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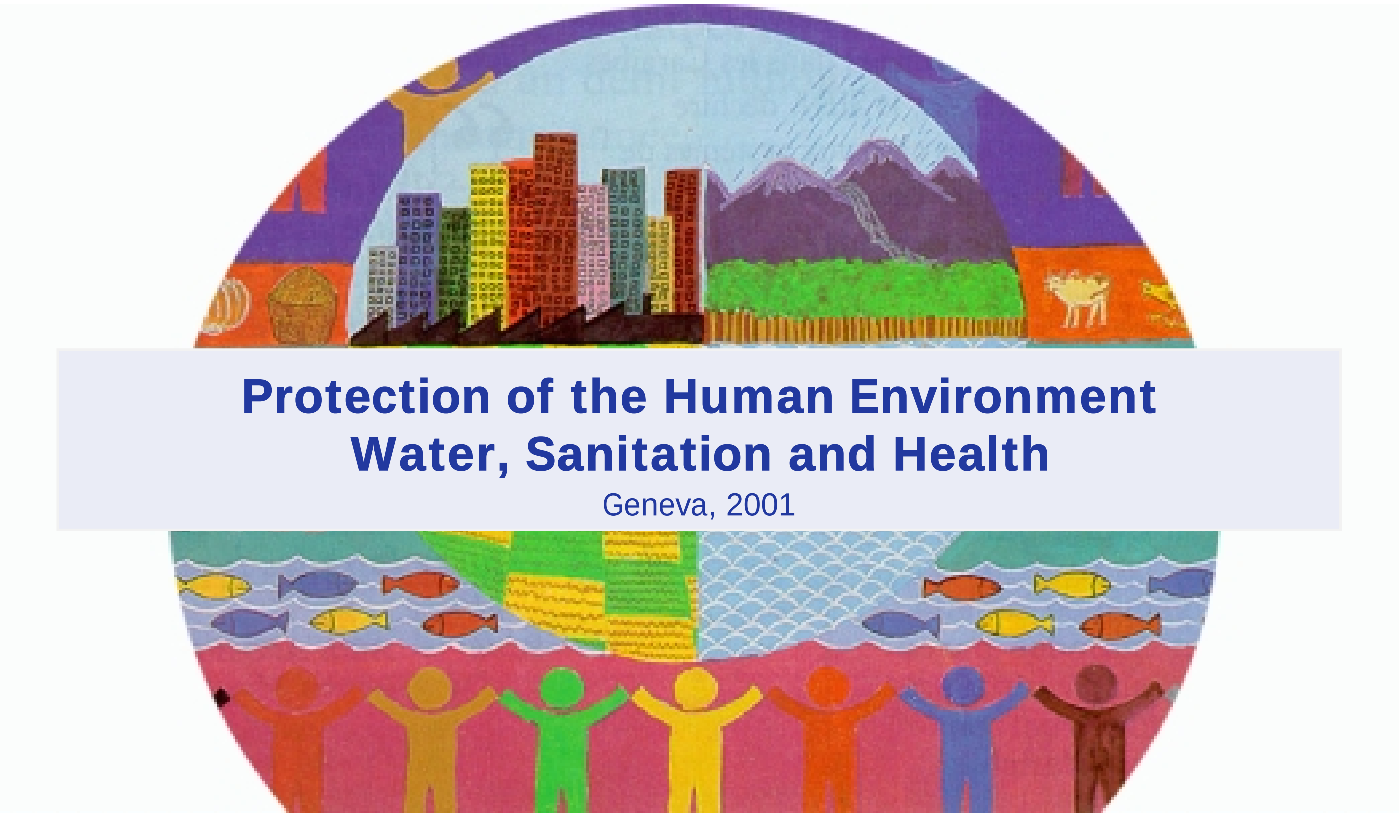
Bathing Water Quality and Human Health



Protection of the Human Environment
Water, Sanitation and Health

Geneva, 2001

Bathing Water Quality and Human Health: Faecal Pollution

Outcome of an Expert Consultation, Farnham, UK, April 2001

Co-sponsored by Department of the Environment, Transport and the Regions, United Kingdom

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Foreword

WHO has been concerned with health aspects of the management of water resources for many years and publishes various documents concerning the safety and importance for health of the water environment.

