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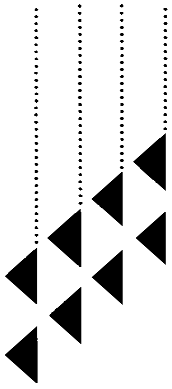
Guidelines for Canadian Recreational Water Quality



Canada

Guidelines for Canadian Recreational Water Quality

Prepared by the
Federal-Provincial Working Group on
Recreational Water Quality of the
Federal-Provincial Advisory Committee on
Environmental and Occupational Health

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Preface

The increasing use of surface waters in Canada for body contact recreational purposes, and the larger number of industrial and municipal wastewater sources entering surface waters, call for guidelines for recreational water quality. In 1988, the Federal-Provincial Advisory Committee on Environmental and Occupational Health requested the formation of a Working Group to revise the recreational water quality guidelines established in 1983.

In preparing this document, the Working Group has thoroughly reviewed the existing criteria, the current indicators of hygienic quality, water quality data from recreational areas in various parts of Canada, and pertinent epidemiological studies. This review took place between April 1988 and July 1989. Maximum limits for indicator organisms in this document are presented on a per-litre basis to conform to the SI (International System of Units) guidelines. It is hoped that the document will serve as a national guideline and that judicious application by responsible operators and authorities will provide a measure of safety for all Canadians.

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1. Purpose and Scope

Recreational waters refer to those natural waters used not only for primary contact activities, such as swimming, windsurfing, and waterskiing, but also for secondary contact activities, such as boating and fishing. In this document, recreational use is defined as any activity involving the intentional immersion (e.g., swimming) or incidental immersion (e.g., waterskiing) of the body, including the head, in natural waters. Natural water is defined as any marine, estuarine or fresh body of water, as well as any artificially constructed flow-through impoundment using untreated natural waters. Because swimming pools are subject to specific management practices and provincial regulations intended to protect public health (e.g., disinfection and construction standards), they are not covered by this publication.

The guidelines deal with health hazards associated with recreational water use, as well as aesthetic and nuisance conditions. Health hazards associated with direct contact with water include infections transmitted by pathogenic microorganisms, as well as injuries and illness due to physical and chemical properties of the water. The guidelines discuss the indicator organisms – enterococci, ***Escherichia coli,*** other fecal coliforms, and coliphages – as well as health risks related to exposure to waterborne pathogenic bacteria, viruses, protozoa, and toxic blue-green algae. Sampling of recreational waters is also addressed. Other sections deal with physical, chemical, and aesthetic characteristics, nuisance organisms, microbiological methods of sampling and analysis, and posting of beaches and other recreational waters.

The limits recommended in this document will be periodically revised or adjusted as new or more significant data become available. They should not be regarded as legally enforceable standards, except when promulgated by the appropriate provincial or federal agency. It is intended that judicious use of these guidelines will result in the provision of safe, attractive recreational waters in all areas of Canada. It is hoped that additional epidemiological studies will be conducted to provide for refinement of the guidelines in the future.

2. General Requirements for Recreational Water Quality

Waters used for recreational purposes should be sufficiently free from microbiological, physical, and chemical hazards to ensure that there is negligible risk to the health and safety of the user. The determination of the risk of disease or harm from microbiological, physical, or chemical hazards is based on a number of factors, including the following:

- environmental health assessments
- epidemiological evidence
- indicator organism limits
- presence of pathogens.

The decision to post a warning to users of recreational areas or to close an area for public use should be made by the Medical Health Officer or other appropriate authority in accordance with the statutes existing in each province. This decision will be based on an assessment of existing hazards using available information on the factors listed above.

2.1 Environmental Health Assessments

An annual environmental health assessment should be carried out prior to the bathing season on the watershed or the area from which water flows to a recreational area, as well as on the recreational area itself. This survey should identify all potential sources of contamination and physical hazards that could affect the recreational area. Appendix 1 provides a suggested checklist for the health inspector or other appropriate authority to use when making an assessment.

Attention should be paid to the following:

- the risk of inadequately treated sewage, fecal matter, or chemical substances entering the water, from either a discharge or a spill
- knowledge of all outfalls or drainage in the area that may contain sewage, including urban storm water and agricultural waste or runoff
- an inspection of the area for physical hazards

- an assessment of the seasonal variability of hazards, the density of bathers, the water temperature, the frequency of change or circulation of the water, changes in water depth, and the occurrence of algal blooms
- the fluctuation of water quality with rainfall (wet and dry conditions)
- a reporting mechanism to ensure that health authorities are informed of any malfunction or change to a municipal, private, or industrial waste treatment facility that might cause a deterioration of the water quality of a bathing area.

2.2 Epidemiological Evidence

The local health authorities responsible for making recommendations for a recreational area should, wherever possible, establish surveillance for bather illness or injuries. This can be established by comprehensive epidemiological studies or by formal and informal reporting from physicians and hospital emergency departments. This surveillance will be increased if there have been reports of suspected illness or injuries. The water quality may be considered impaired and appropriate recommendations made as a result of this surveillance. Procedures for the investigation of illness associated with recreational waters should adhere to the recommendations given in *Procedures to Investigate Waterborne Illness* (International Association of Milk, Food and Environmental Sanitarians, Inc. 1979).

2.3 Indicator Organism Limits

An indicator organism or organisms should be chosen by the local health authority in consultation with the laboratory microbiologists for each area. It is recommended that one of the following indicator organisms be used for routine monitoring of recreational water quality – enterococci, ***Escherichia coli***, or fecal coliforms.

The choice of indicator organism and of enumeration procedures will be determined according to:

- whether the water is marine (salt), fresh, or estuarine (variable salinity)
- the presence of turbidity, which may interfere with microbiological methods
- any known correlation of illness with levels of indicator organisms
- the proportion of fecal coliforms in the area that are ***E. coli***, if fecal coliforms are used as indicator organisms
- local experience of monitoring with a particular organism.

Later sections recommend the limits for each organism and the criteria to assist in the choice of that organism for routine monitoring. Guidelines for sampling and microbiological methods are also discussed.

The decision to carry out routine microbiological monitoring of a recreational area will be made by the local health authorities or other responsible agency, based on the usage of the area, the environmental health assessment, and epidemiological evidence.

2.4 Presence of Pathogens

Tests for pathogenic organisms may be carried out when there have been reports of illnesses of specific etiology, when there is suspected illness of undetermined cause, or when levels of an indicator organism demonstrate a continuous suspected hazard. The tests will help to determine the source of contamination (e.g., sewage pollution, agricultural or urban runoff, bather origin).

The local health authorities should take action when pathogenic organisms are identified in sufficient quantity or frequency to be considered a hazard. Such pathogenic organisms may be ***Aeromonas*** spp., ***Pseudomonas aeruginosa***, ***Staphylococcus aureus***, **Shigella** spp., ***Salmonella*** spp., ***Campylobacter*** spp., ***Giardia*** spp., human viruses, and toxic phytoplankton. An appropriate response should be based on the knowledge of the source of the organism and the probability of the hazard being temporary or continuous.

3. Microbiological Characteristics

Recreational waters may be contaminated by direct human contact and by waterborne pollutants from external sources (e.g., sewage, storm water and agricultural runoff). Many epidemiological studies have identified gastro-intestinal and upper respiratory illnesses in bathers that were a result of such contamination. The indicator organisms recommended in these guidelines are surrogates for the presence of pathogenic organisms that may cause gastrointestinal illnesses.

The ideal indicator of fecal contamination of recreational waters would be one of the enteric pathogens, such as ***Salmonella*** or Norwalk virus, most frequently responsible for waterborne diseases. However, because these are usually present at low levels and are irregularly distributed, even during disease outbreaks, they are difficult to isolate and quantify. Moreover, the absence of one pathogen does not necessarily ensure that other enteric pathogens are also absent. In addition, testing for every possible waterborne disease-causing microorganism would be prohibitively expensive in terms of both time and money. For these reasons, it is common practice to monitor the other more plentiful but non-pathogenic bacteria present in human and animal feces. The presence of elevated numbers of these bacteria in the aquatic environment is indicative of fecal contamination and the possible presence of enteric pathogens.

The best indicators of the presence of enteric pathogens in fecal pollution sources should have the following properties (National Academy of Sciences 1977; Cabelli *et al*. 1983; Elliot and Colwell 1985):

- present in fecal-contaminated waters when enteric pathogens are present but in greater numbers
- incapable of growth in the aquatic environment but capable of surviving longer than pathogens
- equally or more resistant to disinfection than pathogens
- easily and accurately enumerated
- applicable to all types of natural recreational waters (e.g., fresh, estuarine, and marine)
- absent from non-polluted waters and exclusively associated with animal and human fecal wastes

- density of indicator should be directly correlated with the degree of fecal contamination
- density of indicator should be quantitatively related to swimming-associated illnesses.

In the past, the most widely used indicator of recreational water quality was total coliforms. However, because this group does not conform to most of the above characteristics, it is now considered unsuitable. For example, many of the genera in this group, such as ***Klebsiella***, ***Citrobacter***, ***Enterobacter***, and ***Aeromonas***, are not unique to human or animal feces but are commonly present in unpolluted surface waters (Boyd and Boyd 1962; Goodrich *et al*. 1970). ***Escherichia coli***, enterococci, and, to a lesser degree, fecal coliforms are currently considered the best fecal indicators, because they most closely fit the above characteristics. Maximum acceptable concentrations of these indicator organisms are provided below.

The microbiological parameters selected for review are grouped as either fecal indicators (Sections 3.1 and 3.2), coliphages (Section 3.3) and pathogenic organisms (Section 3.4). The list is not intended to be comprehensive; some control agencies may wish to monitor additional parameters to consider regional interests.

3.1 Indicator Organisms for Fresh Waters

3.1.1 *Escherichia coli* and fecal coliforms

Maximum Limits

The geometric mean of at least 5 samples, taken during a period not to exceed 30 days, should not exceed 2000 ***E. coli***/L**.** Resampling should be performed when any sample exceeds 4000 ***E. coli***/L**.** The Working Group agreed that the tests to be used to fulfil these requirements are as follows:

1. When experience has shown that greater than 90 per cent of the fecal coliforms are ***E. coli***, either fecal coliform or ***E. coli*** may be determined.

2. When less than 90 per cent of the fecal coliforms are ***E. coli***, only ***E. coli*** may be determined.

Criteria

Description

For several years, recreational water quality experts in Canada have recognized ***E. coli*** as the indicator of choice for fecal contamination (Department of National Health and Welfare 1983). However, because its enumeration required complicated, time-consuming, and expensive techniques, total coliforms and, more recently, fecal coliforms were established as the indicators of fecal contamination. Fecal coliforms have all the properties of total coliforms, plus they are able to ferment lactose with the production of gas in 24 hours at an incubation temperature of 44.5°C. In practice, fecal coliforms are enumerated either by the most probable number (MPN) multiple tube fermentation test or by the membrane filtration (MF) method (American Public Health Association 1989) (see Chapter 5.0). The organisms most often recovered by these methods are ***E. coli*** and ***Klebsiella*** spp.

The development of the mTEC method specifically designed to enumerate ***E. coli*** (Shaw and Cabelli 1980; Dufour *et al*. 1981) has prompted a re-evaluation of its use for monitoring recreational water quality. In recent tests, the mTEC method has proved to be more effective for ***E. coli*** isolation than the fecal coliform test (mFC method) for both freshwater samples (Pagel *et al*. 1982) and marine water samples, even when resuscitation steps were used with the mFC method (Rippey *et al*. 1987). The U.S. Environmental Protection Agency (1986) recently proposed the use of ***E. coli*** as an indicator for monitoring fecal contamination of fresh recreational waters. The incorporation of methylumbelliferone glucuronide (MUG) into various fecal coliform media has also been examined recently (Feng and Hartman 1982; Freier and Hartman 1987; Mates and Schaffer 1988). The enzyme β-glucuronidase, unique to ***E. coli*** and some strains of ***Salmonella*** and ***Shigella***, will metabolize MUG to produce 4-methylumbelliferone, which fluoresces under longwave ultraviolet light.

Escherichia coli comprises about 97 per cent of the coliform organisms in human feces, with ***Klebsiella*** spp. comprising 1.5 per cent and ***Enterobacter*** and ***Citrobacter*** spp. together comprising another 1.7 per cent (Dufour 1977). Studies by Geldreich (1970) demonstrated that fecal coliforms represented 93 to 99 per cent of the coliforms in feces from humans, poultry, cats, dogs, and rodents. Dufour (1977) demonstrated that ***E. coli*** represented between 90 and 100 per cent of all coliforms in feces from 8 species of domestic animals, including chickens. ***Klebsiella*** spp. constituted a significant proportion of the coliforms only in feces from goats (8 per cent of the total coliforms) and pigs (6.8 per cent).

The fecal coliform and ***E. coli*** tests do not differentiate between animal and human fecal pollution (Wolf 1972; Dufour 1977). If this distinction is necessary, ancillary tests such as the speciation of fecal streptococcal isolates must be performed.

Soil has been shown to be variably contaminated with fecal coliforms due to animal fecal contamination (Geldreich *et al*. 1962; Van Donsel *et al*. 1967) and is known to contribute very significant pollution to storm water runoff (Environment Canada/Ontario Ministry of the Environment 1978; Qureshi and Dutka 1979) and recreational waters (Bastein *et al*. 1974; Cabelli 1977). Runoff from residential areas is usually as contaminated with fecal coliforms and pathogens as dilute sewage (Qureshi and Dutka 1979) and can thus be a significant health hazard when discharged in the vicinity of recreational waters. To counter this problem, some Canadian cities with urban beaches have adopted beach closure policies based on rainfall data. For example, beaches along the Rideau River in Ottawa are closed for 24 and 48 hours following a rainfall of greater than 10 and 20 mm, respectively, in the preceding 24 hours (Corber 1988). This approach has the advantage of posting beaches when the health risk to bathers is highest.

Although ***E. coli*** is undisputed as the fecal indicator of choice, some of the fecal coliform tests used will also enumerate ***Klebsiella*** spp., which are not restricted to fecal sources. Numerous studies have demonstrated the ability of ***Klebsiella*** spp. to survive and replicate in organic-rich environments, including waters receiving effluents from pulp and paper mills (Huntley *et al*. 1976; Rokosh *et al*. 1977; Bell *et al*. 1978) and textile industries (Dufour and Cabelli 1976; Vlassoff 1977). In the past, this disadvantage was somewhat tempered by the fact that ***K. pneumoniae*** was considered pathogenic for debilitated hosts and that environmental isolates of this species were virtually indistinguishable from clinical isolates (Matsen *et al*. 1974; Dufour and Cabelli 1976). However, a recent review of the ***Klebsiella*** literature by Duncan (1988) has convincingly demonstrated that there is no evidence to suggest that any community infections have resulted from exposure to ***Klebsiella*** spp. in the natural environment. The concerns that environmental strains of ***Klebsiella*** pose a health hazard seem to be based on hospital situations that cannot be applied to the community setting. In addition, ***Klebsiella*** may not represent a significant proportion of fecal coliforms in most Canadian recreational waters. Studies in Ontario by Vlassoff (1981) on more than 7700 water samples indicated that 91.4 per cent of fecal coliform isolates were identified as ***E. coli***. However, this value could be too high in recreational waters receiving pulp and paper mill effluents. A study by Caplenas and Kanarek (1984) demonstrated that effluents from pulp and paper mills in northern Wisconsin contained up to 90 per cent non-fecal ***K. pneumoniae*** when fecal coliforms were enumerated by the mFC medium. In 1985, analyses of beach samples impacted by pulp and paper mill effluents in the vicinity of Thunder Bay revealed that most of the fecal coliforms, in

some cases 100 per cent, were ***Klebsiella*** (Young 1989). In 1988, an examination of 162 samples from 19 beaches in the vicinity of St. Catharines that were impacted by pulp and paper mill effluents indicated that only 37 per cent of fecal coliforms were ***E. coli*** (Brodsky 1989).

