; APHA et al., 2005). If the time required for transport is expected to exceed 6 hours, it is recommended that field analysis be considered (APHA et al., 2005). Similarly, if the results are to be used in legal action, it is recommended that special means (e.g., rapid transport, courier service) be used to deliver the samples within the specified time limits and to maintain the chain of custody (APHA et al., 2005).

Data such as the temperature of the samples when received at the laboratory and the times of sample collection, reception and analysis should be recorded as part of the sample record. Such information may prove valuable during the interpretation of results (Bartram and Rees, 2000).

B.4 Methods for microbiological analysis

B.4.1 Recommended indicators of faecal contamination

Consideration should be given to the type of water being analysed when selecting the most appropriate method for analysis. Currently, two main types of methods are used for the routine detection and enumeration of *E. coli* and enterococci in recreational waters: the multiple-tube fermentation (MTF) method and the membrane filtration (MF) method.

Multiple-tube fermentation (MTF) procedure

In this method, serial dilutions of a 100-mL sample are prepared in replicate sets of tubes or wells containing differential media. These are then incubated and examined for the number of positive test reactions. The number of positive tubes per dilution is then matched to a most probable number (MPN) table, which provides a statistical estimate of the number of target organisms in the original 100-mL sample.

An advantage of the MTF method is that it can be useful for samples where conditions have made MF unusable—for example, with turbid, coloured or grossly contaminated water (Health Canada, 2006a). As well, the use of liquid media may more easily permit the recovery of stressed organisms. Limitations are that the method can be time-consuming, can require large amounts of media and glassware and can require a long turnaround time to obtain results, particularly if confirmation steps are required. As well, the method provides only a statistical estimate of the presence of the target organism and not a true count of the total number of bacteria present.

Membrane filtration (MF) procedure

In this procedure, the water sample (usually 100 mL) is passed through a filter that retains the bacteria. The filter is then placed on an appropriate differential or selective medium and incubated. Following incubation, the appropriate colonies are counted, and the result is recorded as the number of target organisms per 100 mL.

Advantages of the MF method are the ability to test greater volumes of water, reduced requirements for labour and materials and the greater reliability and reproducibility associated with a direct count. For these reasons, the MF procedure is largely favoured over the MTF procedure for the routine examination of recreational waters. There are some limitations associated with the MF method, however. Samples that are high in turbidity can interfere with filtration, and highly contaminated samples may overwhelm the filter, preventing its ability to produce an accurate count. As well, the direct transfer onto solid, selective media may hinder the recovery of some organisms, and thus a resuscitation step may be required.

Defined substrate technologies

With both procedures, more recent methods have been based on the ability to detect specific enzymes considered to be characteristic for the target organisms. *E. coli* methods are based on the detection of the enzyme β-glucuronidase, which is thought to be restricted to this organism and a few strains of *Salmonella* and *Shigella*. Enterococci methods are based on the detection of

the enzyme β-glucosidase, which is characteristic for this group. Specifically tailored chromogenic or fluorogenic substrates are incorporated into the growth media, which, when metabolized by the target organism, confer a unique property to the developing colony or surrounding media that can be used for diagnostic purposes. Chromogenic substrates produce a distinct colour change when hydrolysed, whereas fluorogenic substrates produce a fluorescent product that can be detected upon viewing under long-wavelength ultraviolet (UV) light.

Many currently available commercialized methods have used these principles in what has been termed "defined-substrate technology." With this technology, the indicator substrates are specifically designed to function as both the principal source of carbon and energy for the target bacteria. Other competitive bacteria cannot utilize the substrate and are therefore unable to interfere with the recovery of the target.

Escherichia coli

Standard Methods for the Examination of Water and Wastewater (APHA et al., 2005) recommends the mTEC method—an MF method developed by the U.S. Environmental Protection Agency (EPA)—for the detection and enumeration of *E. coli* at natural swimming beaches. This method involves placement of the filter on selective (mTEC) media, a resuscitation step at 35°C for 2 hours to rejuvenate stressed organisms, incubation at 45°C for 22 hours to detect all thermotolerant coliforms and then transfer to a urea substrate medium to differentiate urease-negative *E. coli* from other thermotolerant coliforms, which are mostly urease-positive.