In 1994, following discussions between the WHO Regional Office for Europe and WHO Headquarters, it was agreed to initiate development of Guidelines concerning recreational use of the water environment. Guidelines of this type are primarily a consensus view amongst experts on the risks to health represented by various media and activities and are based on critical review of the available evidence. The *Guidelines for Safe Recreational-water Environments*, which result from this process, were released as drafts in two volumes. Volume 1 was released as a draft in 1998 and addresses coastal and fresh waters. Volume 2, released as a draft in 2000, addresses swimming pools, spas and similar recreational-water environments.

In light of limitations in approaches to both regulation and monitoring of the faecal pollution of recreational waters, an expert consultation co-sponsored by the US Environment Protection Agency was called in 1999. The meeting lead to the preparation of the "Annapolis Protocol." The protocol looked towards an improved approach for control of recreational-water environments that better reflected health risk and provided enhanced scope for effective management intervention.

This report is the outcome of an expert consultation that took place in Farnham, UK, in April 2001. The meeting was called to review experience with and assessment of Annapolis Protocol-type approaches in a number of environments world-wide; to review the evidence concerning health effects of faecal pollution of recreational waters that had become available since the release of the draft *Guidelines for Safe Recreational-water Environments* in 1998; and to merge these two lines of work to form a single coherent view on the protection of recreational-water users from hazards associated with faecal pollution of the waters they use. Included amongst material for discussion at the meeting were reports from individuals involved in trialling Annapolis Protocol-type approaches; a peer-reviewed but as yet unpublished (Kay et al., 2001) reanalysis of the study published as Kay et al. (1994) (which provides significant background for the derivation of Guideline Values in the draft Guidelines); and text revised to take account of comments received on the draft Guidelines, which itself had been subject to further peer review.

The output of the meeting, which follows, comprises a revised draft text for chapter 4 of Volume 1 of the *Guidelines for Safe Recreational-water Environments* as proposed by meeting participants. It is a consensus view amongst the experts who attended the meeting.

WHO wishes to express its appreciation to the experts who contributed to the meeting and who are listed in Annex 1, as well as to those who contributed to the original preparation of the draft Guidelines and those who submitted comments on the draft Guidelines.

Special thanks are also due to the Department of the Environment, Transport and the Regions, UK, for financial support for meeting implementation. (The views expressed in this report do not necessarily represent the views and policy of UK Government.)

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DRAFT *GUIDELINES FOR SAFE RECREATIONAL-WATER ENVIRONMENTS* VOLUME 1: *COASTAL AND FRESH WATERS* CHAPTER 4: FAECAL POLLUTION AND WATER QUALITY

Recreational-water standards have been successful in driving water quality improvements, increasing public awareness, contributing to informed personal choice and contributing to public health benefit. These successes are difficult to quantify, but the need to control and minimize adverse health effects has been the principal concern of regulation.

Regulatory schemes for the microbiological quality of recreational water have been largely based on percentage compliance with faecal indicator counts (EEC, 1976; US EPA, 1998). Constraints to these approaches include the following:

- Management actions are retrospective and can be deployed only after human exposure to the hazard.
- In many situations, the risk to health is primarily from human excreta, yet the traditional indicators of faecal pollution are also derived from other sources.
- There is poor inter-laboratory and international comparability of microbiological analytical data.
- While beaches are classified as either safe or unsafe, there is in fact a gradient of increasing variety and frequency of health effects with increasing faecal pollution of human and animal origin.

Furthermore, regulation tends to focus upon sewage treatment and outfall management as the principal, or only, effective interventions. Due to the high costs of these measures, local authorities may be effectively disenfranchised, and few options may be available for effective local interventions in securing water user safety from faecal pollution. The limited evidence available from cost–benefit studies of point source pollution control suggests that direct health benefits alone rarely justify the proposed investments and may be ineffective in securing regulatory compliance, particularly if non-human, diffuse faecal sources and/or stormwaters are the major contributor(s) (Kay et al., 1999). Furthermore, the costs may be prohibitive or may detract resources from greater public health priorities, such as securing access to a safe drinking-water supply, especially in developing regions. Lastly, considerable concern has been expressed regarding the burden (cost) of monitoring, primarily but not exclusively to developing regions, especially in light of the precision with which the monitoring effort assesses the risk to the health of water users and effectively supports decision-making to protect public health.

These limitations may be addressed through a monitoring scheme that combines microbiological testing with broader data collection concerning sources and transmission of pollution. There are two outcomes from such an approach — one is a recreational-water environment classification based on long-term analysis of data, and the other is immediate actions to reduce exposure, which may work from hour to hour or from day to day.

This chapter briefly describes approaches to risk assessment and risk management. It illustrates the application of approaches involving both an environmental hazard assessment and a microbiological water quality assessment. Guideline Values to be used in the microbiological assessment are also derived.