Several jurisdictions and agencies have promulgated regulations or suggested limits for fecal coliforms. Many agencies, in addition to specifying a limit on the geometric mean, also specify that no more than a certain percentage (usually 10 per cent) should exceed a specific limit (usually twice the specified mean). Provincial authorities may wish to include this additional condition in their guidelines or regulations. It has been pointed out that fecal coliforms are usually sufficiently dispersed so that this percentage, rather than the established mean, is the factor most often exceeded (Fuhs 1975).

Association with Pathogens

Fecal coliforms are considered useful as indicators because they are present in virtually all warm-blooded animals, including humans, in numbers far exceeding the numbers of pathogens. Some attempts have been made to quantify the relationship of fecal coliforms to pathogens, with limited success. The pathogen most extensively studied in relation to indicator densities is ***Salmonella***, mainly because the methodology has been available for a long time. Geldreich (1970) compiled the results of several studies in which the fecal coliform density per 100 mL and the frequency of ***Salmonella*** detection were compared. In fresh water, salmonellae were found in 27.6 per cent of samples where the fecal coliform density was below 200/100 mL, in 85.2 per cent of samples where the fecal coliform density was between 201 and 2000/100 mL, and in 98.1 per cent of samples where it was over 2000/100 mL. In estuarine waters, the results were not as definitive. When fecal coliform densities were less than 200/100 mL, the probability of finding ***Salmonella*** was 28.4 per cent; however, if fecal coliform densities were greater than 2000/100 mL, the probability of finding ***Salmonella*** was only 60 per cent.

A recent study has supported these results. Menon (1985), in an investigation of a tidal river in Nova Scotia receiving municipal and food processing effluents, reported that ***Salmonella*** spp. were always detected when fecal coliform levels were greater than 2000/100 mL and were occasionally detected when fecal coliform levels were greater than 200/100 mL. Conversely, Payment *et al*. (1982) found no relationship between the presence of ***Salmonella*** (most isolates were ***S. typhimurium***) and other bacterial indicators, including fecal coliforms, at four freshwater bathing beaches in Quebec.

In general, samples containing high concentrations of fecal coliforms will likely also contain ***Salmonella***, but the absence of fecal coliforms does not necessarily indicate that ***Salmonella*** or other pathogens will be absent. This relationship is also subject to considerable regional variation. Other comparative studies have been conducted with ***Pseudomonas aeruginosa*** (Cabelli *et*

al. 1976; Sherry 1986), ***Vibrio parahaemolyticus*** (Robertson and Tobin 1983; Larsen and Willeberg 1984), ***Candida albicans*** (Sherry 1986), ***Aeromonas hydrophila*** (Seidler *et al*. 1980; Larsen and Willeberg 1984), and ***Campylobacter jejuni*** (Hill and Grimes 1984; Carter *et al*. 1987).

Enteric viruses have received considerable notoriety as a hazard associated with the use of recreational water; unfortunately, the relative incidence of virus and fecal coliforms has not been constant. In one sewage treatment plant studied extensively, ratios of fecal coliform to viruses ranged from 7500:1 to 2 900 000:1 over a 2-month period (Berg and Metcalf 1978). Sattar (1978a) found a large variation among the fecal coliform to enterovirus ratios from a variety of sources: raw sewage (between 1.1×10^6 and 43×10^6), chlorinated effluent (between 8.5×10^2 and 2.1×10^4), and 2 beaches on the Ottawa River (between 1.2×10^3 and 1.2×10^6). Analysis of the data gathered from freshwater beaches in Quebec by Payment *et al*. (1982) did not reveal any correlations between the presence of enteric viruses and enteric bacteria, including fecal coliforms. It is generally agreed at this time that most bacteriological indicators do not correlate well with virus levels, even though the presence of large numbers of coliforms may indicate the probable presence of enteric viruses. However, the converse – that the absence of fecal coliforms indicates that enteric viruses are not present – cannot be ensured (Berg and Metcalf 1978).

Related Epidemiological Studies

In order to provide rational microbiological guidelines for recreational water, it is necessary to establish that there is some degree of health risk associated with a certain level of contamination. Again, because pathogens are sparse and difficult to quantify, fecal coliforms, including ***E. coli***, have been used in all major epidemiological studies.

The first widely publicized epidemiological studies in North America were carried out by the U.S. Public Health Service (Stevenson 1953). These studies were conducted at 2 freshwater sites on Lake Michigan and the Ohio River and at 2 marine sites. Swimming-associated gastroenteritis was not observed at the marine sites or at the Lake Michigan site. However, the Ohio River study showed increased gastro-intestinal illness at median coliform densities of about 2300/100 mL. Data subsequently collected from the Ohio River site during the 1960s suggested that a level of 400 fecal coliforms was approximately equivalent to the threshold number of 2300 total coliforms (Geldreich 1966). Following application of a safety factor, the guideline of 200 fecal coliforms/100 mL was developed. Although the design of Stevenson's studies has been severely criticized, it is apparent that these studies have formed the basis of most guidelines in use today.

Since then, other microbiological-epidemiological studies have been conducted for recreational water based on fecal coliform concentrations. Seyfried *et al*. (1985a, 1985b) conducted an investigation on 10 beaches in Ontario. In total, 8402 swimmers and non-swimmers were interviewed. Crude morbidity rates for all illnesses, including respiratory, gastrointestinal, eye and ear, skin, and allergy illnesses, were roughly 2.4 times greater for swimmers than for non-swimmers, and 2.6 times greater for swimmers who immersed their heads than for those that did not. Morbidity among swimmers was shown to be related to fecal coliform counts ($r = 0.284$), even though the mean fecal coliform densities in the water (76/100 mL) were within the accepted guidelines. The total staphylococcal count was also related to morbidity among swimmers ($r = 0.439$). Levels of all groups of bacteria surveyed were at least 10 times greater in the sediment than in the surface waters, indicating that the resuspension of sediment may be a significant source of bacteria in recreational waters. The authors concluded that sediment samples should be analyzed during microbiological investigations.

During the summer of 1983, a microbiological-epidemiological study of three coastal beaches in Israel was conducted by Fattal *et al*. (1986). Although all beaches complied with Israel Ministry of Health bacteriological standards, analysis of the results indicated that the incidence of gastroenteritis among swimmers, particularly in the 0- to 4-year age group, was related to elevated densities of ***E. coli*** as well as enterococci and staphylococci. The incidence of ear infections in swimmers of all ages was also related to elevated densities of fecal coliforms, ***E. coli***, and enterococci.

In the United States, a series of controlled epidemiological studies has been performed by the Environmental Protection Agency at both marine (Cabelli 1983) and freshwater (Dufour 1984) beaches. A summary of the studies and a presentation of proposed criteria for recreational waters were recently published (U.S. Environmental Protection Agency 1986). Two beaches were selected at each site, one relatively unpolluted and the other impacted with point or non-point sources of fecal contamination. In the freshwater studies, analyses were conducted for fecal coliforms, ***E. coli***, and enterococci; in the saltwater studies, analyses were conducted for these indicators as well as several other waterborne pathogens. Regression coefficients were determined for levels of each of the parameters and the differential (swimmers minus non- swimmers) gastrointestinal symptom rates. On the basis of this pooled information, it was concluded that, for marine beaches, enterococci presented the best relationship with gastrointestinal symptoms ($r = 0.75$). At freshwater beaches, the best correlations were obtained with ***E. coli*** ($r = 0.80$) and enterococci ($r = 0.74$). Based on these investigations, the U.S. Environmental Protection Agency proposed that at freshwater beaches, the 30-day geometric mean should not exceed 126 ***E. coli***/100 mL or 33 enterococci/100 mL.

From the pooled data, it can be calculated that in fresh waters, the seasonal risk of gastrointestinal illness per 1000 bathers ($\underline{y}$) is related to the density of ***E. coli***/100 mL ($\underline{x}$) by the following relationship:

$$\underline{y} = 9.40 (\log \underline{x}) - 11.74$$

This investigation has been one of the most extensive microbiological-epidemiological studies ever conducted, but it does have certain limitations. Although ***E. coli*** and enterococci are suitable for indicating the risk of gastroenteritis, this illness represents only about 30 per cent of overall swimmer-associated morbidity. ***Escherichia coli*** and enterococci are not suitable indicators for upper respiratory or dermal infections caused by ***Pseudomonas*** and ***Staphylococcus*** spp. Furthermore, the epidemiological information collected on the swimmers and non-swimmers and data on the water quality were averaged over an entire bathing season. Thus, the bacteriological data cannot be used to assess the risk of gastroenteritis on any specific day.

Other epidemiological studies have also noted elevated levels of fecal coliforms, including ***E. coli***, associated with illness. An investigation by Philipp *et al*. (1985) demonstrated that snorkel swimming in fecally contaminated water presents a significant risk to health. In this British study, 27 per cent of the swimmers who participated in a competition experienced gastrointestinal symptoms within 48 hours of entering the water. The incidence of these symptoms was significantly greater than their incidence in the control populations. ***Escherichia coli*** concentrations in water samples collected during the event averaged 1800/100 mL. In light of the fact that this value still conformed with the European Economic Community guidelines, the authors suggested that an appraisal of these guidelines was warranted.

Dewailly *et al*. (1986) conducted an epidemiological study demonstrating certain health risks associated with windsurfing on water contaminated with sewage. Seventy-nine windsurfers and 41 controls were monitored over a 9-day competition for the occurrence of gastroenteritis, otitis, conjunctivitis, and skin infections. Relative risks were 5.5 for gastroenteritis and 2.9 for one or more of the above symptoms. This study also demonstrated that the relative risk increased with the reported number of falls in the water. Based on hydrodynamic simulation using tide levels and actual fecal coliform counts, mean densities of fecal coliforms at high tides during the competition were estimated to be 1000/100 mL.

A recent exercise in the United Kingdom (Brown *et al*. 1987) at two seaside resorts, although not a controlled epidemiological study, demonstrated that swimmers at resort A were more likely to develop stomach upsets, nausea, diarrhea, or headaches than either non-swimmers at the same resort or all vacationers at resort B. Furthermore, swimmers who had immersed their heads

at resort A were most likely of all respondents to have reported gastrointestinal symptoms. The resorts investigated were chosen on the basis of existing microbio-logical data that indicated that fecal coliform concentrations at resort A were 440/100 mL and at resort B were only 10/100 mL.

Although all of the above epidemiological studies were comprehensive and detailed, they used different methods and produced data that are not reproducible or comparable.

Occurrence in the Aquatic Environment

Although most Canadian recreational waters are of high microbiological quality, certain waters are contaminated throughout part, or all, of the bathing season. Fecal coliform values range from nearly 0/100 mL in isolated areas to several thousand per 100 mL in areas directly impacted by sewage discharges (Payment *et al*. 1982; Ontario Ministry of the Environment 1984; Williamson 1988; Smith 1988). In temperate recreational waters, 63 to 100 per cent of the fecal coliforms appear to be ***E. coli***, but this range can be affected by contamination from industrial effluents, particularly pulp and paper and textile mills.

Escherichia coli has been enumerated at a number of recreational beaches in Manitoba and Ontario (Sekla *et al*. 1987; Brodsky 1989; Palmateer 1989; Young 1989).

Summary

1. In fresh waters, ***E. coli*** is the best available indicator of fecal contamination from warm-blooded animals.

2. ***Klebsiella*** is not a good indicator of fecal contamination, but it may be present at high levels in certain industrial wastes (e.g., pulp and paper mills, food processing plants). It is not likely to cause infection or illness in healthy individuals.

3. Where there is evidence that greater than 90 per cent of the fecal coliforms are ***E. coli***, the ***E. coli*** and fecal coliform tests will be considered equivalent.

4. The presence of ***E. coli*** is associated with bather-associated illness, but its absence cannot be equated with the lack of risk of illness.

5. Current microbiological-epidemiological studies are not sufficiently validated to allow calculation of risk levels. However, there is some evidence for increased risk of illness from bathing compared with non-bathing (i.e., wading or remaining on the beach).

6. The 1983 guidelines were, in principle, based on the definitive fecal coliform, ***E. coli***. However, at that time, the more general fecal coliform test was considered the method of choice. The Working Group reaffirms that ***E. coli*** is the indicator of choice and recognizes that either the ***E. coli*** test or the fecal coliform test may be used to enumerate this organism, depending on the circumstances. The maximum acceptable concentration of 2000 ***E. coli*** (or fecal coliforms)/L can be calculated to correspond to a seasonal gastrointestinal illness of 1 to 2 per cent, based on the U.S. Environmental Protection Agency studies.

3.2 Indicator Organisms for Marine Waters

3.2.1 Enterococci

Maximum Limits

The geometric mean of at least five samples, taken during a period not to exceed 30 days, should not exceed 350 enterococci/L. Resampling should be performed when any sample exceeds 700 enterococci/L. However, if it can be demonstrated that ***E. coli*** or fecal coliforms can adequately demonstrate the presence of fecal contamination in marine waters, then the ***E. coli*** or fecal coliform maximum limit for fresh waters may be used. If there is any doubt, samples should be examined for both sets of indicators for extended periods to determine if a positive relationship exists.

Criteria

Description

Enterococci are large, ovoid, Gram-positive bacteria that are generally present in chains. The term ***enterococci*** refers to those species of the fecal streptococcal group that conform to the biochemical characteristics of the Sherman criteria (Clausen *et al*. 1977). They grow at temperatures between 10 and 45°C, survive exposure to 60°C for at least 30 minutes, and grow at pH 9.6 and in 6.5 per cent NaCl. This subgroup, which includes ***Streptococcus faecium*** and ***S. faecalis***, occurs in significant quantities in both human and animal feces. ***Streptococcus avium*** and ***S. gallinarium***, found principally in bird feces, are also classified as enterococci in *Bergey's Manual of Systematic Bacteriology* (1986). In the previous guidelines, the entire fecal streptococcal group, both enterococcal and non-enterococcal species, was addressed. Because the non-enterococcal subgroup includes species that normally occur only in animal feces (e.g., ***S. bovis, S. equinis***), the more specific enterococcal subgroup will be considered in these guidelines. However, the presence of enterococci unique to animal feces may also indicate the presence of pathogenic microorganisms infectious to both humans and animals.

In the past, the main role of the fecal streptococci was in the use of the fecal coliform to fecal Streptococcus ratio as an indicator of the nature of the fecal source (Geldreich 1976; Clausen *et al*. 1977). However, many factors, for example, the differential die-off rates between these two groups in the natural environment, make the routine use of this ratio highly questionable, if not inaccurate. Recent experience indicates that the identification of enterococcal isolates is more useful in the determination of the type, source, and degree of fecal contamination (Rutkowski and Sjogren 1987).

Of all the microorganisms considered as suitable recreational water quality indicators, the enterococci most closely satisfy the desirable characteristics presented in the introduction to this chapter. Enterococci are exclusively associated with fecal wastes. They survive much longer than the other indicators in water and sediment (McFeters *et al*. 1974; Lessard and Sieburth 1983). Enterococci are also more resistant to sewage treatment, including chlorination, and thus may be more sensitive indicators of the survival of enteric pathogens and viruses (Cohen and Shuval 1973). As well, a strong correlation between the concentration of enterococci in marine waters and the risk of gastrointestinal infection has been demonstrated (Cabelli 1983). A membrane filtration method for the enumeration of enterococci in marine waters has recently been described in detail (U.S. Environmental Protection Agency 1985).