The U.S. EPA has published a list of approved methods for enumerating *E. coli* in recreational waters (U.S. EPA, 2006c). Method 1103.1 is the original mTEC method. Method 1603 is a modified (single-step) mTEC method that uses a single medium containing a chromogenic substrate (5-bromo-6-chloro-3-indolyl-β-D-glucuronide, or BCIG). Hydrolysis of the BCIG substrate by *E. coli* imparts a red or magenta colour to the resulting colonies, differentiating them from other thermotolerant coliforms. Method 1604 is another MF method that uses MI medium, which contains both a fluorogen (4-methylumbelliferyl-β-D-galactopyranoside, or MUGal) and a chromogen (indoxyl-β-D-glucuronide, or IBDG) for the simultaneous detection of both total coliforms and *E. coli*, respectively.

Other methods using similar principles have been approved for the enumeration of *E. coli* in water supplies (ISO, 1998; APHA et al., 2005; U.S. EPA, 2006b). Similarly, various commercial methods – prepackaged miniaturized or simplified versions of traditional MF/MTF tests – have also received approval (U.S. EPA, 2006b). Laboratories in Canada may wish to evaluate the applicability of specific methods to recreational waters in their regions.

Enterococci

Standard Methods for the Examination of Water and Wastewater (APHA et al., 2005) contains two official methods for the examination of enterococci at swimming beaches—an MTF procedure and an MF procedure. The MTF procedure involves inoculation of a series of tubes of azide dextrose broth, incubation at 35°C for 48 hours and confirmation of tubes showing growth on a bile-esculin-type agar by the appearance of brownish-black colonies with brown halos. The MF procedure represents the mE method, originally described by the U.S. EPA in 1985. Filters are placed on mE agar, incubated at 41°C for 48 hours and then transferred to esculin-iron agar

(EIA) medium to confirm esculin hydrolysis. Pink to red colonies with black or reddish-brown precipitates are considered enterococci.

The U.S. EPA has similarly published a list of approved methods for the enumeration of enterococci from recreational waters (U.S. EPA, 2006c). Method 1106.1 is the original mE method. Method 1600 is a modified (single-step) method (mEI), which reduces the time of analysis from 48 hours to 24 hours. The method uses a single medium that contains a chromogenic substrate (indoxyl-β-D-glucoside). Hydrolysis of the substrate by enterococci confers a blue halo to the resulting colonies, distinguishing them from other non-enterococci.

As with *E. coli*, other methods using similar principles have been approved for the enumeration of enterococci in water supplies (ISO, 2000; APHA et al., 2005; U.S. EPA, 2006b). Various commercial methods – prepackaged miniaturized or simplified versions of traditional MF/MTF tests – have also received approval (U.S. EPA, 2006b). Laboratories in Canada may wish to evaluate the applicability of specific methods to recreational waters in their regions.

B.4.2 Pathogenic microorganisms

The routine testing of recreational waters for the presence of pathogenic microorganisms (bacteria, viruses, protozoa) is not recommended. However, certain circumstances may warrant testing for the presence of specific organisms, such as during investigations of potential waterborne disease outbreaks.

Bacterial pathogens

*Standard Methods for the Examination of Water and Wastewat*er (APHA et al., 2005) describes methods for the isolation, detection and identification of the bacterial pathogens of concern for recreational waters: *Campylobacter*, *E. coli* O157:H7, *Salmonella*, *Shigella*, *Aeromona*s, *Legionella*, *Mycobacterium*, *Pseudomonas*, *Leptospira* and *Staphylococcus aureus*.

In general, methods for the isolation and detection of bacterial pathogens in recreational waters follow the same basic design: concentration of the target organism (through MF, centrifugation or growth in enrichment media), differentiation from non-target organisms (e.g., via growth in selective media or using antibody-based methods of detection) and confirmation (through a combination of morphological assessment, response to biochemical tests and serological identification).

Methods for more rapid detection of bacterial pathogens in recreational waters using advanced biochemical, immunological or gene sequence-based technologies are currently being explored. Notably, developments in polymerase chain reaction (PCR)-based methods have generated interest in the application of this technology for such purposes. Currently, PCR methods have been described for the waterborne detection of all of the bacterial pathogens considered to be of concern. Similarly, quantitative or real-time PCR methods have been described for a number of important enteric bacterial pathogens, including *Salmonella*, *Campylobacter* and *E. coli* O157:H7. At present, more work is needed in this area to develop standardized methods that can be accurately, reliably and affordably used. Several authors have published reviews describing the current state of the knowledge with respect to emerging technologies for the enumeration and detection of recreational water pathogens (Ashbolt, et al., 2001; Noble and Weisberg, 2006;

Savichtcheva and Okabe, 2006). Laboratories in Canada are advised to consult the literature for further information as to the potential applicability of specific methods to recreational waters in their areas.