4.1 Health effects associated with faecal pollution

Recreational waters generally contain a mixture of pathogenic and non-pathogenic microbes. These microbes may be derived from sewage effluents, the recreational population using the water (from defecation and/or shedding), livestock (cattle, sheep, etc.), industrial processes, farming activities and wildlife; in addition, recreational waters may also contain truly indigenous pathogenic micro-organisms (described in chapter 5). This mixture can present a hazard to the bather where an infective dose of a pathogen colonizes a suitable growth site in the body and leads to disease. In the case of diseases transmitted by the faecal–oral route, this site is typically the alimentary canal. Other potential sites of infection include the ears, eyes, nasal cavity and upper respiratory tract.

What constitutes an ID_{50} (the dose required to infect 50% of individuals) depends upon the specific pathogen, the form in which it is encountered, the conditions of exposure and the host's susceptibility and immune status. For viral and parasitic protozoan illness, this dose might be very few viable infectious units (Fewtrell et al., 1993; Okhuysen et al., 1999). In reality, the body rarely experiences a single isolated encounter with a pathogen, and the effects of multiple and simultaneous pathogenic exposures are poorly understood (Esrey et al., 1985).

The types and numbers of various pathogens in sewage will vary depending on the incidence of disease in the contributing population and known seasonality of the human infections. Hence, numbers will vary greatly across different parts of the world and times of year. A general indication of pathogen numbers is given in Table 4.1.

Table 4.1: Examples of pathogens and indicator organisms in raw sewage

Pathogen/indicator[a]	Disease/role	Numbers per 100 ml
Bacteria		
Campylobacter spp.	Gastroenteritis	3700
Clostridium perfringens[b]	Indicator	$6 \times 10^4 – 8 \times 10^4$
Escherichia coli	Indicator (except specific strains)	$10^6–10^7$
Salmonella spp.	Gastroenteritis	0.2–8000
Shigella spp.	Bacillary dysentery	0.1–1000
Viruses		
Polioviruses	Indicator (virus strains), poliomyelitis	180–500 000
Rotaviruses	Diarrhoea, vomiting	400–85 000
Parasitic protozoa		
Cryptosporidium parvum oocysts	Diarrhoea	0.1–39
Entamoeba histolytica	Amoebic dysentery	0.4
Giardia lamblia cysts	Diarrhoea	12.5–20 000
Helminths (ova)		
Ascaris spp.	Ascariasis	0.5–11
Ancylostoma spp. And *Necator* sp.	Anaemia	0.6–19
Trichuris spp.	Diarrhoea	1–4

[a] Adapted from Yates & Gerba (1998). Many important pathogens in sewage have yet to be adequately enumerated, such as adenoviruses, Norwalk-like viruses, hepatitis A virus, etc. Parasite numbers vary greatly due to differing levels of endemic disease in different regions.
[b] From Long & Ashbolt (1994).

In both marine and freshwater studies of the impact of faecal pollution on the health of recreational-water users, several faecal indicator bacteria have been used for describing water quality. These bacteria are not postulated as the causative agents of illnesses in swimmers, but appear to behave in a similar way to the actual faecally-derived pathogens (Prüss, 1998).

The most frequent adverse health outcome associated with exposure to faecally contaminated recreational water is enteric illness, such as self-limiting gastro-enteritis. Transmission of pathogens that can cause gastro-enteritis is biologically plausible and is analogous to waterborne disease transmission in drinking-water, which is well documented. The association has been repeatedly reported in epidemiological studies, including studies demonstrating a dose–response relationship (Prüss, 1998).

A cause–effect relationship between faecal or bather-derived pollution and acute febrile respiratory illness (AFRI) is also biologically plausible. A significant dose–response relationship (between AFRI and faecal streptococci) has been reported in Fleisher et al. (1996a). AFRI is a more severe health outcome than the more frequently assessed self-limiting gastrointestinal symptoms (Fleisher et al., 1998). When compared with gastro-enteritis, probabilities of contacting AFRI are generally lower and are associated with similar faecal streptococci concentrations.

A cause–effect relationship between faecal or bather-derived pollution and ear infection has biological plausibility. However, ear problems are greatly elevated in bathers over non-bathers even after exposure to water with few faecal indicators. Associations between ear infections and microbiological indicators of faecal pollution and bather load have been reported (Fleisher et al., 1996a). When compared with gastro-enteritis, the statistical probabilities are generally lower and are associated with higher faecal indicator concentrations than those for gastrointestinal symptoms and for AFRI.