Epidemiological Studies

During the summer of 1983, a microbiological-epidemiological examination of 3 coastal beaches in Israel was conducted by Fattal *et al*. (1986). Although all beaches complied with Israel Ministry of Health bacteriological standards, analysis of the results indicated that the incidence of gastroenteritis among swimmers, particularly in the 0- to 4-year age group, was related to elevated densities of indicator organisms, most notably the enterococci ($\underline{p} < 0.03$).

A series of prospective epidemiological studies has also been performed in the United States at 3 marine and 2 freshwater sites (U.S. Environmental Protection Agency 1986). Two beaches were selected at each site, 1 relatively unpolluted and the other receiving point or non-point fecal contamination. Regression coefficients were determined for levels of each of the indicators measured and the differential (swimmers minus non-swimmers) gastrointestinal symptom rates. On the basis of results pooled for the entire season, enterococci displayed the best correlation with these symptoms at marine beaches ($\underline{r} = 0.75$). At freshwater beaches, the best correlations were obtained with ***E. coli*** ($\underline{r} = 0.80$) and enterococci ($\underline{r} = 0.74$).

From the pooled data, it can be calculated that in marine waters, the seasonal risk of gastrointestinal illness per 1000 swimmers ($\underline{y}$) is related to the densities of enterococci per 100 mL ($\underline{x}$) by the following relationship:

$$\underline{y} = 0.20 + 12.17 (\log \underline{x})$$

Based on these investigations, the U.S. Environmental Protection Agency proposed that at marine beaches, the 30-day enterococci geometric mean should not exceed 35/100 mL.

This investigation has probably been one of the most extensive epidemiological studies conducted to date on recreational beaches, but it does have certain limitations. In developing the model, it was assumed that the non-swimming and swimming populations were equally susceptible to illness acquired from other sources. Although enterococci appear suitable for estimating the risk of gastrointestinal illness, they are not suitable indicators for the more prevalent upper respiratory or dermal infections. Furthermore, the data collected on both the participants and the water quality were averaged over the entire bathing season. Thus, the indicator data cannot be used to assess the risk of gastroenteritis on any specific day.

Occurrence in the Aquatic Environment

There have been a few published investigations in Canada on the distribution of fecal streptococci and enterococci in the marine environment. In a continuing study of the use of enterococci as a suitable indicator of water quality at marine recreational beaches, Gibson and Smith (1988) described the distribution of enterococci at 26 beaches in the Vancouver region. Only 1.6 per cent of the 30-day geometric means exceeded the 35 enterococci/100 mL limit proposed by the U.S. Environmental Protection Agency. In 1988, fecal streptococcal concentrations were also monitored at eight marine beaches along the Northumberland Strait, New Brunswick (Allen 1989). The overall geometric mean was only 3.5/100 mL, and fecal streptococci were absent in 60 per cent of the samples.

Summary

1. In marine waters, the enterococci group is the best available indicator of fecal contamination from warm-blooded animals.

2. Fecal coliforms do not survive well in marine waters and thus may not be reliable indicators of fecal contamination.

3. Enterococci survive longer than fecal coliforms in marine waters and thus are preferred when there is considerable time or distance between the source of fecal pollution and the bathing area.

4. There is a positive correlation between gastrointestinal illness and levels of enterococci in marine waters, but the absence of enterococci does not indicate a lack of risk.

5. Based on the U.S. Environmental Protection Agency epidemiological study, a seasonal geometric mean of 35 enterococci/100 mL corresponds to a seasonal gastrointestinal illness rate of 1 to 2 per cent. Because fecal coliforms do not survive well in marine waters, the use of the fresh water maximum limit may increase the risk of illness.

3.3 Coliphages

Maximum Limits

No limits are specified for coliphages in recreational waters.

Criteria

Description

Bacteriophages are virus-like entities that invade bacterial cells. Coliphage is the general name applied to bacteriophages that attack bacteria of the coliform group. Bradley (1967) described 6 morphological types of coliphages and suggested that the diversity among coliphages was greater than the known diversity among mammalian viruses. They differ in size, shape, site of attachment, and genetic material (e.g., single-stranded DNA, double-stranded DNA). The numbers of phages replicated may vary from 100 to several thousand.

Association with Pathogens

An excellent review of the literature on bacteriophages as indicators of bacterial and viral contamination has been published by Scarpino (1975). In the review, Scarpino reported that "correlations appear to exist in fresh and marine waters between bacterial pathogens such as ***Salmonella*** and ***Shigella*** species and fecal indication bacteria such as ***E. coli*** and their bacteriophages." Studies by Kott *et al*. (1974, 1978), Petrovicova *et al*. (1988), Dutka *et al*. (1987), and Borrego *et al*. (1987b) have also indicated a correlation between the presence of coliphage and other bacteriophages and the presence of viruses and pathogenic bacteria in rivers, lakes, and sewage effluents.

Occurrence in the Aquatic Environment

Recent research studies on four continents have found the presence of coliphages and bacteriophages in natural river waters, potable water, and coliform-free potable waters. Implications of these observations are that human enteric viruses could also survive in those waters. A Czechoslovakian study (Simkova and Cervenka 1981) has also shown that both coliphages and enteroviruses can survive for similar long periods of time in river water, and thus coliphages can be used as long-term indicators of enterovirus contamination, even in the presence of chemical pollution. The view of Russian researchers (Petrovicova *et al*. 1988) is that "an increase of coliphage numbers in sewage and surface and recreational (swimming pool) waters shall be considered as an indication for aimed virological studies."

It may be useful to include coliphage and bacteriophage enumeration as part of the continuing evaluation of the impact of fecal pollution on recreational waters. Dutka *et al*. (1987) suggested that coliphage levels in recreational fresh waters should not exceed 20 PFU (plaque-forming units)/100 mL.

Summary

1. No limit on coliphages can be established at this time. Monitoring and epidemiological studies are required to determine the levels of coliphages in water and the health effects associated with swimming in water containing coliphages.

3.4 Pathogenic Organisms

3.4.1 *Pseudomonas aeruginosa*

Maximum Limits

No numerical limit is proposed; however, it is recommended that ***Pseudomonas aeruginosa*** be used as a parameter to assist in interpreting the results of sanitary and microbiological surveys.

Criteria

Description

Pseudomonas aeruginosa is a motile, Gram-negative, rod-shaped bacterium that produces oxidase, pyocyanin, and fluorescein. This organism typically requires minimal growth factors and can multiply in mineral media containing very low levels of organic material. ***Pseudomonas aeruginosa*** exhibits extensive biochemical versatility and resistance to anti-microbial agents. As well as being a pathogen to man and animals and causing a variety

of infections, including skin rashes and otitis externa, ***P. aeruginosa*** is active as a spoilage organism, attacking many common and exotic substrates (Hoadley 1977).

Pathogenicity

Pseudomonas aeruginosa has been identified as the causative agent in many infections. The organism seems capable of infecting plants, insects, birds, and mammals including humans. ***Pseudomonas aeruginosa*** infections are most frequent and dangerous in nurseries and among hospital patients with cancer, burns, and tracheotomies (Hoadley 1977); the organism is a common causative agent for nosocomial or hospital-acquired infections in patients that are in a debilitated state or have a compromised immune system, as well as where there is widespread use of antibiotics.

Pseudomonas aeruginosa is known to cause skin rashes (Kush and Hoadley 1980) and eye infections (Wilson and Ahearn 1977) and is the primary organism associated with external ear infections (otitis externa) (Cassisi *et al*. 1977). Ratnam *et al*. (1986) reported that ***P. aeruginosa*** in a hotel whirlpool had caused folliculitis in hotel guests that used the whirlpool. Cabelli *et al*. (1979) and McKee and Wolfe (1963) claimed that most reports of swimming-associated illness implicated non-enteric infections; ***P. aeruginosa*** has been implicated in non-enteric infections associated with bathing. Jones (1965) claimed that ***P. aeruginosa*** was the major etiological agent in otitis externa. Seyfried (1973) and Hoadley and Knight (1975) supported this finding. Seyfried isolated an identical serological strain from both a pool and an otitis externa-infected person who was using the pool. Hoadley and Knight, using telephone surveys, reported that the frequency of "earaches" among swimmers was 2.4 times higher than among non-swimmers; also, the chance of acquiring otitis externa, as reported by physicians, increased by five times for swimmers compared with the general population. Young and Armstrong (1972) reported the isolation of ***P. aeruginosa*** from skin and the outer ear. Craun (1976) reported the isolation of the same serotype of ***P. aeruginosa*** from a swimming pool and from the skin lesions of two affected bathers. McClausland and Cox (1975) reported that ***P. aeruginosa*** caused a rash outbreak among bathers at a pool; however, it was never proven whether the source of the organism was endogenous or exogenous (Cabelli *et al*. 1975).

Although Hoadley and Knight (1975) observed measurable health effects associated with swimming in sewage-polluted waters, Cabelli *et al*. (1979) were not able to show a good agreement between ***P. aeruginosa*** levels and the differential (swimmers minus non-swimmers) rate of gastrointestinal symptoms.

Pseudomonas aeruginosa in the water enters the ear canal and may lead to either colonization or infection. The actual process of infection appears to be, in part, related to the sensitivity of the individual to ***P. aeruginosa***

infections. ***Pseudomonas aeruginosa*** can be spread into the water if an infected ear is immersed while swimming (Seyfried *et al.* 1984). Data from a 4-year comprehensive Ontario study were used to develop a relationship between the concentration of ***P. aeruginosa*** in the bathing waters and the risk of ear infection (Ontario Ministry of the Environment 1984). In this study, 10 per cent of the sample bather population reported having ear infections in the first year of the study (1978), and 8 per cent reported ear infections in the second year. In both years, over 70 per cent of the cases were in children 14 years old or younger, and 90 per cent of these had reported previous ear infections.

Occurrence in the Aquatic Environment

Pseudomonas aeruginosa has been considered ubiquitous in U.S. waters. Cabelli *et al*. (1976) claimed that ***P. aeruginosa***, "because of its poor reliability as an indicator of fecal pollution, cannot be used as a basis of water standards for the prevention of enteric diseases during the recreational use of surface waters." They based this statement on their observations of geographical and seasonal density fluctuations and of many non-fecal reservoirs of ***P. aeruginosa***, as well as the ability of this organism to multiply in waters with low nutrient content.

Studies done on lakes in Ontario indicated that ***P. aeruginosa*** is most likely to be found in bathing areas impacted by high human activity. Both sewage and bathers are possible sources of this organism in recreational waters. In raw domestic sewage, concentrations of 10^5 to 10^6 ***P. aeruginosa***/100 mL are common, as slightly in excess of 10 per cent of the healthy adults in the United States are intestinal carriers of ***P. aeruginosa***. In addition, ***Pseudomonas*** levels in excess of 100 organisms/100 mL can be measured in waters receiving surface drainage from urban areas (Ontario Ministry of the Environment 1984).

Pseudomonas aeruginosa survives longer in waters than do coliforms (Lanyi *et al*. 1966). Drake (1966) suggested that ***P. aeruginosa*** levels from 1 to 10/100 mL could be expected in rivers with low but definite sources of contamination. Levels of ***P. aeruginosa*** in Ontario recreational waters range from 0/100 mL to more than 100/100 mL. The median level is typically less than 1/100 mL (Ontario Ministry of the Environment 1984).

Summary

1. ***Pseudomonas aeruginosa*** is typically isolated from fresh recreational waters in low numbers. The levels of ***P. aeruginosa*** in a bathing area are influenced by density of bathers, especially individuals that are infected with ***P. aeruginosa*** or are carriers.

2. Levels of ***P. aeruginosa*** are influenced by sewage or urban drainage sources.

3. ***Pseudomonas aeruginosa*** has been associated with the occurrence of otitis externa in bathers.

4. One Ontario study has demonstrated that when levels of ***P. aeruginosa*** exceed 10/100 mL in at least 25 per cent of the seasonal samples, otitis externa may be expected to occur.

3.4.2 *Staphylococcus aureus*

Maximum Limits

No limits are specified for ***Staphylococcus aureus***. Sampling for this pathogen should be carried out when there is epidemiological or other evidence of its presence in the water or in order to assess the hazards of excessive utilization of the water with possible person-to-person transfer of pathogens.

Criteria

Description

The ***Staphylococcus*** genus comprises Gram-positive, catalase-positive cocci that ferment glucose and grow aerobically and anaerobically. ***Staphylococcus aureus*** is identified by its ability to coagulate rabbit plasma in the presence of anti-coagulating agents. Other important diagnostic tests include the anaerobic fermentation of mannitol and the production of heat-stable nuclease (Evans 1977). Six biotypes exist that can be related to the animals that are their usual hosts (Dimitracopoulos *et al*. 1977).

Staphylococcus is not considered to be a natural inhabitant of environmental waters and is usually unable to grow there. It requires many organic nutrients to grow in water at about 20°C and grows little, if at all, below 10°C. It is, however, resistant to many environmental influences and can survive for relatively long periods of time. The presence of staphylococci in recreational waters is considered to be mainly due to discharges from the mouth, nose, and throat of swimmers, as well as from their skin surface. These organisms have been shown to be derived from bathers in studies on swimming pools (Mallman 1962; Favero *et al*. 1964; Paul 1972; Palmquist and Jankow 1973) and, to a lesser extent, natural waters (Oriz 1977). The 16th edition of *Standard Methods for the Examination of Water and Wastewater* (American Public Health Association 1989) identifies staphylococci as among the pathogens of human origin in natural bathing beaches. The extent and magnitude of the contribution of animals and land runoff to the total staphylococcal load have not been well documented.

Pathogenicity

Staphylococcus aureus is considered to be the major pathogen of this genus. It is responsible for most purulent infections, including boils and infected cuts and scratches (Evans 1977). Seyfried (1973) reported 1 case of ear infection associated with swimming in a contaminated pool. At present, there are no convincing data relating the frequency of illness to the degree of pollution of recreational waters.

In spite of the paucity of information that would allow an estimate of risk to be made for water of a particular quality, some maximum levels have been proposed. Limits suggested for safe bathing range from a "cocci" index of less than 15/100 mL (Seligmann 1951) to 100/100 mL (Favero *et al*. 1964). Seyfried (1987) found that total staphylococcus, fecal coliform, and fecal streptococci correlated best with swimming-associated morbidity. This was based on a 1983 study of freshwater beaches in southern Ontario and included an epidemiological component to the study. The study recommended that fecal coliforms, ***Escherichia coli***, ***Pseudomonas aeruginosa***, and total ***Staphylococcus*** be employed as recreational freshwater quality indicators.

Seyfried (1987) found geometric mean levels of total ***Staphylococcus*** of 142/100 mL for beaches in use and 96/100 mL for beaches that had been closed. The total staphylococci index has, in general, been favoured over a ***S. aureus*** index for purposes of health protection.

Occurrence in the Aquatic Environment

Limited data indicate that ***Staphylococcus*** is present at relatively low levels at Canadian beaches. One study revealed the presence of 10 to 40 staphylococci/100 mL in urban runoff (Environment Canada/Ontario Ministry of the Environment 1978). Studies performed by Seyfried (1973) demonstrated levels of between 30 and 90/100 mL on one of three Lake Erie beaches that were tested. Two of three Lake Ontario beaches were positive for ***S. aureus***. Two of three Ontario conservation areas were positive for ***staphylococci***, one species being coagulase-positive.

A more recent study (Seyfried 1980) of several Great Lakes beaches revealed the consistent presence of staphylococci. Of 12 Lake Ontario beaches studied, the modes ranged between 10 and 122/100 mL, with very few ***S. aureus*** being detected. In water from 10 Lake Erie beaches, modes ranged from 28 to 380/100 mL. In water from seven Lake Huron beaches, modes ranged from seven to 49/100 mL.

Summary

1. Staphylococcus aureus is known to be a major pathogen to man. It is responsible for boils, ear infections, and other purulent infections.