Viral pathogens

The detection of pathogenic viruses is beyond the scope of most water microbiology laboratories. When necessary, testing should be performed only by trained virologists having adequate facilities for the proper handling of these organisms.

The methodology for detection of waterborne viruses has been standardized to a certain extent; however, even the most accepted methods continue to be researched, modified and improved. *Standard Methods for the Examination of Water and Wastewater* (APHA et al., 2005) contains procedures for the concentration of viruses and provides guidance on methods for virus detection and identification. As well, the U.S. EPA has published a *Manual of Methods for Virology* (U.S. EPA, 2001a), which provides detailed, step-by-step procedures for the recovery, detection, enumeration and identification of viruses from water, sewage and other related effluents.

In general, the recovery and detection of pathogenic viruses from surface waters samples are difficult processes. Viruses are present in small numbers in faecally contaminated waters; thus, large volumes of water (up to thousands of litres) must be concentrated in order to detect their presence. Adsorption-elution methods or ultrafiltration represent the main methods used to collect and concentrate viral particles from water samples. Further concentration of the filtered sample can be accomplished through the application of flocculation or precipitation techniques.

Traditionally, cell culture methods have been the most widely used for the detection of enteric viruses in water. These methods provide valuable information on virus infectivity and concentrations, yet they can be laborious and time-consuming, and not all viruses can be grown in cell culture. More recently, conventional PCR-based methods have been applied to the detection of viruses in environmental samples. Currently, no suitable alternative to cell culture exists to enable the assessment of virus viability. Specific applications of PCR technology (reverse transcriptase PCR, quantitative PCR, integrated cell culture PCR) are being explored for their ability to overcome some of the limitations associated with conventional PCR methods.

Numerous variations of the PCR method have been described that have been successfully used for the detection of viruses in recreational waters. Although much progress has been made in this area, standardized procedures have not yet been developed. Individuals are thus advised to consult the literature for further information regarding specific methods. Review articles summarizing the existing methods as well as potentially emerging technologies have been published (Bosch, 1998; Griffin, et al., 2003; Fong and Lipp, 2005).

Protozoan pathogens

Testing for pathogenic protozoa is outside the scope of analytical services provided by most water testing laboratories. Analysis requires the use of trained analysts and highly specialized equipment.

Standard Methods for the Examination of Water and Wastewater (APHA et al., 2005) does provide an overview of the methods that can be used for the recovery and detection of *Giardia* and *Cryptosporidium* in environmental samples; however, specific procedures are not described.

The U.S. EPA has approved two methods for the detection of *Giardia* and *Cryptosporidium* in water samples: Method 1622, which is a stand-alone method for *Cryptosporidium*, and Method 1623, which can be used for the simultaneous detection of both organisms (U.S. EPA, 2006c). These methods are considered the most widely used for the detection of *Giardia* and *Cryptosporidium* in water. The detection of *Giardia* cysts and *Cryptosporidium* oocysts is a complex and difficult procedure. Even the most widely used and accepted methods suffer from limitations associated with the recovery of these organisms. Work is ongoing to continue to refine and improve the accuracy and sensitivity of these methods.

The recovery and detection of *Giardia* cysts and *Cryptosporidium* oocysts in water samples comprise three basic steps: concentration (via flocculation, centrifugation or filtration), separation from interfering debris (through the use of density-gradient centrifugation, immunomagnetic separation or fluorescence activated cell sorting) and detection (via immunofluorescent staining or PCR-based methods).

Of the methods used for detection, immunofluorescent staining currently represents the most widely used technique. This procedure involves the application of fluorescent antibodies directed against cyst and oocyst antigens, followed by identification of the labelled cysts and oocysts under an immunofluorescence microscope.

Various conventional PCR-based methods have been described for the detection of *Giardia* and *Cryptosporidium* in recreational waters. Other analytical techniques, such as restriction fragment length polymorphism (RFLP) analysis, have been described for the further characterization of *Cryptosporidium* species and genotypes. Genotype information can be used to help identify the potential host sources of *Cryptosporidium* responsible for an outbreak.

One limitation of the current detection methods is that they do not provide information on the viability or infectivity of the cysts or oocysts. Separate assays have been described for these purposes, which involve observing the degree of excystation or the inclusion/exclusion of specific fluorescent dyes. Other methods require the use of cell culture or animal subjects. Specific variants of the PCR method (reverse transcriptase PCR, quantitative PCR) have also been designed to facilitate quantification of the organisms and to provide estimates of cyst or oocyst viability. In general, these tests can be expensive and difficult to perform and are typically reserved for specific research purposes.