Increased rates of eye symptoms have been reported among bathers, and evidence suggests that bathing, regardless of water quality, compromises the eye’s immune defences, leading to increased symptom reporting in marine waters. Despite biological plausibility, no credible evidence for increased rates of eye ailments associated with water pollution is available (Prüss, 1998).

No credible evidence for an association of skin disease with either water exposure or microbiological water quality is available.

Most epidemiological investigations either have not addressed severe health outcomes (such as hepatitis, enteric fever or poliomyelitis) or have been undertaken in areas of low endemicity or zero reported occurrence of these diseases. By inference, from the very strong evidence for transmission of self-limiting gastro-enteritis, much of which may be of viral etiology, transmission of infectious hepatitis (hepatitis A and E viruses) and poliomyelitis, should exposure of susceptible persons occur, is biologically plausible. However, it was not reported in a 5-year retrospective study relying on total coliforms as the principal faecal indicator (Public Health Laboratory Service, 1959). Furthermore, sero-prevalence studies for hepatitis A among wind-surfers and water skiers who were exposed to contaminated waters have not identified any increased health risks (Philipp et al., 1989).

Nonetheless, there has been a documented association of transmission of *Salmonella paratyphi*, the causative agent of paratyphoid fever, with recreational-water use (Public Health Laboratory Service, 1959).

More severe health outcomes may occur among recreational-water users bathing in grossly sewage-polluted water who are short-term visitors from regions with low endemic disease incidence. Special measures may be justified under such circumstances.

Outbreak reports have noted cases of diverse health outcomes (e.g., gastrointestinal symptoms, typhoid fever, meningo-encephalitis) with exposure to recreational water and in some instances have identified the specific etiological agents responsible (Prüss, 1998). The causative agents of outbreaks may not be representative of the "background" disease associated with bathing in faecally-polluted water as detected by epidemiological studies. Table 4.2 lists pathogens that have been linked to swimming-associated disease outbreaks in the USA between 1985 and 1998.

Table 4.2: Outbreaks associated with recreational waters in the USA, 1985–1998[a]

Etiological agent	Number of cases	Number of outbreaks
Shigella spp.	1780	20
Escherichia coli O157:H7	234	9
Leptospira sp.	389	3
Giardia lamblia	65	4
Cryptosporidium parvum	429	3
Norwalk-like viruses	89	3
Adenovirus 3	595	1
Acute gastrointestinal infections (no agent identified)	1984	21

[a] From Kramer et al. (1996); Craun et al. (1997); Levy et al. (1998).

Two pathogenic bacteria, enterohaemorrhagic *Escherichia coli* and *Shigella sonnei*, and two pathogenic protozoa, *Giardia lamblia* and *Cryptosporidium parvum,* are of special interest because of the circumstances under which the associated outbreaks occurred — i.e., usually in very small, shallow bodies of water that were frequented by children. Epidemiological investigations of these, and similar, outbreaks suggest that the source of the etiological agent was usually the bathers themselves, most likely children (Keene et al., 1994; Cransberg et al., 1996; Voelker, 1996; Ackman et al., 1997; Kramer et al., 1998; Barwick et al., 2000). Each outbreak affected a large number of bathers, which might be expected in unmixed small bodies of water containing large numbers of pathogens. Management of these small bodies of water is similar to management of swimming pools (see Volume 2 of the *Guidelines for Safe Recreational-water Environments*).

Outbreaks caused by Norwalk-like viruses and adenovirus 3 are more relevant, in that the sources of pathogens were external to the beaches and associated with faecal contamination. However, high bather density has been suggested to account for high enterovirus numbers in a Hawaiian beach (Reynolds et al., 1998). *Leptospira* are usually associated with animals that urinate into surface waters, and swimming-associated outbreaks attributed to *Leptospira* are very rare (see chapter 5). Conversely, outbreaks of acute gastrointestinal infections with an unknown etiology are more common, but the symptomatology of the illness is frequently similar to that observed with viral infections.

Very few studies, other than those associated with outbreaks, have been conducted to determine the etiological agents related to swimming-associated illness. Some studies confirm that viruses are candidate organisms for the gastroenteritis observed in epidemiological studies conducted at bathing beaches. The serological data shown in Table 4.3 suggest that Norwalk virus, but not rotavirus, is a pathogen that has the potential to cause swimming-associated gastro-enteritis (El Sharkawi & Hassan, 1982). Application of reverse transcriptase-polymerase chain reaction has also indicated the presence of Norwalk-like viruses in fresh and marine waters (Wyn-Jones et al., 2000).