2. There appears to be a relationship between bather numbers and staphylococci levels in the water, but there does not appear to be a significant relationship between bather illness and concentration of ***S. aureus*** in the water.

3. For these reasons, no limit is being established for staphylococci at this time. Monitoring and epidemiological studies for this pathogen are recommended.

3.4.3 *Salmonella*

Maximum Limits

No limit is proposed for ***Salmonella*** concentrations in recreational waters. Instead, it is recommended that ***Salmonella*** be used as a parameter to assist in interpreting the results of sanitary and microbiological surveys. Because virtually all ***Salmonella*** species are pathogenic, a health hazard exists when ***Salmonella*** can be consistently isolated from a bathing area.

Criteria

Description

Salmonella species are members of the family Enterobacteriaceae and are Gram-negative, motile, straight, rod-shaped bacteria that ferment glucose but not lactose. Because of the antigenic complexity of the genus, organisms within this group are often identified by serotype as opposed to species. More than 1400 different ***Salmonella*** serotypes have been recognized (Bergey's Manual of Systematic Bacteriology 1986).

Pathogenicity

Salmonellosis is any disease in man or animal for which the causative agent is the ***Salmonella*** bacterium. The symptoms of this infection include acute gastroenteritis, enteric fever, and septicemia. Salmonellosis is a world-wide animal and human health problem that is compounded by the large number of serotypes and the widespread occurrence of the causative organism.

Dudley *et al*. (1976) developed a mathematical model to quantitatively estimate the recreational user health risk due to exposure to ***Salmonella*** in recreational waters. He concluded that more research was required to develop standardized quantitative ***Salmonella*** detection techniques for routine use with recreational waters before his model could be used effectively. In addition, Dutka and Bell (1973) suggested that all members of the genus are potentially pathogenic and should be considered a health hazard. There have been a number of reports of salmonellosis in communities where water supplies are

considered to meet coliform standards (Dutka and Bell 1973). Van Donsel *et al*. (1967) indicated that ***S. typhimurium*** was still able to infect humans after surviving 280 days in contaminated soil.

Occurrence in the Aquatic Environment

Cherry *et al*. (1972) found that 65 per cent of the samples taken from moderately polluted waters contained ***Salmonella***; 38 per cent of the samples taken from minimally polluted waters were positive for Salmonella; and 11 per cent of the samples from unpolluted streams were positive for ***Salmonella***. A relationship between the presence of ***Salmonella*** and the levels of fecal coliforms in water has also been noted (Geldreich *et al*. 1968; Smith and Twedt 1971; Smith *et al*. 1973; Hoadley *et al*. 1974). It appears that ***Salmonella*** can be consistently isolated from surface waters where the fecal coliform levels are above 200/100 mL.

Menon (1985) isolated five ***Salmonella*** serotypes (***S. infantis, S. typhimurium, S. saint-paul, S. tennessee,*** and ***S. heidelberg)*** from a tidal river in Nova Scotia that received both municipal and food processing plant effluents. The sources of ***Salmonella*** were traced to effluents from meat and poultry plants and several sewage treatment plants. Fecal coliform counts in the river also exceeded recommended recreational and shellfish harvesting guidelines. Palmateer (1980) also isolated ***Salmonella*** serotypes from surface water containing the effluent from poultry and meat packing plant operations. It was confirmed that four of the serotypes isolated (***S. bareilly, S. infantis, S. schwarzengrund***, and ***S. typhimurium***) had been identified as the causative agents of several outbreaks of salmonellosis in Ontario. ***Salmonella thompson*** and ***S. typhimurium*** were also frequently recovered from a storm sewer outfall during periods of rainfall (Qureshi and Dutka 1979). Bell et al. (1978) isolated ***Salmonella*** species from the North Saskatchewan River that were resistant to five commonly used antibiotics. Qureshi (1977) isolated four ***Salmonella*** serotypes from urban storm water samples collected in Toronto and Guelph. These four serotypes are among those often isolated from humans during recent years.

Several studies have demonstrated that ***S. typhimurium*** can survive longer than ***Escherichia coli*** in fresh and estuarine waters (Gosselin 1979; McCambridge and McMeekin 1981). In Lake Ontario and Hamilton Harbour, ***S. thompson*** survived for at least 20 days (Dutka and Kwan 1980). Such survival data demonstrate that occasional discharges of storm water or sewage may have cumulative effects, so that natural temperate waters could contain pathogens and therefore constitute a potential health hazard.

Salmonella have been isolated in higher numbers from bottom sediments than from the overlying waters (Van Donsel and Geldreich 1971; Hendricks 1971). This was linked with prolonged survival of ***Salmonella*** in the

sediments. Van Donsel and Geldreich (1971) suggested that *Salmonella* should be isolated from the sediments only when the fecal coliform levels reach 200/100 mL.

Summary

1. ***Salmonella*** organisms are pathogenic, and a health hazard exists if these organisms can be consistently isolated from a bathing area.

2. The methods for the isolation of ***Salmonella*** have not been standardized, and routine enumeration is not practical.

3. ***Salmonella*** can be considered as a support parameter to aid regulatory agencies in determining the health risk involved in using waters for recreation.

3.4.4 Shigella

Maximum Limits

No maximum limits are specified for ***Shigella*** in bathing areas. Sampling for these organisms in waters used for recreation should be carried out when there is epidemiological or other evidence of their presence in the water or in order to assess the hazards of excessive utilization of the water with possible person-to-person transfer of pathogens.

Criteria

Description

Members of the genus ***Shigella*** are typically Gram-negative, non-motile, lactose-negative organisms that do not produce hydrogen sulphide. Shigellae are enteric bacilli and have been the main etiological agents of bacillary dysentery (Lund 1978).

Pathogenicity

Shigellosis, the infection caused by ***Shigella***, can be transmitted through person-to-person contact, poor-quality drinking water, or contaminated food. The symptoms of shigellosis range from a mild transitory diarrhea to vomiting, abdominal pains, fever, and profuse bloody feces. On a few occasions, its transmission has been reported through bathing in contaminated waters. However, very few studies have been carried out to detect ***Shigella spp.*** in the aquatic environment, mainly because there are no standardized methods for detection or enumeration of ***Shigella*** in water. The methodologies that have been used to examine water for ***Shigella*** spp. are reported to have low sensitivity and can be considered qualitative only (Wang et al. 1966; Geldreich 1972).

McCabe and Craun (1975) reported that ***Shigella*** spp., especially ***S. sonnei***, were the most commonly identified pathogens causing waterborne disease outbreaks during 1971 and 1972 in the United States and Canada. These outbreaks were associated with polluted drinking water. A severe outbreak of shigellosis associated with swimming in the section of the Mississippi River below Dubuque, Iowa, has been documented (Anonymous 1974; Rosenberg et al. 1976). As a result of the follow-up on this epidemic, Rosenberg et al. (1976) remarked that shigellosis – "a disease that can be caused by ingestion of only 10 to 100 organisms – can be contracted by swimming in polluted waters." This was the first epidemic of shigellosis that could be linked with bathing. Cabelli (1979), in reconstructing the epidemiological evidence from the shigellosis outbreak at Dubuque, concluded that, although the fecal coliform levels exceeded 200/100 mL for a period of several years, a critical level of ***Shigella*** carriers or ill individuals discharging into the environment had to occur before there would be an outbreak of shigellosis.

More recently, Makintubee et al. (1987) reported the occurrence of at least 62 cases (38 primary or co-primary and 24 secondary) of shigellosis **(*S. sonnei*)** associated with swimming in a natural reservoir. Although excessive concentrations of fecal indicator bacteria were present in the water, ***Shigella*** was not recovered.

Occurrence in the Aquatic Environment

Shigella can be isolated from the feces of warm-blooded animals and sewage. Lund (1978) suggested that the bacterium tends to survive for a relatively short period of time in the environment. Hendricks (1972), using autoclaved river water collected downstream from a sewage outfall, found that ***Shigella*** grows best at 30°C and reproduces at a rate 2 to 3 times slower than coliforms. He observed little or no growth at temperatures of 5 to 20°C. Wang *et al*. (1966) found that the organism survives longer in wastewater at 15°C than at 25°C. Andre *et al*. (1967) observed that ***Shigella*** persisted in untreated farm pond water for 12 days. Other survival characteristics, reviewed by Geldreich (1972), include an appreciable reduction in survival time by the use of aeration or by the reduction of the pH below 7.6.

Summary

1. ***Shigella*** organisms are pathogenic, and a health hazard exists if these organisms can be consistently isolated from a bathing area.

2. The methods for the isolation of ***Shigella*** have not been standardized, and routine enumeration is not practical.

3. ***Shigella*** can be considered as a support parameter to aid regulatory agencies in determining the health risk involved in using waters for recreation.

3.4.5 *Aeromonas*

Maximum Limits

No maximum limit has been proposed for ***Aeromonas*** in bathing areas. It is recommended that sampling for ***Aeromonas*** in waters used for recreation be considered only for epidemiological investigations.

Criteria

Description

Organisms belonging to the genus ***Aeromonas*** are facultatively anaerobic, Gram-negative rods possessing polar flagella. The temperature range for bacillus growth is between 0 and 41°C (Ewing *et al*. 1961). ***Aeromonas*** species are widely distributed in stagnant and flowing fresh waters, in sludge, and in sewage (Hazen *et al*. 1978). *Bergey's Manual of Systematic Bacteriology* (1986) and Popoff *et al*. (1981) recognize three species of ***Aeromonas*** found in clinical specimens: ***A. hydrophila***, ***A. caviae***, and ***A. sobria***. However, ***Aeromonas*** can be divided into 9 to 12 DNA hybridization groups, and the taxonomy has yet to be clarified.

Pathogenicity

Human infections with ***Aeromonas*** occur predominantly during the period from May to November, probably because of the aquatic origin of the bacteria (Davis *et al*. 1978). Infections caused by the species have been divided into four categories (von Graevenitz 1985):

- cellulitis or wound infection related to exposure to water
- acute diarrheal disease of short duration
- septicemia, mostly in association with hepatic biliary or pancreatic disease
- other infections, such as soft-tissue infections, urinary tract infections, meningitis, peritonitis, otitis, and endocarditis, particularly in immuno-compromised people.

Aeromonas species cause an acute, self-limiting diarrheal illness in humans; this is supported by the finding of various exotoxins. One of them, enterotoxin, has been detected by infant mouse tests (Turnbull *et al*. 1984); it is a heat- and acid-labile molecule and causes cell lysis in tissue culture.

Carrier rates of 2 to 3 per cent for ***Aeromonas*** in feces have been observed in England, the United States, and Australia, with no associated gastrointestinal disease.

Occurrence in the Aquatic Environment

Aeromonas species can be isolated from the feces of warm-blooded animals, sewage, fresh waters, and salt waters that interface with fresh water. They have been found at pH values of 5.2 to 9.8 and at temperatures between 4 and 45°C. They are not considered halophilic, because their salt tolerance ranges from 0 to 4 per cent. They have also been isolated from soil and foodstuffs (Ewing *et al*. 1961).

Hanson *et al*. (1977) reported a case, in a previously healthy young man, of severe cellulitis that developed from a laceration that occurred during swimming. ***Aeromonas hydrophila*** was recovered in large numbers from the wound and the freshwater lake where the injury occurred.

Joseph *et al*. (1979) reported a case of primary soft-tissue infection caused by two species of ***Aeromonas*** (***A. hydrophila*** and ***A. sobria***) in a student diver conducting scuba operations in a freshwater lake.

Summary

1. As water appears to be the natural habitat of ***Aeromonas***, this organism should not be used as an indicator of fecal pollution or as a sanitary indicator for bathing areas.

2. ***Aeromonas*** may be pathogenic, but sampling of recreational water should be considered for epidemiological investigations only.

3.4.6 *Campylobacter jejuni*

Maximum Limits

No limits are specified for ***Campylobacter jejuni*** in recreational waters. Sampling for this pathogen should be conducted when there is epidemiological or other evidence of its presence in the water. Sampling should also be considered to assess the hazards of excessive utilization of the water with possible person-to-person transfer of pathogens.

Criteria

Description

Campylobacter jejuni (***C. fetus*** subsp. ***jejuni***) and ***C. coli*** are now recognized as important enteric pathogens, often responsible for diarrhea in humans (Benenson 1985). Campylobacteriosis, also known as campylobacter enteritis or gastroenteritis, is the name of the illness caused by ***C. jejuni***.

Campylobacter is thought to be responsible for a greater proportion of enteritis than either ***Salmonella*** or ***Shigella***. Diagnosis is based on isolation of the organisms from feces using selective media, reduced oxygen tension, and an incubation temperature of 43°C. Visualization of motile curved, spiral, or S-shaped rods similar to those of ***Vibrio cholerae*** by phase- contrast or dark-field microscopy of feces can provide rapid presumptive evidence for campylobacter enteritis.

Source and Pathogenicity

Campylobacter jejuni has been isolated from water, mud, livestock, and dogs and cats. Birds, in particular, have been a well-documented reservoir, as small amounts of bird droppings in water can release many ***Campylobacter*** organisms (Benenson 1985; Sacks *et al*. 1986). Modes of transmission to humans include contact with animals, handling raw chicken, person-to-person contact, and consumption of contaminated food, raw milk, and water. The infective dose of ***Campylobacter*** in water is unknown. Comparatively, as few as 500 organisms in milk can be infectious (Robinson 1981).

Waterborne outbreaks of campylobacter enteritis have been associated with municipal water systems in North America (Sacks *et al*. 1986; Borczyk *et al*. 1987) and various European countries (Bolton *et al*. 1987).

Surface waters can contain ***Campylobacter*** spp., but their survival is temperature dependent. At 4°C, they can survive from 11 days to 4 weeks, and at 25°C, from 2 to 4 days (Mentzing 1981). In another study, ***C. jejuni*** inoculated into autoclaved mountain stream water remained viable for 33 days at 4°C. At 25°C, the organisms became non-viable within 4 days (Blaser *et al*. 1980). In an investigation of a waterborne outbreak, it was thought that, despite warm ambient temperatures, an accumulation of algae and scum in an open settling tank could have contributed to the organisms' extended viability (Sacks *et al*. 1986).

Occurrence in the Aquatic Environment

In a study by Taylor *et al*. (1982), ***C. jejuni*** was responsible for sporadic summertime diarrheal disease with the consumption of untreated surface waters in wilderness areas. Subsequently, ***C. jejuni*** was isolated from animals in the area. Another study (Taylor *et al*. 1983) involved a campylobacter enteritis outbreak that was epidemiologically linked to the community raw water system.

The survival of ***C. jejuni*** has been tested in drinking water, river water, and sewage; results indicate that its survival is restricted to a few days (Pickert and Botzenhart 1985). The concentration of oxygen and nutrients in the samples did not appear to affect survival, whereas temperature was demonstrated to be the most significant variable.

Campylobacter appears in the environment more commonly during summer. Studies of birds indicate an increase in ***Campylobacter*** carriage in the summer compared with the winter; this coincides with waterborne campylobacter outbreaks, which tend to occur only in summer and fall (Sacks *et al*. 1986).

In a study by Bolton *et al*. (1987) involving a river system that traversed rural and urban areas, the lowest frequency of isolation and the lowest counts obtained (10 ***Campylobacter***/100 mL) were associated with samples collected from rural sites and areas with faster river flow. The greatest frequency of isolation and the highest counts (20 to 30 ***Campylobacter***/100 mL) were associated with samples collected adjacent to, or downstream from, sewage works. A seasonal trend was demonstrated: the highest counts and most isolations were obtained in late autumn and winter, and the lowest counts and fewest isolations were obtained in the spring and summer. Heavy rainfall and the subsequent runoff from adjacent farmland were found to contribute to increased counts of ***Campylobacter*** in the river system.