Readers are advised to consult the literature for further information regarding specific methods. Published reviews of molecular techniques used for the detection and identification of *Giardia* and *Cryptosporidium* are similarly available (Caccio, 2003).

Cyanobacteria and their toxins

Several methods are available for the detection of cyanobacteria and microcystins in water and bloom samples. These methods vary markedly in their level of complexity and the level of

information that they provide. Selection of an appropriate method will depend on the type and degree of information required as well as the availability of laboratory facilities and experienced personnel.

Published texts are available that discuss the methods that may be used for the detection of cyanobacteria and microcystins in recreational waters (Chorus and Bartram, 1999; Falconer, 2005). Review articles summarizing the existing and potentially emerging technologies have similarly been published (e.g., McElhiney and Lawton, 2005). Individuals are advised to consult the literature for information on specific methods.

Cell counting

Standard Methods for the Examination of Water and Wastewater (APHA et al., 2005) contains procedures for the enumeration of cyanobacteria (as phytoplankton) in water. Concentrations are determined by direct microscopic count with a counting chamber of known dimensions and then back-calculating to the volume of the original sample. The majority of cyanobacteria are present as colonies or filaments, which can be difficult to distinguish; thus, the use of a trained microscopist with experience in identifying cyanobacteria is recommended.

Microcystin testing

Total toxin analysis requires the determination of both the free and cell-bound microcystins. Thus, initial processing steps are often required to extract and concentrate the toxins in the sample. These may include concentration of the cyanobacterial cells, cell lysis and toxin extraction and purification. A field (commercialized) test kit could be used as a screening tool to determine the presence or absence of toxin in a water supply. If the presence of toxin is detected in a sample using the field test kit, a sample can then be submitted to a recognized laboratory for confirmatory analysis.

The mouse bioassay had historically been used as the primary method for providing evidence of a bloom's toxicity. In recent years, it has been replaced by more sensitive and reliable laboratory methods, such as the protein phosphatase inhibition assay and enzyme-linked immunosorbent assay (ELISA) tests. One drawback of these methods is that they do not provide direct evidence of the sample's toxicity. A number of invertebrate and cell culture assays have been investigated as possible alternatives for the mouse bioassay.

The protein phosphatase inhibition assay is a biochemical assay used for the detection of microcystins based on their ability to inhibit the activity of protein phosphatase enzymes. It is a rapid, sensitive method for the detection of microcystins. The test provides a quantitative estimate of the total microcystins present, but does not provide information on the composition of microcystin variants within the sample. A commercialized kit for the detection of microcystins in water samples is currently available

The ELISA technique is regarded as a very promising method for the detection of microcystins in bloom and water samples. ELISA methods are rapid and sensitive, but are presently not specific enough to distinguish between individual microcystin variants. A number of commercialized kits for the detection of microcystins in water samples are currently available.

Liquid chromatography (LC) coupled with mass spectroscopy (MS) is the most commonly used laboratory method for the identification and quantification of microcystins and represents the reference standard against which other methods are judged. Analysis using this method is time consuming, technically demanding and expensive as it requires specialized equipment. However, standardized procedures have been described, and many analytical laboratories possess the necessary instrumentation. The lack of available standards has limited this method's usefulness in identifying individual microcystin variants.

Recently, PCR-based methods have been described for the detection of cyanobacteria based on primers directed against gene fragments belonging to *Microcystis* spp., as well as the genes responsible for microcystin production (*mcy* genes). These methods have demonstrated some measure of success in the identification and quantification of cyanobacteria and their toxins in bloom samples; however, the techniques are still under development, and further investigation is required.

Appendix C: Composite sampling for faecal contamination

C.1 Description

The Guidelines recommend that when assessing water quality for adherence with the guideline values for the recommended indicators of faecal contamination, a minimum of five samples should be collected, at a minimum frequency of one sample per week.

Composite sampling (collection of multiple samples from across a stretch of beach, combining the samples into one large composite, analysing a subsample of the resulting mixture) presents a possible means for increasing the area covered under a beach monitoring program, while potentially minimizing the costs associated with analysis.

Authorities and beach operators and/or service providers may wish to investigate composite sampling as a tool to potentially improve the utility of their faecal indicator monitoring programs.