Table 4.3: Serological response to Norwalk virus and rotavirus in individuals with recent swimming-associated gastroenteritis[a,b]

Antigen	Number of subjects	Age range	Number with 4-fold titre increase
Norwalk virus	12	3 months – 12 years	4
Rotavirus	12	3 months – 12 years	0

[a] From El Sharkawi & Hassan (1982).

[b] Acute and convalescent sera were obtained from swimmers who suffered from acute gastro-enteritis after swimming at a highly contaminated beach in Alexandria, Egypt. The sera were obtained from 12 subjects, all of whom were less than 12 years old, on the day after the swimming event and about 15 days later.

4.2 Approaches to risk assessment and risk management

4.2.1 Harmonized approach and the Annapolis Protocol

A WHO expert consultation in 1999 formulated a harmonized approach to risk assessment and risk management for microbial hazards across drinking, recreational and reused waters. Priorities can therefore be addressed across all water types or within a type, when using the risk assessment/risk management scheme illustrated in Figure 4.1 (Bartram et al., 2001).

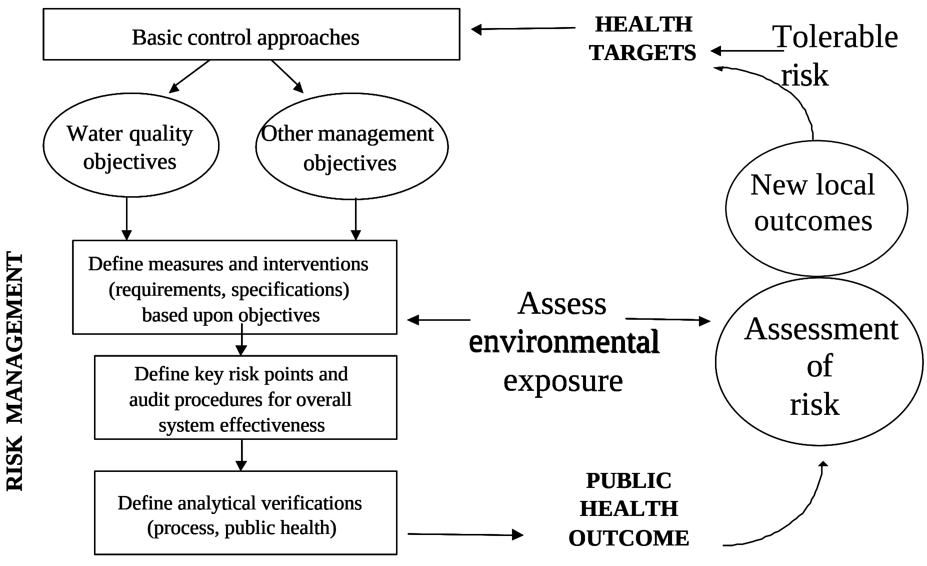


Fig. 4.1: WHO harmonized approach suitable to any water (adapted from Bartram et al., 2001)

The “Annapolis Protocol” (WHO, 1999; Bartram & Rees, 2000) represents an adaptation of the above “harmonized approach” to recreational water and was developed in response to concerns regarding the adequacy and effectiveness of approaches to monitoring and management of faecally-polluted recreational waters.

The most important changes recommended in the Annapolis Protocol were:

- the move away from the sole reliance on “guideline” values of faecal indicator bacteria to the use of a qualitative ranking of faecal loading in recreational-water environments, supported by direct measurement of appropriate faecal indicators; and
- provision to account for the impact of actions to discourage water use during periods or in areas of higher risk.

The protocol was tested in various countries, and recommendations resulting from these tests have been included in the Guidelines described here. These include the classification scheme that results from application of the Annapolis Protocol to the development of Guidelines for recreational-water environments, which is described in section 4.4.

4.2.2 Risk management

To meet health targets ultimately derived from tolerable risk, achievable objectives need to be established for water quality and associated management. A successful model is the Hazard Analysis Critical Control Point (HACCP) approach used in the food and beverage industry (Figure 4.2) (Deere et al., 2001). HACCP is an effective quality assurance system that has become the benchmark means to ensure food and beverage safety since its codification in 1993 by the Food and Agriculture Organization of the United Nations (FAO) and WHO Codex Alimentarius Commission.