Several studies have isolated ***Campylobacter*** spp. from surface water in association with ***Escherichia coli*** (Carter *et al*. 1987). These studies indicated that ***Campylobacter*** spp. were detected only in the presence of ***E. coli***.

Summary

1. Recent improvements in environmental sampling techniques and means to distinguish strains have made environmental studies more feasible (Taylor *et al*. 1982).
2. Water is potentially an important reservoir of the thermophilic ***Campylobacter*** and is an established vehicle for the transmission of ***Campylobacter*** to man and domestic animals.
3. The use of a standard indicator of fecal pollution would be helpful in determining potential health hazards related to ***Campylobacter*** spp. as well as other bacterial intestinal pathogens.
4. All data to date link campylobacter enteritis to water consumption and not recreational contact. For this reason, no limit is being established at this time. Monitoring may be conducted where warranted on the basis of epidemiological or other data. A health hazard exists if ***Campylobacter*** can be consistently isolated from a bathing area.

3.4.7 *Legionella*

Maximum Limits

No limits are specified for ***Legionella*** spp. in recreational waters.

Criteria

Description

At least 22 species of ***Legionella*** have been identified. These are Gram-negative, non-spore-forming, aerobic bacilli, which grow at 35 to 37°C in special media (e.g., buffered charcoal yeast extract).

Pathogenicity

Since the outbreak of Legionnaires' Disease in Philadelphia in 1976, Legionellaceae have been recognized as an important cause of respiratory illnesses, causing legionellosis and Pontiac fever. Legionellosis is a multiple-system disease that can be fatal in immuno-compromised persons. Pontiac fever is a self-limited, flu-like illness, mainly affecting immuno-competent persons. The mode of infection is by inhalation of infected aerosols. Almost all outbreaks of endemic and epidemic, community-acquired, and nosocomial infections have been related to indoor plumbing and air conditioning systems (Benenson 1985).

Occurrence in the Aquatic Environment

All ***Legionella*** are aquatic bacteria, found mainly in water and mud (Edelstein 1985). Five reports can be found on legionellosis associated with recreational waters: two outbreaks of Pontiac fever associated with the use of whirlpool (Mangione *et al*. 1982; Goldberg *et al*. 1989); one case of a wound infection in a hydrotherapy tank (Brabender *et al*. 1983); one case associated with near drowning (Sekla *et al*. 1982); and 1 case following immersion in a river (Farrant *et al*. 1988).

Summary

1. ***Legionella*** are natural aquatic bacteria and cannot be used as an indicator of fecal pollution or as a sanitary indicator of bathing areas.

2. There are very few data available on legionellosis associated with recreational waters.

3. Routine testing of recreational waters for ***Legionella*** is not recommended.

3.4.8 Viruses

Maximum Limits

In Canada, no limits are specified for viruses in recreational waters. Sampling for viruses should be conducted when there is epidemiological or other evidence of their presence in the water or in order to assess the hazards of excessive utilization of the water with possible person-to-person transfer of pathogens.

Criteria

Description

Viruses are submicroscopic microorganisms that are unable to replicate outside their normal host. Among the more than 100 enteric viruses that are excreted in feces and could possibly be found in recreational waters, some can remain infective for several months in water and underlying sediments (Sattar 1981), including enteroviruses (polio, coxsackie, echo, and hepatitis A viruses), adenoviruses, rotaviruses, reoviruses, Norwalk viruses, caliciviruses, astroviruses, and coronaviruses. The infectious dose of some human enteric viruses can be as low as one tissue culture unit (Plotkin and Katz 1967; Ward and Akin 1984; Ward *et al*. 1986) and at least one order of magnitude lower than that of bacteria (Blaser and Newman 1982).

Pathogenicity

The diseases produced by the enteric viruses range from unapparent to severe. Enteric viruses can cause gastroenteritis, hepatitis A and hepatitis E (non-A, non-B hepatitis), fever, respiratory ailments, eye infections, central nervous system infections, poliomyelitis, etc. (Sattar 1978b; Gerba *et al*. 1985; Gust and Purcell 1987).

The lack of a central reporting of such infections coupled with rapid person-to-person transmissions have made it extremely difficult to determine the existence of waterborne viral diseases (Pipes 1978). The risk associated with bathing in virus-contaminated waters has been discussed by Payment (1984) and Craun (1986).

Epidemiology

Reports of recreational waterborne viral infections are rare (Paffenbarger *et al*. 1959; McLean 1965). Transmission of adenoviruses from swimming pools has been reported (Foy *et al*. 1968; Caldwell *et al*. 1974) and usually causes eye infections. Transmission of enteric viruses from lake waters has also been documented for coxsackievirus B3 (Hawley *et al*. 1973), coxsackie A16 (Denis *et al*. 1974), hepatitis A virus (Bryan *et al*. 1974), Norwalk virus (Koopman *et al*. 1982; Kappus *et al*. 1982), and, finally, echovirus (Walter-Offenhauser and Horn 1974).

Occurrence in the Aquatic Environment

A large number of enteric viruses may be found in the aquatic environment (Department of National Health and Welfare 1977; Bitton *et al*. 1985) as a result of pollution by animal wastes, municipal sewage, and other sources of human waste. In contrast to fecal coliform bacteria, which are present in all feces, viruses are excreted only from infected individuals (U.S. Environmental Protection Agency 1978a), who often are symptomless carriers in the

under-15-years age group (Ramoz-Alverez and Sabin 1956). It is established that virus levels in water vary markedly on an hourly, daily, and seasonal basis (Berg and Metcalf 1978), whereas total and fecal coliform levels are generally more stable. For these reasons, the ratio of viruses to total coliforms of 1:65 000 does not always hold true (Scarpino 1975). Berg and Metcalf (1978) also demonstrated that fecal coliforms and enteric viruses do not occur in any constant ratio. Viruses have been found in the absence of detectable fecal coliforms (Berg 1978), and significant levels of viruses have been found in waters well within the bacteriological limits recommended for recreational waters (Goyal *et al*. 1978). Resistance to chlorine inactivation has been reported by Bates *et al*. (1977).

Several studies have investigated virus levels in sewage (Sattar and Westwood 1977, 1978; Subrahmanyan *et al*. 1979; Sekla *et al*. 1980), rivers (Subrahmanyan 1977; Sattar and Westwood 1977; Sekla *et al*. 1980; Payment *et al*. 1988), and lakes (McLean 1965; Subrahmanyan 1977). An extensive investigation of water in the Ottawa River (Sattar 1978a) revealed that, at one site, seven out of 16 samples were positive for virus; at a recreational beach, 11 of 20 samples were positive, with a range of 0.6 to 16.8 infective units/ 10 L. The sources of contamination were raw sewage inputs, as well as treated, chlorinated sewage effluents (Sattar and Westwood 1978).

Payment (1977) suggested a limit of one tissue culture infectious dose/40 L, whereas Sattar (1978a) proposed one infective unit/10 L when a 100-L sample is examined. In Israel, Shuval (1975) recommended that tentative limits be established. In the United States, Melnick (1976) suggested a limit of one detectable infectious virus unit/10 U.S. gallons (37.9 L) of recreational water. The State of Arizona adopted a surface water standard of not greater than one enteric virus/40-L sample (Gerba 1988).

The investigation of viral waterborne outbreaks requires special laboratory facilities. Occasional monitoring for viruses can be carried out to determine their distribution and relationship to disease incidence, but routine tests of recreational waters is not recommended.

Summary

1. Viruses are known to be pathogenic in low numbers. As few as one infective tissue culture unit can cause disease when ingested. Concentration and enumeration steps are too detailed to make routine monitoring practical.

2. Very few data are available on current virus levels in recreational waters.

3. There is no correlation between viral and bacterial counts in recreational waters.

4. No limit can be established for viruses at this time. Monitoring and epidemiological studies are needed to determine the levels of viruses in water and the health effects of swimming in virus-contaminated waters.

3.4.9 Protozoa

Maximum Limits

No limits are specified for pathogenic protozoa in recreational waters.

Criteria

Description, Pathogenicity, and Occurrence

A large number of pathogenic parasites can occur in the aquatic environment. Of potential importance in Canada are 4 protozoa (***Giardia, Cryptosporidium, Naegleria***, and ***Entamoeba histolytica***) and 1 helminth (***Schistosoma***).

Giardia is currently the most common pathogenic intestinal protozoan in Canada and the United States. The ingestion of a few (10 to 100) viable cysts can cause a diarrheal illness (giardiasis). Transmission can be from person to person or via food or water. Waterborne giardiasis has received much attention lately, as outbreaks have been traced to pristine waters, as well as to sewage-contaminated potable waters (Lin 1985). ***Giardia*** are more resistant to chlorination than indicator organisms, pathogenic bacteria, and viruses (Sobsey 1989). Thus, fecal coliform counts cannot be used as indicators of protozoal contamination of recreational waters. Giardiasis associated with toddler swim classes has been discussed by Harter *et al*. (1984), and transmission in a swimming pool was reported by Porter *et al*. (1988). A water slide has also been incriminated in an outbreak of giardiasis (Greensmith *et al*. 1988).

Cryptosporidium, a newly recognized pathogenic protozoan, may be as important as ***Giardia*** (Rose 1988). The ingestion of low levels of viable oocysts can also result in a diarrheal illness known as cryptosporidiosis. Like ***Giardia, Cryptosporidium*** can also be transmitted from person to person or via food or water. The illness, which may be fatal in immuno-compromised patients, has occurred in Canada (Mann *et al*. 1986). No outbreaks have been linked to recreational waters, but major outbreaks in the United States and the United Kingdom have resulted from the ingestion of inadequately treated drinking water (D'Antonio *et al*. 1985; Rose 1988). Although information is only preliminary, it appears that ***Cryptosporidium*** is even more resistant to disinfection than ***Giardia*** (Sobsey 1989). Oocysts have been recovered in surface waters in British Columbia (Isaac-Renton *et al*. 1987) and the United States (J. Rose and C. Gerba, unpublished report).

Naegleria fowleri and other members of the freshwater amoeba group have caused more than 100 cases of an often-fatal primary amoebic meningo-encephalitis (PAM). PAM has been reported from the United States, South America, Europe, Australia, and New Zealand in persons swimming in fresh waters, lakes, and ponds, and even in indoor pools filled with chlorinated heated river water, as these amoeba are very resistant to chlorine and thrive in warm waters (Griffin 1977; Hallenbeck and Brenniman 1989). ***Naegleria fowleri*** has been recovered from surface waters in Canada (Seyfried *et al*. 1984).

Entamoeba histolytica is found throughout the world, affecting 10 per cent of the world population. It can cause amoebic dysentery and liver abscesses. The cysts can survive well in water. The effect of temperature has been studied by Jones and Newton (1950). The best-studied outbreak of waterborne amoebiasis was related to the drinking of contaminated water in the United States, as reported by Le Maistre *et al*. (1956).

Schistosoma spp. are digenetic trematodes (also known as flukes or flatworms). The larvae (cercariae), released in water by infected aquatic snails, must enter the skin of a susceptible host to complete their life cycle. The species responsible for human schistosomiasis, a disease of considerable morbidity affecting 200 million people in Africa, South America, the Middle East, and parts of Asia, does not occur in North America.

Avian schistosomes can be found in Canada. The cercariae of bird and rodent schistosomes may penetrate the human skin, causing a dermatitis known as swimmers' itch. This itch occurs in bathers using recreational lakes in many parts of the world, including North America, as well as on certain coastal seawater beaches. These schistosomes do not mature in humans (Levy and Folstad 1969; Benenson 1985) but die just beneath the epidermis. Subsequent exposure to cercariae stimulates an allergic response. At the time of cercarial penetration, a prickling sensation is noted (Hoeffler 1977).

Prevention of swimmers' itch can be assured only by complete avoidance of aquatic sports in areas where the disease has been a problem. An acceptable alternative might be the elimination of the molluscan hosts, using approved molluscicides. Prevention of secondary infection by bacteria can be achieved by appropriate hygienic measures. Attempts to prevent the dermatitis during or after cercarial penetration are largely ineffective. Time-honoured measures such as rough towelling and alcohol rubdowns are without significant merit. Clothing and chemical repellents are of limited value. Treatment of the dermatitis is directed towards the relief of symptoms with the usual doses of antihistaminic and antipruritic medications.

Summary

1. Routine monitoring of waters for protozoa is not recommended. However, provincial laboratories should be able to participate in the investigation of documented waterborne outbreaks.

3.4.10 Toxic phytoplankton

Maximum Limits

No limits are specified for toxic phytoplankton in recreational waters. However, water containing a blue-green or turquoise scum is indicative of an algal bloom. Such waters should be avoided because of reduced clarity and the possible presence of algal toxins.

Criteria

Description

Phytoplankton, which are microscopic floating plants, can become a hazard and a nuisance in recreational waters, especially when they concentrate at the water surface in "blooms." This can be a natural phenomenon, but it is often caused by cultural eutrophication. The presence of certain species in a freshwater community is often a sensitive indicator of recreational water quality. Nutrient enrichment in bodies of water affects the density and diversity of the fauna and flora.

The algae of concern in lakes and ponds are usually blue-green algae. These algae are unusual because they share some typical features of both algae and bacteria. In some species, the algae are tiny single cells that cannot be seen with the naked eye; in most species found in Canadian lakes, the cells are grouped in colonies. The colonies may form strings, flakes, or globules and can reach a size of several millimetres. To the naked eye, they may look like fine grass clippings in the water or a homogeneous soupy mass.

In a typical summer, a lake water sample usually contains 20 or more blue-green algal species, along with dozens of other species of algae. The blue-green algae may form massive blooms under certain conditions during summer (Reynolds and Walsby 1975).

Blue-green algal cells contain small gas bubbles (vacuoles) that allow them to control their buoyancy. Usually the algae are well distributed in the zone where nutrients and light are in the optimum ranges. Blooms are not the result of a sudden surge in algal growth rate but occur when their buoyancy regulation is disrupted. Reynolds and Walsby (1975) stated that bloom formation requires that a substantial population of the algae already exists, that they have excess buoyancy, and that the water is calm enough to allow them to float up to the surface. The algae may develop excess buoyancy when turbulence sends them too deep, during the hours of darkness, when the concentration of

carbon dioxide in the water becomes limiting, when the algal population is at the end of its growth cycle and is aging, or any combination of these factors. Blooms often occur in late August and in September when the algal population is aging and the hours of darkness are growing longer. Dillenberg and Dehnel (1960) and Senior (1960) reported a striking example of a sequence of meteorological events that illustrated these factors in a Saskatchewan lake.

Toxicity

Toxic algae are found in all aquatic environments and have been responsible for the death or illness of livestock, waterfowl, fish, and humans (Carmichael *et al*. 1985). The most important taxonomic phyla are Pyrrhophyta (dinoflagellates), Chrysophyta, and Cyanophyta (blue-green algae). Well-known instances of marine algal toxicity are associated with "red tides," which cause fish kills and poison edible shellfish and other bottom fauna (Tangen 1977; Cross and Southgate 1980). On the west coast of North America, clams and mussels become poisonous by ingesting the dinoflagellate ***Gonyaulax catenella*** and accumulating saxitoxin in the digestive tract. On the east coast, the associated poisoning is caused by ***Gonyaulax tamarensis***, with a mixture of saxitoxin and three related toxins.

Blue-green algae have been known to cause animal toxicity in lakes, ponds, and dugouts for over 100 years. The three blue-green algae most often identified as the cause of poisoning are ***Anabaena flos-aquae, Aphanizomenon flos-aquae***, and ***Microcystis aeruginosa*** (McLeod and Bondar l952; Senior 1960; Aziz 1974; Moore 1977, Richard *et al*. 1983).