C.2 Introduction

With some monitoring programs, single samples may be used to characterize long stretches of swimming water, across several days of swimmer activity. It is known that the fluctuation of water quality can be significant, even over short distances and time periods. Collecting multiple samples more frequently is recommended, because increasing the number of samples increases the reliability of the data (Whitman and Nevers, 2004). The costs associated with increased monitoring, however, can be prohibitive. One solution that has been proposed to address this problem is the use of composite sampling (U.S. EPA, 2005a).

The process of composite sampling requires collecting multiple samples. Equal volumes from each sample are then mixed together to form a composite, which is then analysed as a single sample. This technique can broaden the coverage of a sampling strategy where the analysis of large numbers of samples would otherwise be required. Subsequently, it can be used to increase sampling reliability without significantly affecting the costs associated with monitoring (Patil, 2002). Composite sampling has numerous applications in biomonitoring and environmental sampling and has been used for assessing contamination in a variety of media, including soils, air, water and biological tissue. Recently, investigations have been conducted to explore whether composite sampling can be applied to the assessment of the quality of recreational waters (Kinzelman et al., 2006). Preliminary evidence has indicated that when properly conducted, composite sampling can be used in making water quality decisions with a degree of accuracy comparable to that of traditional sampling regimens.

There are challenges associated with composite sampling that need to be taken into consideration before introducing this technique into a recreational water monitoring program. These are described briefly below.

Potential sources of bias

Composite sampling adds another layer of uncertainty to the water quality results, since a subsample is being used to estimate the average indicator density over all samples, and this

estimate in turn is used to characterize the water quality for the whole beach. It is suggested that including more samples in the composite compensates for the effects of this bias.

There is the potential for the presence of an individual sample with a high concentration to be masked when combined with samples with lower concentrations, owing to the effects of dilution (Kinzelman et al., 2006). Samples from hot spots (areas where poor water quality is likely to be persistent) should not be composited with other samples. The identification of hot spots may be determined by conducting an Environmental Health and Safety Survey (EHSS) or through an initial period of intensive sampling. In programs where hot spots have been characterized and determined as unlikely to occur, the occurrence of a single sample with high concentrations may be considered to be the result of natural, random variability (Kinzelman et al., 2006).

Comparing results with the guideline values

The composite sampling result approximates the arithmetic mean of the indicator counts of the individual samples. However, when analysing bacteriological water quality data, the geometric mean is recommended as the best estimate of central tendency of microbial populations. The guideline values for the recommended indicators of faecal contamination are based on geometric mean values. In determining whether the results are in accordance with the guidelines, operators/service providers or responsible authorities would need to convert the composite result to an approximation of the geometric mean.

Wymer et al. (U.S. EPA, 2005a) pointed out that the difference between the composite value and the geometric mean can be compensated if the variance (v) of the $\log_{10}$ indicator densities is known. An estimate of the variance can be calculated from historical $\log_{10}$ data. Once the variance has been determined, multiplying the count per 100 mL obtained from the composite sample by the factor $10^{-1.15v}$ produces a value that is approximately equivalent to the geometric mean of the individual samples (U.S. EPA, 2005a).

C.3 Study results

Kinzelman et al. (2006) produced swimming beach water quality data comparing the accuracy of composite sampling with traditional monitoring practices at two Lake Michigan beaches in Racine, Wisconsin. Water samples were collected over 68 days in 2003 from two public swimming beaches and analysed for *E. coli* using single-sample analysis with arithmetic and geometric mean averaging and composite sample analysis.

The resulting data indicated that composite sampling appeared to be an effective alternative to traditional monitoring procedures. In general, the value of the composite sample fell within the range of the single-sample values, and the data indicated an approximate 1:1 ratio between the composite sample and the arithmetic mean of the individual samples. In comparing what would have been the ultimate management decision (i.e., issue a water quality advisory or allow the beach to remain open) resulting from the use of composite sampling versus the results from individual analyses (singly or with averaging), the outcome for both methods remained constant at one beach and differed in only two instances at the other. Compositing resulted in additional advisories in both instances. Therefore, compositing appeared to introduce neither bias nor additional variability into the monitoring results (Kinzelman et al., 2006). Verification studies,

performed on a smaller scale in subsequent years, have yielded similar results, and Racine has successfully used composite sample analysis for compliance monitoring since 2004.

C.4 Conclusions

Under the appropriate circumstances, compositing of samples may present a viable alternative to current monitoring schemes that employ a single sample to characterize water quality over long stretches of swimming beaches. Composite sampling may encourage more sampling to take place, thus expanding the coverage of the monitoring program and increasing sampling reliability, while at the same time maintaining the costs associated with monitoring or even lowering them. Monitoring programs that require a large number of samples to be analysed could benefit from adoption of this approach.