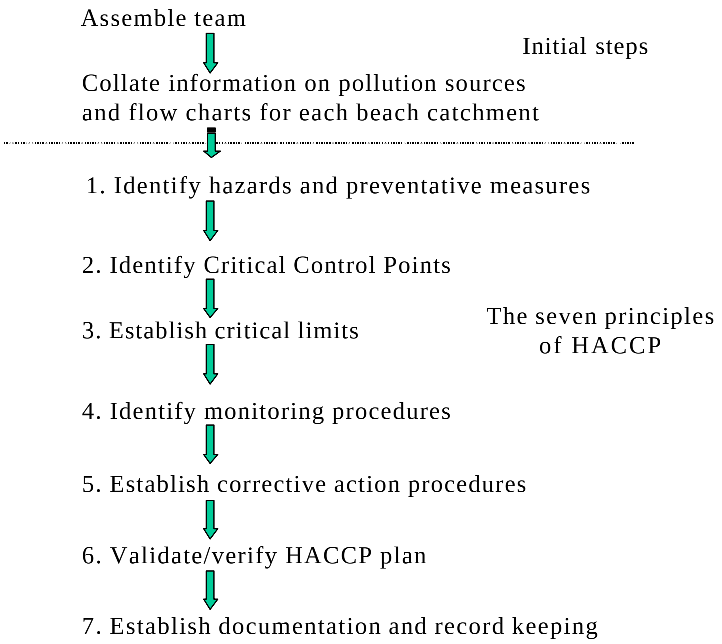


Fig. 4.2: The principles of HACCP as applicable to recreational waters

For recreational waters, the HACCP approach has been interpreted as described in Box 4.1, using the generic framework illustrated in Figure 4.2. This risk management procedure should be approached in an iterative manner, with increasing detail proportional to the scale of the problem and resources available. By design, HACCP addresses principally the needs for information for immediate management action; when applied to recreational-water use areas, however, its information outputs are also suitable for use in longer-term classification.

Box 4.1: Implementation of HACCP approach for recreational-water management

Principle	Implementation
Initial steps	
Assemble HACCP team	• The team is formed to steer the overall process. Composition of the team should be such as to represent all stakeholders and cover all fields of expertise as much as possible. Representatives of health agencies, user groups, tourism industry, water and sewage industry, communities, competent authorities, potential polluters, experts in hazard and risk analysis, etc., should all therefore be considered.
Collate historical information	• Summarize previous data from sanitary surveys, compliance testing, utility maps of sewerage, water and stormwater pipes and overflows. • Determine major animal faecal sources for each beach catchment. • Reference development applications and appropriate legal requirements. • If no (historical) data are available, collect basic data to fill data gap/deficiency.
Produce and verify flow charts	• Produce and verify flow charts for faecal pollution from source(s) to recreational exposure area(s) for each beach catchment. This may require a new sanitary survey. • The series of flow charts should illustrate what happens to water between catchment and exposure in sufficient detail for potential entry points for different sources of faecal contaminants to be pinpointed and any detected contamination to be traced.
Core principles	
1. Hazard analysis	• Identify human versus different types of animal faecal pollution sources and potential points of entry into recreational waters. • Determine significance of possible exposure risks (based on judgement, quantitative and qualitative risk assessment, as appropriate). • Identify preventive measures (control points) for all significant risks.
2. Critical control points	• Identify those points where monitoring would provide information to management so that management actions can have an impact on the exposure risk. Examples include municipal discharge points, treatment works operation, combined sewer overflows, illegal connections to combined sewers, etc.
3. Critical limits	• Determine measurable control parameters and their critical limits. Ideally, assign target and action limits to pick up trends towards critical limits (e.g., >10–20 mm rainfall in previous 24-h period or notification of sewer overflow by local agency).
4. Monitoring	• Establish a monitoring regime to give early warning of exceedances beyond critical limits. Those responsible for the monitoring should be closely involved in developing monitoring and response procedures. Note that monitoring is not limited to water sampling and analysis, but could

Principle	Implementation
	also include, for example, visual inspection of potential sources of contamination in catchment.
5. Management actions	• Prepare and test actions to reduce or prevent exposure in the event of critical limits being exceeded. Examples include building an appropriate treatment and/or disposal system, training personnel, developing an early warning system, issuing a media release and (ultimately) closing the area for recreational use.
6. Validation/ verification	• Obtain objective evidence that the envisaged management actions will ensure that the desired water quality will be obtained or that human recreational exposures will be avoided. This would draw from the literature and in-house validation exercises. • Obtain objective data from auditing management actions that the desired water quality or change in human exposure is in fact obtained and that the good operational practices, monitoring and management actions are being complied with at all times.
7. Record keeping	• Ensure that monitoring records are retained in a format that permits external audit and compilation of annual statistics. These should be designed in close liaison with those using the documents and records.