A number of toxins produced by freshwater blue-green algae have now been identified. Two neurotoxins and two hepatotoxins have been associated with ***Anabaena flos-aquae***. Anatoxin-a is a potent postsynaptic, depolarizing, neuro-muscular blocking agent that causes death by respiratory arrest within minutes to a few hours depending on species, dosage and prior food consumption. The purified toxin intraperitoneal LD50 for mice is about 200 mg/kg body weight, with survival time of 4 to 7 minutes (Carmichael 1988). Another neurotoxin produced by ***Anabaena flos-aquae*** is anatoxin-a(s), an anticholinesterase, which also causes viscous salivation, lachrymation, urinary incontinence and defecation prior to death by respiratory arrest. Anatoxin-a(s) is about four times more toxic than anatoxin-a. The two hepatotoxins produced by ***Anabaena flos-aquae*** appear to be heptapeptides similar to two analogues produced by Microcystis aeruginosa (Carmichael 1988).

The toxins produced by ***Aphanizomenon flos-aquae*** consist mainly of two neurotoxic alkaloids that strongly resemble saxitoxin and neosaxitoxin (Sasner *et al*. 1984) These are more generally known as the paralytic shellfish poison associated with "red tides", and received their names from the Alaska

butter clam ***Saxidomus giganteus*** (Schuett and Rapoport 1962). Three other neurotoxins have been detected that are labile and not similar to any known paralytic shellfish poisons (Carmichael 1988).

The hepatotoxins produced by ***Microcystis aeruginosa*** attack the liver, causing severe necrosis and hemorrhaging. Death from hypovolemic shock caused by interstitial hemorrhage into the liver may occur within 1 to 3 hours in acutely dosed and sensitive animals. In other animals, death may occur as long as 36 hours later, with the animals displaying symptoms of incoordination, muscular weakness and fibrillation, unsteady gait, recumbency, labouredrespiration, salivation, lacrimation, diarrhea, and in those suffering lingering deaths, icterus, photodermatitis, and loss of condition (Soll and Williams 1985). A diarrhea toxin has also been isolated from ***M. aeruginosa***, and this may be a cause of gastroenteritis when no other known etiological agent can be identified (Aziz 1974).

Toxicity due to blue-green algae can occur only if there is a bloom dominated by the toxic strains of the bloom species. Toxic strains and non-toxic strains of a bloom species may occur at the same time in a lake; as a result, some parts of the lake could become toxic, while others could remain safe. Toxicity in a lake is normally transient, lasting only as long as the bloom or signs of the bloom persist.

The reason why toxic strains suddenly become more dominant than the non-toxic strains is not known (Carmichael *et al*. 1985). Consequently, toxicity due to blue-green algae is even less predictable than the blue-green blooms themselves. Lakes that have never had a problem can suddenly become toxic. Conversely, lakes that have shown toxicity in the past may never show it again. Routine monitoring of lakes where blooms have occurred is the best approach to identifying a problem.

When a lake becomes toxic as a result of a blue-green bloom, the only sign of a problem may be dead fish, waterfowl or other wildlife along the shoreline. Occasionally, domestic animals such as cattle or dogs may be poisoned if they have no other source of drinking water.

In Canada, animal poisonings have been reported for Alberta, Saskatchewan, Manitoba, and Ontario (Stewart *et al*. 1950; O'Donoghue and Wilton 1951; McLeod and Bondar 1952; Neil 1957; Senior 1960; Carmichael and Gorham 1978). Most algal poisonings in western Canada are associated with blooms of ***Anabaena flos-aquae***.

There is some concern, particularly in the Prairie provinces, about possible poisoning of swimmers from blooms of blue-green algae. Humans are just as susceptible to blue-green toxins as animals, but it is unlikely that people would voluntarily drink much lake water during a bloom because of the objectionable appearance and odour of the water. However, people may suffer acute discomfort after accidentally ingesting or contacting blue-green algae or water containing toxins. Symptoms may include fever, headache, dizziness,

stomach cramps, vomiting, diarrhea, skin and eye irritations, sore throat and swollen lips. Symptoms seldom persist for more than two to eight days. Children may be more intensely affected because they spend more time in the water than adults and are more likely to have accidentally swallowed contaminated water. In addition, they may have lower tolerances to the toxins than adults.

Dillenberg and Dehnel (1960) recorded the poisoning of a man who fell into a Saskatchewan lake containing a dense bloom of blue-green algae. Nausea, diarrhea, headache, cramps, and high temperature occurred. ***Microcystis*** and ***Anabaena***, but no other pathogens, were found in his feces. Similar illness occurred among children who swam in another Saskatchewan lake also containing a bloom of ***Microcystis*** and Anabaena. Schwimmer and Schwimmer (1968) reported a near-fatal case of a boy who developed fever, laboured breathing, pneumonitis, generalized pains, and coma after swimming in water containing ***Microcystis aeruginosa***. They also reported the poisoning of another boy who fell into a lake, accidentally swallowed water containing ***Aphanizomenon***, and developed symptoms similar to those experienced by the first case described above. Schwimmer and Schwimmer (1968) cited reports of allergic responses to ***Microcystis***, ***Anabaena***, and ***Aphanizomenon***, including papulo-vesicular eruption and acute allergic conjunctivitis, specifically with ***Anabaena***. Billings (1981) also reported possible contact irritation associated with ***Anabaena***. The recorded instances of illnesses in humans associated with blooms of toxic algae are few, but such illnesses may be more common than is thought because of lack of diagnosis.

Recreational lake users should take particular care where the water contains algae with the distinctive blue-green or turquoise colour and should treat any intense bloom with suspicion. Humans should not drink water from bloom-infested areas of lakes and reservoirs, nor should they swim or wade in water containing concentrated blue-green algal material. People should also take care to provide alternative water sources for domestic animals and pets. Bodies of water that have the green colouration of normal plants, like grass or the pond-dwelling plant duckweed, are most unlikely to be toxic regardless of the thickness of the scum.

Controlling Phytoplankton

Attempts at controlling blue-green algae by chemical treatment may be detrimental to lake water use and lake ecology. Hanson and Stefan (1984) reported short-term effects, such as the death of some algae, depletion of dissolved oxygen resulting in occasional fish kills, and the recycling of phosphorus from the lake bed, which fosters conditions that allow regrowth of the algae within 7 to 21 days. Further, because the toxins are released on

the death of the algae, water that has been treated with copper or other chemicals to kill the algae may be especially dangerous for the first 24 hours after treatment.

The detrimental long-term effects of algal control are the accumulation of copper in the lake bed sediments, increasing resistance of the phytoplankton to copper, increasing domination of blue-green algae species, particularly ***Aphanizomenon***, over green algae species and rough fish over game fish, the disappearance of macrophytes, and the loss of benthic macroinvertebrates. When the water overlying the bottom sediments becomes anoxic, the copper will re-enter the limnetic zone (Prepas and Murphy 1988). The use of copper suphate may be forbidden in some jurisdictions.

There are potential alternatives to the use of copper sulphate and aquatic herbicides such as Reglone A (Diquat). A finding that total phosphorus may be the best predictor of relative biomass of blue green algae (Trimbee and Prepas 1987) has lead to lake treatment based on decreasing the total phosphorus concentration in the water. Babin *et al*. (1989) report that the application of lime (calcium hydroxide) slurry has dramatically decreased both soluble reactive phosphorus and chlorophyll a (used to estimate algal biomass) in storm water retention ponds and in hypereutrophic natural lakes. Stable stratification of lake water also tends to favour the dominance of blue-green algae because of their buoyant properties. Experimental mixing has been successful in suppressing their growth in reservoirs previously subject to frequent blooms (Reynolds and Walsby 1975).

Summary

1. No limits are recommended for toxic phytoplankton, but swimming in waters containing blue-green algal blooms should be avoided.

2. Bather poisonings have occurred after immersion in lakes and ponds containing dense blooms of blue-green algae.

3. Sampling recreational waters for toxic phytoplankton should be considered only for epidemiological investigations.

4. Nuisance Organisms

Maximum Limits

The bathing area should be as free as possible from nuisance organisms that could affect swimmers. Nuisance is defined as something that can cause harm or is annoying, unpleasant, or obnoxious (Webster's Third New International Dictionary 1986).

It is impossible to have natural areas "free" from nuisance organisms, so no limits can be quantified. Only the possible hazards caused by such organisms and the environmental situations that could promote their presence are discussed.

Recreational areas should not be developed if there is an excessive growth of aquatic plants where entanglement could occur, thus causing a hazard to water-related recreational activities, unless control measures are taken to remove the plants from areas used for swimming.

Criteria

Description

Two principal types of biological factors influence the recreational value of surface waters: those that endanger the health or physical comfort of people and animals, and those that render water aesthetically objectionable or unusable as a result of excessive nutrient enrichment or the presence of unsightly substances. The former include vector and nuisance organisms; the latter include aquatic growth of microscopic and macroscopic plants.

Vector organisms

Massive emergences of non-biting midges, phantom midges, caddisflies, mayflies, etc. cause serious nuisances in shoreline communities and impede recreational pursuits (National Academy of Sciences 1973). In addition to the physical annoyance of their presence, biting insects such as mosquitoes and black-flies can inflict serious irritation from their biting attacks.

Human respiratory allergic reactions such as hay fever have been reported to be due to inhalation of caddisflies, mayflies, and midges, or their parts, when large insect swarms are present (Henson 1966).

Abrupt changes in water quality, especially if accompanied by organic loading, may precipitate high midge production. A sudden decline in oxygen supply in organically enriched bodies of water can disrupt established faunal communities and favour the less sensitive species of midge larvae and other

tolerant organisms that can withstand low dissolved oxygen. Part of this change is caused by the increase in food from organic waste. Upon emergence as adults, these midges can cause a nuisance with their dense swarms.

In the marine situation, nuisance or hazardous conditions can exist at bathing beaches because of the presence of organisms such as jellyfish and some species of sea urchins.

Aquatic vascular plants and algae

Aquatic vascular plants (macrophytes) affect water quality, other aquatic organisms, and the uses that are made of the water. It is difficult to estimate the magnitude of the adverse effects of aquatic macrophytes in terms of loss of recreational opportunities or the degree of interference with recreational pursuits. For example, extensive growths of aquatic macrophytes interfere with boating of all kinds, but the extent of interference depends, among other things, on the growth form of the plants, their density, the fraction of the water body affected, and the purposes, attitudes, and tolerance of the boaters.

Dense growths of aquatic macrophytes are generally objectionable to the swimmer, diver, water-skier, and scuba enthusiast. Plants obstruct the view of the bottom and underwater hazards, and swimmers can become entangled in the fronds. Water-skiers' preparations in shallow water are hampered by dense growths of plants, and fear of falling into such growths while skiing detracts from enjoyment of the sport.

Recreational activities such as boating and fishing are less appealing and may even be almost impossible if aquatic plants are very dense. Rafts of free-floating plants or attached plants that have been dislodged from the substrate often drift onto beaches or into swimming areas. Drying and decaying aquatic plants often produce objectionable odours and provide breeding areas for a variety of insects.

Increased plant growth can be caused by the presence of excess nutrients, for example, from various agricultural practices and private waste inputs that increase the amount of phosphorus and nitrogen. The results of this increase in nutrients is called cultural eutrophication. The natural aging (eutrophication) of bodies of water occurs much more slowly and does not change on a time scale to be of significant interest in the context of this document. Increased silt loads, changes in shorelines, and land use all contribute to alterations in aquatic habitats.

Some odours from lake water originate from natural decomposition of algae and other aquatic plant materials. These kinds of odours are typically vegetable or "earthy" in nature; however, decomposing masses of the filamentous alga ***Cladophora*** in Lake Erie and Lake Ontario have been described as "pigpen" (Neil 1975).

Several species of algae produce very different, sometimes offensive odours while actively growing in lake water. Odours described as fishy, grassy, aromatic, or musty have been attributed to several species of diatoms, blue-green algae, and Chrysophyceae (Palmer 1962; Taft 1965). Nicholls *et al*. (1980) reported serious odour production by the prymnesiophyte ***Chrysochromulina breviturrita*** in recreational lakes in Ontario and New Hampshire. The odour produced by this species, described by cottagers as "rotten-cabbage" or "garbage-dump," seems to be restricted to slightly acidic lakes. ***Chara***, a rooted macroscopic alga often mistaken for an aquatic vascular plant, has a very strong, objectionable odour. It is commonly found growing in extensive mats on lake bottoms (Warrington 1989).

Odours from lake water can be measured by the Threshold Odour Test (American Public Health Association 1989). The Threshold Odour Number (TON) is the ratio by which the sample must be diluted with odour-free water so that the odour is just detectable by a test panel of several people. Many natural surface waters not influenced by odour-producing algae have a TON of 5, whereas others with excessive algal growth may have a TON value exceeding 200.

Summary

1. Some biota that could be a nuisance to bathers if present in large numbers – e.g., leeches, mussels, biting insects, floating or rooted aquatic plants, phytoplankton, periphyton, and growths such as sewage fungus – should be absent from areas intended for development as bathing beaches.

2. The presence of large numbers of midges and aquatic worms, which can tolerate polluted, especially organically enriched, conditions (e.g., as caused by sewage), would indicate that the water quality was probably not good enough for recreational use.

5. Physical and Chemical Characteristics

Methods for determining physical and chemical characteristics of recreational water can be found in *Standard Methods for the Examination of Water and Wastewater* (American Public Health Association 1989) and the **Analytical Methods Manual** (Environment Canada 1981).

5.1 pH

Maximum Limits

Both alkaline and acidic waters may cause eye irritation; consequently, the pH of the waters used for total body contact recreation should be in the pH range of 6.5 to 8.5. If the water has a very low buffering capacity, pH values from 5.0 to 9.0 should be acceptable.

Criteria

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 physico-chemical properties as tears, including a pH of 7.4, although there is some evidence to suggest that ophthalmologic 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 to 200 μeq/L (microequivalents per litre) of acid-neutralizing capacity (ANC), and many of these lakes have depressed pHs. Detailed maps depicting sensitive areas in some provinces have been prepared by the United States-Canada Research Consultation Group on the Long-Range Transport of Air Pollutants (1979).

Studies completed by Basu *et al*. (1984) used water from two inland lakes in Ontario: Clearwater Lake (pH about 4.5) with an ANC of -40 μeq/L (Yan 1980), and Red Chalk Lake (pH about 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).

In all cases, 1 eye was exposed to the low-pH water and the other to the higher-pH water. The human eyes were exposed for 5-minute periods, and no unusual signs or symptoms occurred. 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 leucocyte content of tears, changes 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.

5.2 Temperature

Maximum Limits

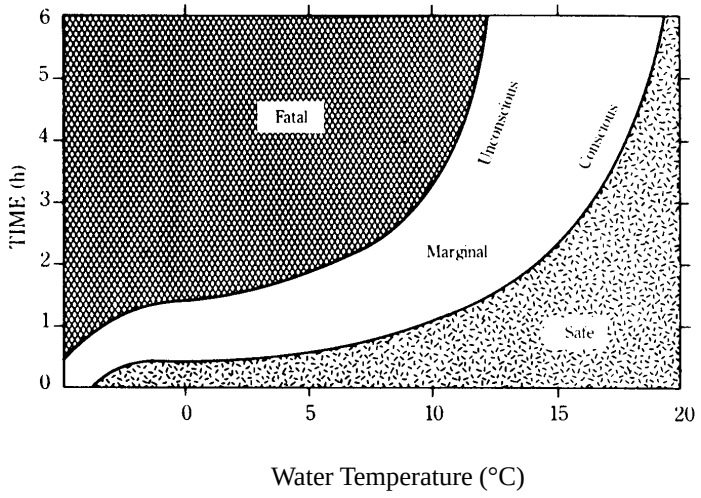
The thermal characteristics of waters used for bathing and swimming should not cause an appreciable increase or decrease in the deep body temperature of bathers and swimmers.