Appendix D: Recreational swimming area Environmental Health and Safety Survey (EHSS) checklist

Identification

Beach Name: _____
Address: _____
_____

Responsible Authority: _____
_____

Tel.: _____ Fax: _____ E-mail: _____

Person(s) Conducting Survey: _____

Date: _____ Time: _____

Background Information

Water Body Type: _____

Dimensions of Beach: Length (m): _____ Width (m): _____
Dimensions of Swimming Area: Length (m): _____ Width (m): _____
Number of Sampling Sites: _____

[Attach Map or Aerial Photo of Suitable Scale (including location of sample sites)]

Water Temperature	High/Low (°C): _____	Average (°C):	_____
Prevailing Winds	Direction: _____	Avg. Speed (km/h):	_____
Prevailing Currents	Direction: _____	Avg. Speed (km/h):	_____
Seasonal Rainfall	Total (mm): _____	24-h High (mm):	_____
Wave Height	Average (m): _____	Range (m):	_____

Surrounding Land Uses (check all that apply):

Urban	□	Rural	□	Agricultural (specify): _____	□
Residential	□	Forest	□	Commercial (specify): _____	□
Field	□	Hills/Uplands	□	Industrial (specify): _____	□
Marsh/Swamp	□	Landfill	□	River/Stream/Ditch:	□
Harbour	□	Other: _____	□	Other: _____	□

Microbiological Hazards

Potential Sources of Faecal Contamination

Municipal Sewage Discharges	□	Combined Sewer Overflows (CSOs)	□
Stormwater Drains/Discharges	□	Septic Waste Systems	□
Wastes from Animal Feeding Operations	□		
Other Discharges Containing Faecal Wastes (List):		Other Sewage Collection/Disposal/ Treatment Systems (List):	
_____	□	_____	□
_____	□	_____	□

Stormwater Runoff from:

Agricultural Areas	□	Areas Receiving Sewage Sludge	□
Beach and Surrounding Facilities (e.g., parking)	□	Other: _____	□
		Other: _____	□

Other Environmental Sources:

Discharging Rivers/Streams/Creeks	□	
Birds (e.g., gulls, ducks, geese, other)	□	(#'s: None Low Med High [circle one])
Other wild animals	□	(#'s: None Low Med High [circle one])
Pets	□	(#'s: None Low Med High [circle one])
Swimmers	□	(#'s: Low Med High [circle one])
Other: _____	□	
Other: _____	□	

Items for Consideration during the Resulting Risk Assessment:

- Proximity of potential contamination sources to the swimming area.
- Potential for contamination sources to have an impact on the swimming area (including an indication of their risk priority: Low, Medium, High).
- Evaluation of water quality according to historical microbiological data (e.g., frequency of exceedances of the guideline values for the recommended indicators of faecal contamination [e.g., continuous/periodic/sporadic]).
- Discharges: Assessment of such factors as volume, flow rate, treatment type, applicable indicator standards, periodicity (continuous, sporadic) and predictability.
- Effects of rainfall: Levels triggering contamination events and typical event duration.
- Assessment of swimming area circulation: Effect of onshore winds, tides, currents, flow patterns in transporting faecal contamination to and entrapping it within the swimming area.
- Animals and birds: Assessment of their types, numbers and droppings.
- Impact of swimmers on water quality—numbers, ages.
- Assessment of potential barriers: Barrier types and points at which they may be applied to reduce impact of the contamination source and/or swimmer exposure.

Chemical Hazards

Potential Sources of Chemical Contamination

Commercial/Industrial Discharges	□	Marinas	□
Motorized Watercraft	□	Other: _____	□
Other: _____	□		

Stormwater Runoff From:

Areas subject to Pesticide Application	□	Urban Areas	□
		Other: _____	□
Areas subject to Fertilizer Application	□		

Items for Consideration during the Resulting Risk Assessment:

- Proximity of potential contamination sources to the bathing area.
- Potential for contamination sources to have an impact on the swimming area (including an indication of their risk priority: Low, Medium, High).
- Discharges: Assessment of such factors as volume, flow rate, treatment type, periodicity (continuous, sporadic) and predictability.
- Effects of rainfall: Levels triggering contamination events and typical event duration.
- Assessment of swimming area circulation: Effect of onshore winds, tides, currents, flow patterns in potentially transporting chemical contamination to and entrapping it within the swimming area.
- Motorized watercraft: Assessment of their types and numbers.
- Assessment of potential barriers: Barrier types and points at which they may be applied to reduce impact of the contamination source and/or swimmer exposure.