Variation in water quality may occur in response to events (such as rainfall) with predictable outcomes, or the deterioration may be constrained to certain areas or sub-areas of a single recreational-water environment. It is possible to effectively discourage use of areas that are of poor quality or discourage use at times of increased risk. Since measures to predict times and areas of elevated risk and to discourage bathing during these periods may be inexpensive, greater cost–benefits and improved possibilities for effective local management intervention are possible (see Box 4.1, management actions, validation and verification).

4.2.3 Risk assessment

Epidemiological studies can be used to demonstrate a causal relationship between exposure to faecally-polluted recreational water and an adverse health outcome (see section 4.1 and Prüss, 1998). Some types of studies are also suitable for use to quantify excess risk of infection attributable to recreational exposure.

The problems and biases in a range of epidemiological studies of recreational water and the suitability of studies to determine causal or quantitative relationships have been discussed in detail by Prüss (1998) (Box 4.2).

Box 4.2: Biases in epidemiological studies of recreational water (adapted from Prüss, 1998)

Type of bias	Description
Use of indicator microorganisms for assessing water quality of exposure	Temporal and spatial indicator variation is substantial and difficult to relate to individual bathers (Fleisher, 1990), unless the study design is experimental (Kay et al., 1994; Fleisher et al., 1996a). Also, the limited precision of methods for counting indicator organisms adds substantial measurement error (Fleisher, 1990; Fleisher et al., 1993). Furthermore,

Type of bias	Description
	the indicator organisms used are not at all times representative of viruses, which may represent an important part of the etiological agents.
Use of seasonal means for assessing water quality	Several studies use seasonal means of indicator organisms rather than daily measurements for characterizing individual exposure, thus adding substantial inaccuracy.
Assessment of exposure pathway	Certain studies do not take into account the potential infection pathway for defining exposure, e.g., mainly head immersion or the ingestion of water for gastroenteric symptoms. Difficulties in exposure recall further increase inaccuracy of individual exposure.
Non-control for confounders	The non-control for confounders, such as food and drink intake, age, sex, history of certain diseases, drug use, personal contact, additional bathing, sun, socio-economic factors, etc., may influence the observed association.
Selection of unrepresentative study population	Results reported for certain study populations (e.g., limited age groups or from regions with certain endemicities) are *a priori* not directly transferable to populations with other characteristics.
Self-reporting of symptoms	Most observational studies relied on self-reporting of symptoms by the study population. Validation of symptoms by medical examination (Kay et al., 1994; Fleisher et al., 1996a) would have reduced potential bias. External factors, such as media or publicity, may have influenced self-reporting.
Response rate	The response rate was more than 70% in all, and more than 80% in most, studies. Differential reporting, e.g., higher response among participants experiencing symptoms, would probably not have major consequences.
Recruitment method	The recruitment method consisted of approaching persons on the beach in almost all observational studies and was operated by advertisement for the randomized controlled studies.
Interviewer effect	Differences in the methodology of data collection among interviewers may influence the study results.

From a review of the literature, one or more key studies may be identified that provide the most convincing data with which to assess risk quantitatively. The United Kingdom's randomized epidemiological investigations provide such data for gastro-enteritis (Kay et al., 1994) and for AFRI and ear ailments associated with marine bathing (Fleisher et al., 1996a). These studies are described in more detail in section 4.3.1.

Useful insight into the effects of faecal pollution of water on human health can also be obtained from quantitative microbial risk assessment (QMRA). Rather than disease types being characterized, QMRA attempts to predict infection or illness rates from given densities of particular pathogens, assumed rates of ingestion and appropriate dose–response models for the population exposed (Haas et al., 1999).

While the conceptual framework for both chemical and microbial risk assessments is the same (Table 4.4), pathogens differ from toxic chemicals in several key ways:

- Different strains of the same pathogen have a variable ability to cause disease (differing virulence).
- This virulence can evolve and change as the pathogen passes through various infected individuals.

- Pathogens are generally not evenly suspended in water.
- Pathogens can be passed from one person to many (secondary spread), from either healthy but infected (asymptomatic) or ill (symptomatic) hosts (e.g., ratio of secondary to primary cases of 0.33 for *Cryptosporidium parvum* to over 1.0 for *Giardia lamblia* and Norwalk virus; Haas et al., 1999).
- Whether a person becomes infected or ill depends not only on the health of the person, but also on the person's pre-existing immunity and the pathogen dose.