Criteria

The temperature of natural waters is an important factor governing the character and extent of recreational activities, primarily in the summer months. The upper recommended limit of temperature is 30°C. Scientific evidence suggests that prolonged immersion in water warmer than 34 to 35°C is hazardous. The degree of hazard varies with the water temperature, immersion time, and the metabolic rate of the swimmer.

Persons engaging in winter water recreation such as ice skating and fishing do so with the knowledge that whole body immersion must be avoided. Accidental immersion in water at or near freezing temperatures is dangerous, because the median lethal immersion time is less than 30 minutes for children and most adults (Molnar 1946; National Academy of Sciences 1973) (Figure 1).

Figure 1. Relationship between water temperature and survival time in cold water



(Adapted from the Royal Life Saving Society of Canada 1978)

There is considerable variation from one individual to another in the rate of body cooling and the incidence of survival in cold water. The variability is a function of body size, fat content, prior acclimatization, 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).

In cold temperatures, the critical problem is to maintain body temperature. In cold water, body heat is lost primarily by conduction from the inner organs through the trunk. Exposure of the limbs plays a relatively minor role in overall heat loss. In several instances where drowning was reported as the cause of death, exposure to cold was probably the responsible factor (Keatinge 1969).

Contrary to earlier opinion, exercise in the water increases the loss of body heat and correspondingly decreases survival time. This is reflected in frequent reports of drownings of expert swimmers who tried to reach shore after a sinking, whereas those who remained in the water near the lost ship survived until rescued. A careful study of reported drowning cases carried out by Press (1969) seemed to bear out much of the above as regards survival in cold waters. He reported that 299 out of 874 drownings, or 34 per cent, occurred in waters that were listed as very cold (assumed to be below 20°C). In addition, a much higher percentage of those succumbing in cold water were considered to be good swimmers.

The safe upper limit of water temperature for recreational immersion varies from individual to individual and seems to depend on psychological rather than physiological considerations. Unlike cold water, the mass to surface area ratio in warm water favours the child. Physiologically, neither adult nor child 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). The rate at which heat is conducted from the immersed human body is so rapid that thermal balance for a body at rest in water can be attained only if the water temperature is about 34°C (Beckman 1963). The survival of an individual submerged in water at a temperature above 34 to 35°C depends on the tolerance to an elevation of the internal temperature, and there is a real risk of injury with prolonged exposure. Water ranging in temperature from 26 to 30°C is comfortable for most swimmers throughout prolonged periods of moderate physical exertion.

5.3 Aesthetics

Webster's Third New International Dictionary (1986) defines aesthetic as "appreciative of, responsive to the beautiful" in nature. Not only should a recreational area be free from objectionable factors, but various aesthetic components of the aquatic ecosystem and surrounding land should be present; for example, trees, other plants, birds, mammals, fish, and insects all play a role in the natural beauty of a recreational area.

All waters should be free from substances attributable to wastewater or other discharges in amounts that would interfere with the existence of life forms of aesthetic value:

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

The absence of visible debris, oil, scum, and other matter resulting from human activity is a strict requirement of aesthetic acceptability. Similarly, suggested values for light penetration, colour, and turbidity must be measured as being not significantly increased over natural background.

5.3.1 Turbidity

Maximum Limits

A limit of 50 Nephelometric Turbidity Units (NTU) is suggested.

Criteria

Because filtration equipment and modern water treatment processes are not feasible at natural bathing areas, safety hazards associated with turbid or unclear water are dependent upon the intrinsic quality of the water itself. However, 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.

The current method of choice for turbidity measurements is the nephelometric method (American Public Health Association 1989). Nephelometric turbidimeters measure the intensity of light scattered at 90 degrees to the path of incident light, and levels can be approximately related to the standard Jackson candle method.

One special component of organic turbidity is microorganisms, which may accumulate in such large amounts that waters become unsightly and turbid. The summer blooms of blue-green algae in recreational surface waters and algal debris are examples of turbidity due to microorganisms (Mackenthun and Keup 1970).

Raw water levels can vary from 1 to 1000 NTU. Runoff water quality measurements indicated levels of 4.8 to 130 NTU during the first hour of an urban rainfall occurrence (U.S. Environmental Protection Agency 1978b). In the quiescent zone of a bathing beach or reservoir impoundment, turbidity measurements in the vicinity of 50 NTU would be sufficient to satisfy most recreational uses, including boating and swimming.

The natural turbidity of some bathing and swimming waters is often so high that visibility through the water is dangerously limited. If such areas conform with all other requirements, they may be used for bathing and swimming, provided that subsurface hazards are removed and the depth of water is clearly indicated by signs that are easily readable (National Academy of Sciences 1973).

5.3.2 Clarity – Light penetration

Maximum Limits

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

Criteria

It is important that water at bathing and swimming areas be clear enough for users to estimate depth, to see subsurface hazards easily, and to detect the submerged bodies of swimmers or divers who may be in difficulty. Aside from the safety factor, clear water fosters enjoyment of the aquatic environment. The clearer the water, the more desirable the swimming area (National Academy of Sciences 1973).

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).

The Secchi disc is a device used to measure visibility depths in water. The upper surface of a circular metal plate, 20 cm in diameter, is divided into 4 quadrants and painted so that the 2 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 plants and animals, suspended mineral particles, stains that impart a colour, detergent foams, dense mats of floating and suspended debris, or a combination of these factors.

5.3.3 Colour

Maximum Limits

An objective for the colour of recreational water largely depends on the preferences of users, and it is impossible to put an absolute value on it. Colour should not be so intense as to impede visibility in areas used for swimming. A maximum limit of 100 platinum-cobalt (Pt-Co) units was proposed by Environment Canada (1972), but no supporting evidence was given.

Criteria

There are two measures of colour in water – true and apparent. The true colour of natural water is the colour of water from which turbidity has been removed (i.e., filtered water) (American Public Health Association 1989).

Natural minerals give true colour to water; for example, calcium carbonate in limestone regions gives a greenish colour, ferric hydroxide, red. Organic substances, tannin, lignin, and humic acids from decaying vegetation also give true colour to water (Reid and Wood 1976).

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 (living) blue-green algae imparts a dark greenish hue; diatoms 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.

To measure true colour, the water has to be filtered or centrifuged to remove the sources of apparent colour. True colour is measured on the platinum-cobalt scale (Pt-Co units) and ranges from very low numbers in clear lakes to over 300 units in the very dark waters of peat bogs (Reid and Wood 1976). Apparent colour is an aesthetic quality and cannot be quantified.

The colour imparted by organic compounds has been discussed by a number of authors. Colloidal material with a diameter of 3.5 to 10 nm accounted for most of the colour in water studies by Black and Christman (1963), and Schindler and Alberts (1974) confirmed this statement. Humic substances are compounds with high molecular weights (ranging from several hundred to tens of thousands) (Schnitzer and Khan 1972) that are resistant to bacterial decomposition (Felbeck 1965; Christman and Ghassimi 1966). The compounds are the result of polymerization and microbial synthesis, which alter plant components such as lignin (Flaig 1964; Felbeck 1971).

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.

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 toward blue or green caused by differential light absorption and scattering of different wavelengths (Ruttner 1963; Reid and Wood 1976).

The colour of stream water is due to the same factors contributing to the colour of lake water, but there is not as much variety as in lakes. The upper reaches of most streams are characterized by clear water, except in the flood season, because of the lack of true plankton. Streams draining swamps are usually coloured by dissolved plant substances such as tannin.

Many industrial effluents and irrigation cause both true and apparent colour in receiving water.

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 colour of water affects aquatic life, but this subject will not be discussed in this document. The main effects of water colour on recreational activities are aesthetic and safety related. The aesthetic colour of water cannot be quantified, as there are as many preferences as there are people. 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 is not altered by any anthropogenic activities.

5.3.4 Oil and grease

Maximum Limits

Oil 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
- can form deposits on shorelines and bottom sediments that are detectable by sight or odour (International Joint Commission 1977).

Criteria

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. Natural biological populations release lipid compounds, which can form natural slicks.

The man-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.

The analytical method for oil and grease (detection limit 1.0 mg/L) gives only a gross idea of the amount present, and individual compounds cannot be identified (Environment Canada 1981).

It is very difficult to establish criteria for oil and grease, as the mixtures falling under this category are very complex. Very small quantities of oily substances make water aesthetically unattractive. The water may have an odour, or it may foul equipment or the bodies of bathers and shorelines, but the possibility exists that recreationists might still use the water in cases of low contamination. The toxicity of oily substances from ingestion, skin absorption, or inhalation of vapours is relatively low except in the case of aromatics (Gage 1924).

5.4 Chemical Characteristics

Some concern has been expressed about a risk to bathers from the presence of chemicals in recreational waters. Very few studies have been found in the literature that equate a hazard to the health of swimmers and others to skin absorption of contaminants present in river or lake water (Brown *et al*. 1984).

5.4.1 Inorganic chemicals

National surveys of the water quality of lakes and rivers used for recreational activities indicate that concentrations of inorganic chemicals are low (National Water Quality Data Bank 1988). Analyses (1983-1988) for heavy metals indicated that they are present in concentrations considerably below those recommended as guidelines for drinking water (Department of National Health and Welfare 1989). Aquatic life is considerably more sensitive to most toxic chemicals than are humans. It is very unlikely that there is a hazard to people engaged in recreational activities in and around rivers and lakes as a result of the presence in water of inorganic chemicals.

5.4.2 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 (National Water Quality Data Bank 1988).

National surveys have analyzed 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 (Department of National Health and Welfare 1989) and should not pose any threat to human health.

A review by Brown *et al*. (1984) of volatile solvents compared the estimated skin absorption dose with the oral dose ingested under a number of exposure situations, including swimming, bathing, and drinking. Their findings indicated that skin absorption contributed between 29 and 91 per cent of the total dose. For example, if a 21.9-kg child swam 1 hour in water (90 per cent submerged) and drank 1 L (his normal daily intake) of water containing 0.5 mg/L of toluene, the dose of toluene absorbed by the dermal route would be 91 per cent of the daily absorbed dose from the two sources.

In summary, there are some chemical contaminants that could be a cause for concern in recreational waters. Given the paucity of information on the type of chemical, the effective concentration, and the effects, it is difficult to set guidelines at the present time.

Summary

1. It is recommended that no measurable limits be established for chemicals in recreational water for human exposure risk because of lack of sufficient scientific information. Decisions for use should be based on aesthetic quality (e.g., presence of odour or visible oil and grease) and other factors considered in the environmental health assessment (e.g., proximity to industrial discharge).

6. Microbiological Sampling and Analysis

6.1 Sampling

In recreational water quality investigations, the purpose of sampling is to obtain aliquots that are as representative as possible with respect to the microbiological properties of the area. Sampling should be conducted during the bathing season, but it is most appropriate when recreational waters are suspected as a source of waterborne disease. Regular sampling may not be necessary at all recreational water use areas. Historical data, combined with an annual environmental health assessment, may indicate that only occasional sampling is necessary. If a deterioration of water quality has occurred, then routine monitoring of the area should be carried out. Such an approach will allow health officials to concentrate their resources on beaches of questionable quality. By taking the following factors into consideration, an effective sampling program can be designed to optimize the estimation of fecal indicator bacteria in recreational waters.

6.1.1 Sampling locations

Most bodies of water used for recreational purposes frequently lack homogeneity with respect to their microbiological properties, thus making multi-point sampling necessary. The sites should be selected on the basis of information gathered during the environmental health assessment. Ideally, the sites chosen should be representative of the water quality throughout the whole bather exposure area. The selection of sites should pay particular attention to site-specific conditions that may influence the levels and distribution of indicator organisms and pathogens.

The sampling sites should include points of greatest bather activity as well as peripheral points subject to external fecal pollution. Natural or artificial streams discharging storm water and sewage can give certain sections of a body of water very different microbiological qualities than the body at large. The degree of heterogeneity can also be affected by rainfall, wind velocity and direction, and tides. In larger bodies, the contribution of local events is somewhat diminished by the large volumes of water involved.

The importance of sampling location has been considered in a number of studies. Brenniman *et al*. (1981), in a study of water sampling design at two Lake Erie beaches, observed that levels of indicator bacteria varied significantly with the time and day of collection, but not within the various sampling sites in the bathing areas. The authors concluded that at beaches

where the dispersion of any fecal input is incomplete, sampling at various locations as well as during periods of maximum bather load should be conducted.

The collection of subsurface samples at wading depth should be considered where the water has been stirred up either by bather activity or by the person collecting the sample (Warrington 1989). Two recent epidemiological studies have discussed the importance of sampling in shallow water frequented by young children (Fattal *et al*. 1986; Seyfried 1987).

6.1.2 Frequency of sampling

A water sample provides a quantitative estimate of the bacteria present at a particular site and instant. As the total number of samples increases, the more representative the data will be of the overall water quality.

Samples should be collected at random intervals and at times of greatest user activity (i.e., mid-afternoon on weekends or holidays, as recommended by Sherry 1986), as well as at times when maximum fecal contamination can be expected (e.g., periods of storm water runoff and high onshore winds that upwell bottom sediments). If concentrations of fecal indicator bacteria fluctuate cyclically (e.g., effluent discharges at regular intervals or tidal variations), as shown by Churchland and Kan (1982), then samples should be collected during all phases of the cycle in addition to periods of high bather density.

The minimum recommended sampling frequency for routine investigations is five samples in not more than 30 days from each sampling location. At beaches with higher bather densities or at those known to have poor water quality, or in cases of suspected waterborne diseases associated with bathing, then the sampling frequency should be increased. The number of samples will be determined from the identification of sampling locations described above. Occasional sampling should be adequate in areas that historically have had acceptable water quality. However, if information gathered prior to the bathing season indicates that the water quality may have deteriorated, then routine monitoring should be initiated.

When analyses indicate that a single sample contains more than 4000 ***Escherichia coli*** or fecal coliforms/L or more than 700 enterococci/L, resampling of the area is required. The number of samples collected and their location should be sufficient to indicate the possible sources of contamination.

6.1.3 Sampling procedures for water

Samples for microbiological examination should be collected in sterile, 200- to 500-mL “environmental sensitive” containers. When sampling is done by hand, the bottle should be held near the base with one hand, the cap removed, and the bottle mouth plunged downward into the water. The bottle is tilted slightly upward to displace the air, then pushed forward against the

current away from the hand and the boat or sampling platform (if used) to avoid contamination. The sampling depth should be 15 to 30 cm below the surface in both deep and shallow waters. When collecting is done with a sampling pole, the bottle should be fit into the holder in the recommended manner, the cover removed, and the sample collected, upstream away from the collector, by simulating the scooping motion of the hand-collected sample.

With either method, a small amount of sample should be poured out, leaving an airspace to allow for proper mixing prior to analysis. The cap should be replaced and the bottle labelled and stored in an ice chest. The samples must be collected and processed individually. Composite samples are not acceptable.

6.1.4 Sampling procedures for sediments

When evidence indicates that bathing beaches could be the source of waterborne diseases among bathers, sediment sampling and analysis for suspected pathogens are also recommended. Many investigations have demonstrated that pollution indicator bacteria and pathogenic bacteria survive for extended periods in sediments (e.g., Burton *et al*. 1987).

Sediment samples may be collected using sterile, 250- to 500-mL wide-mouth jars, observing the same precautions used with water sampling to ensure aseptic collection. In shallow waters, the jars are pushed along the bottom, collecting the material at the sediment-water interface, until half full. The excess water is poured off, and the sample is stored as described above. In deeper waters, sediment samplers used for collecting benthic invertebrates, such as the Ponar or Ekman grabs, can also be used (American Public Health Association 1989). When sediments are brought to the surface, a subsample is aseptically transferred from the centre of the material to the sterile jar.