Other Biological Hazards

Other Biological Hazards Known to Affect the Recreational Water Area
(Presence may be continuous, seasonal or sporadic.)

Cyanobacterial Blooms	□	Schistosomes (Swimmer's Itch)	□
Large Numbers of Aquatic Plants	□	Other (specify): _____	□
Other (specify): _____	□		

Items for Consideration during the Resulting Risk Assessment:

- Seasonal nature of the hazard: continuous, annual, sporadic.
- Presence of contributing factors (as applicable): water conditions, local geography, temperatures, nutrient levels, presence of appropriate host species.
- Assessment of potential barriers to control hazard and/or reduce human exposure in areas/during times of increased risk.

Physical Hazards and Aesthetic Considerations

Subsurface Hazards:

Steep Slopes or Dropoffs	□	Depths greater than 4.5 m	□
Large Rocks	□	Slippery or Uneven Bottom	□
Other: _____	□		
Other: _____	□		

Water Conditions:

Strong Currents or Rip Tides	□	Undertows	□

Other:

Litter on Beach	□	(None Low Med High [circle one])
Floating Debris	□	(None Low Med High [circle one])
Broken Glass or Other Sharp Objects	□	(None Low Med High [circle one])
Medical Wastes	□	(None Low Med High [circle one])
Seaweed/Algae on Beach	□	(None Low Med High [circle one])

Vehicles Permitted on Beach or Near Bathing Area:

Automobiles	Y / N	Boats/Watercraft	Y / N
		Specify:_____	

Items for Consideration during the Resulting Risk Assessment:

- Assessment of the physical characteristics of the beach and their potential impacts on safe enjoyable use of the area. Includes evaluation of physical layout (geography, topography), composition of shoreline and bottom material, influence of existing structures.
- Assessment of potential risks posed by specific hazards/factors in causing injury or illness or otherwise interfering with the enjoyable use of the area.
- Shoreline and water free from obstructions and of sufficient clarity to permit viewing of persons who may be in distress.
- Assessment of the nature and origin of litter and floating debris.
- Applicable physical and aesthetic parameters (pH, temperature, turbidity, colour, clarity, litter) in agreement with recommendations given in the *Guidelines for Canadian Recreational Water Quality*.
- Assessment of potential barriers to control hazard and/or reduce human exposure in areas/during times of increased risk.

Facilities and Safety Provisions

Facilities:

Toilets	#: ____	□	Showers	#: ____	□
Drinking Water Fountains	#: ____	□	Litter Bins	#: ____	□
Other:	#: ____	□	Other	#: ____	□

Access for Persons with Disabilities □

Safety Provisions:

Lifeguard Stations	#: ____	□	Lifesaving Equipment	#: ____	□
Emergency Telephone	#: ____	□	First Aid Stations	#: ____	□

Signs/Communication Materials:

Beach Posting/Suitability for Swimming	□	Emergency Contact Information	□
Other Hazards (list):		Other: _____	□
_____	□		
_____	□		

Formal Procedures or Reporting Mechanisms in Place to Deal with:

Municipal or Industrial Spills/Discharges/Treatment Bypasses	□
Waterborne Disease Outbreaks	□
Swimmer Injuries	□

Items for Consideration during the Resulting Risk Assessment:

- Assessment of the adequacy of facilities and safety provisions.
- Evaluation of signs and other materials for public communication: Message clear and concise, signs placed in locations highly visible to the public.

Date: Time:

Appendix E: Example: Sample collection and reporting form

Beach Name: _____ Address: _____

Responsible Authority: _____ Contact Information: _____

Person Collecting Sample: _____

Site	Indicator Counts □ *E. coli* □ Enterococci

Geometric Mean:	
Period Covered:	

Air Temperature (°C): _____

Water Temperature (°C): _____

pH: _____

Salinity: _____

Turbidity: □ Low □ Medium □ High
Value (NTU): _____

Wave Activity: □ Low □ Medium □ High
Height (m): _____

Rainfall: (during sampling) □ None □ Medium □ Low □ High

Rainfall: (Past 48 h) □ None □ Medium □ Low □ High
Value (mm): _____

Wind Direction: □ None □ Offshore □ Onshore □ Parallel to Shore

Sunlight: □ Sunny □ Overcast □ Partially Cloudy □ Rainy

Swimmer Density: □ None □ Low □ Medium □ High
Approximate Number: _____

Birds: □ None □ Low □ Medium □ High
▪ gulls
▪ ducks Approximate Number: _____
▪ geese

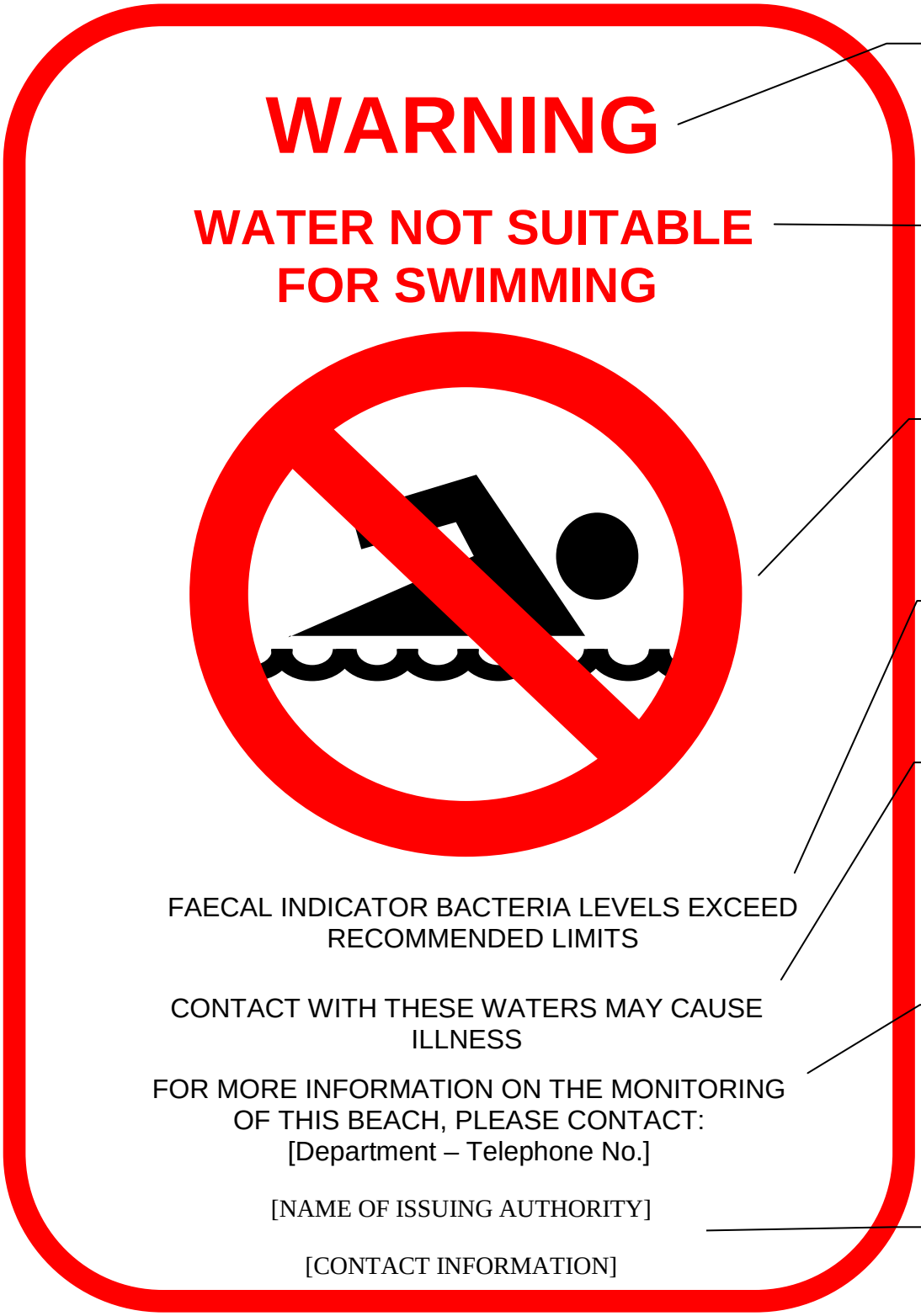
Litter: □ None □ Low □ Medium □ High

Seaweed/ Algae (on beach) □ None □ Low □ Medium □ High

Notes:

Appendix F: Examples: Informative beach signs

F.1 Beach posting: example of an informative swimming advisory sign



F.2 Beach posting: example of an informative water suitable for swimming sign