6.1.5 Sample preservation and storage

Water and sediment samples should be maintained at 1 to 5°C and processed within 30 hours after collection. For transport to the laboratory, the sample bottles should be placed in an insulated ice chest containing melting ice or freezer packs; to prevent the possibility of contamination, total immersion of bottles in the water should be avoided. The samples should never be frozen. If freezer packs are used, the samples should be protected from direct contact to avoid freezing. Storage in the dark under these conditions (or at 4 to 5°C in a refrigerator) minimizes die-off and multiplication for at least 30 hours after collection. A sample preservation experiment conducted by Dutka and El-Shaarawi (1980) indicated that when water samples were stored at 1.5°C, the concentrations of indicator organisms were stable for at least 24 hours. Neither the original water temperature nor bacteria and nutrient load appeared to affect preservation.

If the microbiological counts are to be used in legal action, the samples must be delivered to the laboratory within six hours of collection and processed within two hours of receipt with proof of continuity of possession (American Public Health Association 1989).

6.2 Methods for Microbiological Analysis

6.2.1 *Escherichia coli* and fecal coliforms

The 16th edition of *Standard Methods for the Examination of Water and Wastewater* (American Public Health Association 1989) contains two official methods for the determination of fecal coliforms: the multiple tube fermentation or most probable number (MPN) procedure, and the membrane filtration (MF) technique.

Multiple Tube Fermentation or Most Probable Number (MPN) Method

This procedure is not an actual enumeration of bacteria but is a statistically derived index that gives the probable number of bacteria present in a sample. A series of replicate tubes of lauryl tryptose broth (usually five) is inoculated with decimal quantities of the sample and examined for growth or gas production after a 48-hour incubation at 35°C. Positive tubes are transferred to EC broth and incubated at 44.5°C for 24 hours. The presence of gas within 24 hours or less is considered a positive reaction and indicates coliforms of fecal origin. The number of positive tubes per dilution is referred to a standard MPN (Most Probable Number) table, which provides an estimate of the fecal coliforms per 100 mL of sample. If an estimate of ***E. coli*** is desired, positive EC tubes are streaked on EMB agar and incubated at 35°C for 24 hours, and isolated colonies are subcultured and identified by routine IMViC procedures.

The method is time-consuming to perform, requires large amounts of media, glassware, and incubator space, and requires 72 hours for a confirmed test. Except for turbid waters or water suspected of containing stressed organisms, the method has been replaced by the membrane filtration technique. However, the addition of 4-methylumbelliferone glucuronide (MUG) to coliform and fecal coliform broths for the direct detection of ***E. coli***, described by Feng and Hartman (1982), should make it easier and quicker to enumerate ***E. coli*** in turbid samples.

Membrane Filtration (MF) Technique

In this procedure, fecal coliforms in water are measured directly. The sample (usually 100 mL) is passed through a filter that retains the bacteria; the filter is placed on the surface of an appropriate medium (mFC broth for fecal coliforms) and incubated. After 24 hours' incubation in a water bath at 44.5°C, blue colonies typical of fecal coliforms are counted and recorded as the number of fecal coliforms per 100 mL. In recreational water contaminated with sewage, dilution of the sample to avoid confluent growth may be necessary. The anaerobic incubation of membranes to suppress the growth of non-coliform bacteria has also been described (Doyle *et al.* 1984). One disadvantage with the mFC broth is its inability to distinguish between ***E. coli*** and other thermotolerant, lactose-fermenting species. To overcome this problem, an MF method for enumerating ***E. coli*** in water has been developed (Dufour *et al*. 1981). The procedure uses a medium (mTEC) for lactose-fermenting, Gram-negative bacteria, a resuscitation step for stressed organisms, and an ***in situ*** urease test to differentiate ***E. coli*** (urease-negative) from other thermotolerant fecal coliforms (mostly urease- positive). This method should be useful for counting ***E. coli*** in most surface waters. However, because some ***Klebsiella pneumoniae*** subspecies are also urease-negative, the method may not be useful in waters receiving industrial effluents known to contain high levels of ***Klebsiella pneumoniae***. In this case, the mTEC medium incorporating indoxyl ß-D-glucoside might be more appropriate (Shaw and Cabelli 1980). Most recently, the incorporation of 4-methylumbelliferone glucuronide (MUG) into the various fecal coliform media to increase their specificity for ***E. coli*** has also been examined (Freier and Hartman 1987; Brodsky 1989; Young 1989). Various laboratories in Canada may wish to evaluate the applicability of these methods to recreational waters in their regions.

The advantages of the MF technique include reduction of space, labour, and equipment necessary, ability to test large volumes of water, rapidity and ease of testing, and a high degree of reproducibility. With the use of portable equipment, it can also be employed directly in the field.

Widespread use of the MF technique has revealed some unforeseen problems. Many investigators have shown that there are significant differences between various membranes in their ability to recover fecal coliforms from natural waters. For example, Tobin and Dutka (1977) concluded that membrane filters were not equivalent in their ability to recover bacteria from water samples and pointed out the great need for standardization. Apart from brand differences in filters, many studies have also shown that the MPN method often yielded better recovery of fecal coliforms than the MF technique. It is believed that the broths used in the MPN test provide a more favourable environment than the selective medium and membrane structure of the MF test for the recovery and growth of stressed fecal coliforms. However, despite

this problem, the MF technique, particularly when used with resuscitation techniques, is probably sufficiently precise to detect small differences in the pollution index of a given area when sampled regularly.

6.2.2 Enterococci

Enterococci in marine and freshwater recreational areas are usually enumerated by the MF technique described by the U.S. Environmental Protection Agency (1985). After filtering a portion of the sample, the membrane is placed on mE agar and incubated at 41°C for 48 hours. The membranes are then transferred to EIA plates and re-incubated for an additional 20 minutes. All pink to red colonies with black or reddish-brown precipitates are considered enterococci. A 1-step modification of this MF procedure is also being investigated (Dufour 1989). Highly turbid waters and those directly influenced by chlorinated sewage should be examined by an MPN method, using azide dextrose broth followed by confirmation with Pfizer selective enterococcus agar (American Public Health Association 1989).

6.2.3 *Pseudomonas aeruginosa*

A variety of enumeration procedures for ***Pseudomonas aeruginosa*** in natural waters is available. Levin and Cabelli (1972) described an MF technique and medium (mPA) that were more efficient and accurate than the MPN methods in use. Dutka and Kwan (1977) confirmed their findings and, by slightly modifying the mPA medium and using a longer incubation period, increased the sensitivity of the test. Brodsky and Ciebin (1978) further modified the medium and reported recoveries of ***P. aeruginosa*** comparable with those of Dutka and Kwan after only 24 hours' incubation.

However, if turbid waters or sediments are examined, the MPN method must be employed (Environment Canada 1978; American Public Health Association 1989). This procedure requires extended incubation periods and confirmation of presumptive positive tubes. Using an MPN procedure, Seyfried *et al*. (1985b) reported higher recoveries of ***P. aeruginosa*** from sediments than from ambient waters.

6.2.4 *Staphylococcus aureus*

The American Public Health Association (1989) lists a tentative MPN method for enumeration of ***Staphylococcus aureus*** from waters. An MF technique designed to count ***S. aureus*** in swimming pools (Alico and Dragonjac 1978) was also found useful for recreational waters (Seyfried 1980). Recently, a new MF medium for enumerating total staphylococci as well as ***S aureus*** has been formulated (Borrego *et al*. 1987a).

6.2.5 *Salmonella and Shigella*

Many methods are available for the isolation of ***Salmonella and Shigella*** from water and sediments using concentration and enrichment techniques followed by identification procedures or detection by fluorescent antibody techniques (Environment Canada 1978; American Public Health Association 1989). MPN and MF techniques for the quantitative determination of ***Salmonella*** have also been described (American Public Health Association 1989).

6.2.6 *Aeromonas*

There are a few methods available for the enumeration of ***Aeromonas*** in fresh and marine waters. MPN methods were used to count ***A. hydrophila*** in an estuary (Kaper *et al*. 1981). MF techniques for both fresh and marine waters have also been described (Rippey and Cabelli 1979; Havelaar *et al*. 1987).

6.2.7 *Campylobacter jejuni*

At present, there are no standard methods for the enumeration or detection of ***Campylobacter jejuni*** in water (American Public Health Association 1989). However, the enumeration of ***Campylobacter*** spp. using MPN methods followed by steps to identify ***C. jejuni*** has been reported (Bolton *et al*. 1987; Carter *et al*. 1987).

6.2.8 *Legionella*

Media and methods for the isolation and enumeration of ***Legionella*** in water have been described (Calderson and Dufour 1984; Hsu *et al*. 1984; Voss *et al*. 1984). The American Public Health Association (1989) has also summarized available information on specimen collection, identification, and isolation of ***Legionella*** species.

6.2.9 Protozoa

The technique for recovery of protozoa from waters is complex and involves two steps: the concentration of large volumes of water, and a microscopic identification using ordinary, phase-contrast, or fluorescent microscopy.

The American Public Health Association (1989) details a sampling device used for the detection of ***Giardia lamblia***. Quantification of ***Giardia*** cysts by membrane filtration has been suggested by Spaulding *et al*. (1983). Jakubowski and Ericksen (1979) reviewed the methods for the detection of ***Giardia*** cysts in water, and Sauch (1985) detailed their microscopic identification.

Cryptosporidium spp. can be concentrated from waters using polypropylene cartridge filters, as described by Musial *et al*. (1987). The identification of oocysts in river water has been reported by Ongerth and Stibbs (1987) and by Gallaher *et al*. (1989).

6.2.10 Viruses and coliphages

The development of methods for the concentration of viruses from large volumes of water (Wallis *et al*. 1972; Payment *et al*. 1976; Sobsey *et al*. 1980; Gerba and Goyal 1982; Block and Schwartzbrod 1982; Gerba 1983; Payment and Trudel 1988) and their detection by highly sensitive methods (Payment and Trudel 1985; Margolin *et al.* 1986) now allow the virological analysis of surface waters. The methodology for concentration and isolation of viruses in large volumes of water has been standardized to some extent (American Public Health Association 1989), so that monitoring of recreational waters is possible if epidemiological data indicate a need. The use of positively charged micro-porous filters has been recommended by Sobsey and Jones (1979), although wound fibreglass depth filters were found to be less expensive by Payment and Trudel (1988).

There are now available some rather simple, quick, and inexpensive procedures for the monitoring of coliphages and bacteriophages in water. One of the most sensitive procedures for enumerating coliphages from water or effluents is described in *Standard Methods for the Examination of Water and Wastewater* (American Public Health Association 1989) using ***E. coli*** (ATCC 13706) as host.

6.2.11 Toxic phytoplankton

The presence of potentially toxic blue-green species can be determined microscopically, but this technique cannot distinguish toxic from non-toxic strains because the strains look alike.

Rapid chemical analyses using reversed-phase, high performance liquid chromatography (Harada *et al*. 1988), HPLC and internal surface reversed-phase columns (Meriluoto and Eriksson 1988) and high performance, thin-layer chromatography (Jamel Al-Layl *et al*. 1988) have been proposed, for toxins that affect the liver from ***Microcystis aeruginosa*** and ***Anabaena flos-aquae*** to replace the previous, more time- consuming methodology using gel filtration (Krishnamurthy *et al*. 1986).

The standard mouse bioassay (Bishop *et al*. 1959; Elleman *et al*. 1978) provides a rapid general assessment of the presence and toxicity of hepatotoxins. The survival time is a measure of toxicity. Falconer *et al.* (1981) and Siegelman *et al.* (1984) also provide guides to interpreting the results. Codd *et al*. (1989) describe in vitro cytotoxicity tests, immunoassays and other procedures now emerging to supplement the mouse bioassay.

Definite chemical analysis for the alkaloid neurotoxin anatoxin-a from ***Anabaena flos-aquae*** can be performed within a matter of hours (Smith and Lewis 1987). Ikawa *et al*. (1982) and Sasner *et al*. (1984) describe chemical analyses for aphantoxins from ***Aphanizomenon flos-aquae***.

A 100-mL glass jar with a snap cap lid should be used to collect water samples for microscopic identification and enumeration of algal species. When the sample is taken, it should be preserved by the addition of Lugol's solution from an eye dropper until the sample is the colour of tea. The sample should be kept refrigerated.

For the toxicity assay and the extraction and identification of toxins, samples should be collected in two 1-L Nalgene containers. Enough algal mass should be collected to fill each container about three-quarters full. The samples should be frozen immediately after collection and kept frozen to preserve the toxins from decomposition.

Animals suspected of dying from blue-green algae poisoning should be autopsied by the local veterinary surgeon. It may also be appropriate during any investigation to sample and test the water for priority pollutants and pesticides, bacteriological quality, routine chemical constituents, and nutrients.

7. Posting of Recreational Waters

When the appropriate authority has determined that a beach or body of water is not suitable for recreational use, the public should be notified. Normally this involves placing one or more signs in conspicuous places along the beach or shoreline. These signs should be clear and concise as to the health risk and recommended course of action. They should be written in simple understandable text and symbols. The authority making the determination should be clearly indicated on the signs. The signs should be left in place only as long as necessary and promptly removed when the health hazard no longer exists.

Appendix 1:

Bathing Area Environmental Health Assessment

Beach Name: _____ Body of Water: _____

Type of Water: Fresh _____ Marine _____ Estuarine _____

Location: _____

Owner/Operator: _____

Address: _____ Phone: _____

Microbiological Hazards

As appropriate, consult with Public Health, Environment, and/or Agriculture Departments.

Sewage	Y	N
Is the water quality likely to be affected by discharges from:		
1. private on-site sewage disposal systems?	___	___
2. communal sewage treatment facilities?	___	___
3. agricultural activities?	___	___

Storm Water Runoff	Y	N
Is the water quality likely to be affected by runoff from:	___	___
1. municipal storm drains?	___	___
2. agricultural fields?	___	___
3. natural drainage?	___	___

Note: any of the above with a Yes answer require(s) a Detailed Investigation and Risk Analysis.

Physical Hazards

Access	Y	N
1. Is the beach protected from vehicle access?	___	___
2. Is the swimming area protected from water craft access?	___	___

Shoreline	Y	N
1. Is the shoreline free of large rocks, sharp objects, or other impairments?	___	___
2. Is the shoreline free of trees and shrubs that may impair visibility?	___	___

Bottom Conditions	Y	N
1. Does the bottom consist of material that is not easily stirred up?	___	___
2. Are the slopes gentle?	___	___
3. Is the bottom free of large rocks, sharp objects, or other obstructions?	___	___
4. Is the maximum depth of the swimming area less than 4.5 metres?	___	___
5. Is the bottom free of weeds?	___	___

Water Conditions	Y	N
1. Is the water elevation constant throughout the season?	___	___
2. Have lateral and helical currents been assessed as safe?	___	___
3. Have surf conditions been assessed for potential to create undertows and rips?	___	___
4. Are there 2.8 to 3.7 square metres of space available per swimmer?	___	___

Note: any of the above with a *No* answer require(s) a Detailed Investigation and Risk Analysis.

Chemical Hazards

As appropriate, consult with Environment and/or Agriculture Departments.

Chemical	Y	N
Is the water quality likely to be affected by:		
1. discharges from industrial sources?	___	___
2. agricultural drainage?	___	___
3. water craft mooring or use?	___	___

Note: any of the above with a *Yes* answer require(s) a Detailed Investigation and Risk Analysis.

Reporting Systems

	Y	N
1. Are there formal mechanisms for the reporting of abnormal waste discharges, spills, treatment bypasses, etc., to the local health authority?	___	___
2. Is there an illness or injury reporting mechanism in place that would be effective for epidemiological monitoring?	___	___

Sampling or Posting Recommendations

_____

_____

_____

_____

_____

Date of Assessment _____ Responsible Authority _____

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