Table 4.4: Risk assessment paradigm for any human health effect (adapted from Haas et al., 1999)

Step	Aim
1. Problem formalization and hazard identification	To describe acute and chronic human health effects (toxicity, carcinogenicity, mutagenicity, developmental toxicity, reproductive toxicity and neurotoxicity) associated with any particular hazard, including pathogens.
2. Exposure assessment	To determine the size and nature of the population exposed and the route, amount and duration of the exposure.
3. Dose–response assessment	To characterize the relationship between various doses administered and the incidence of the health effect.
4. Risk characterization	To integrate the information from exposure, dose–response and health effects steps in order to estimate the magnitude of the public health problem and to evaluate variability and uncertainty.

Constraints to the application of QMRA to recreational-water use include the current lack of specific data for many pathogens and the fact that pathogen numbers, as opposed to faecal indicators, vary according to the prevalence of specific pathogens in the contributing population and may exhibit seasonal trends. These factors suggest a general screening-level risk assessment (SLRA) as the first step to identify where further data collection and quantitative assessment may be most useful.

In the SLRA approach, a particular pathogen is used to represent its microbial group; for example, the occurrence of adenovirus, with its associated dose–response curve, is used as a predictor for all enteric viruses. Hence, conservative estimates of exposure to each pathogen group (viruses, bacteria, parasitic protozoa and helminths) may be used to characterize "total" risks from each of the groups of pathogens. The results of the SLRA should then indicate where further data are needed and if risks are likely to be dominated by a single class of pathogen or source (potentially defining options for risk management).

Given the limited array of microorganisms for which a dose–response has been estimated, SLRAs are currently limited to a few microorganisms, such as rotavirus, adenovirus, *Cryptosporidium parvum, Giardia lamblia* and *Salmonella* (Haas et al., 1999). An example of how to apply QMRA to bathing waters is provided in Box 4.3 (adapted from Ashbolt et al., 1997).

Box 4.3: Screening-level QMRA approach for bather risk (Ashbolt et al., 1997)

For a predominantly sewage-impacted bathing water, the concentration of pathogens in waters may be estimated from the mean pathogen densities in sewage and their dilution in bathing waters (based on the numbers of index organisms; see Table 4.5 below). As an initial conservative approximation of pathogen numbers in bathing waters, enterococci may be used as an index for the dilution of sewage-associated bacterial pathogens (e.g., *Shigella*) and spores of *Clostridium perfringens* or enterococci for the enteric viruses and parasitic protozoa. Alternatively, direct presence/absence measurement of pathogens in large volumes of bathing waters may be attempted (Reynolds et al., 1998). Next, an assumed volume of bathing water ingestion is required to determine the pathogen dose, typically 20–50 ml of water per hour of swimming.

Table 4.5: Geometric mean faecal indicators (index organisms) and various pathogens in primary sewage effluent in Sydney, Australia[a]

Thermotolerant coliforms (cfu/100 ml)	*Clostridium perfringens* (cfu/100 ml)	*Cryptosporidium* (oocysts/litre)	*Giardia* (cysts/litre)	Rotavirus (pfu/litre)[b]
1.33×10^7	7.53×10^4	24 (107)	14 000 (39 000)	470 (1245)

[a] Index bacteria and parasite data are from Long & Ashbolt (1994). Upper 90th percentile of mean is in parentheses.
[b] Total enteric virus estimate of 5650 and upper 90th percentile of 15 000 for raw sewage are from Haas (1983). Long & Ashbolt (1994) quoted a 17% reduction for adenoviruses, enteroviruses and reoviruses by primary treatment (discharge quality), and rotavirus was assumed to be 10% of total virus estimate.

After the general concentrations of pathogens from the three microbial groups of pathogens have been given, selected representatives are used for which dose–response data are available (e.g., *Shigella*, *Cryptosporidium*, *Giardia*, rotavirus and adenoviruses). Note that these specific pathogens may not necessarily be the major etiological agents, but are used as representatives characteristic of the likely pathogens. Risks from viral, bacterial and protozoan pathogens can then be characterized per exposure by applying published dose–response models for infection and illness (Haas et al., 1999). Risks experienced on different days are assumed to be statistically independent, and the daily risks are assumed to be equal. According to Haas et al. (1993), the annual risk can be calculated from a daily risk as follows:

$$P_{ANNUAL} = 1 - (1 - P_{DAILY})^N$$

where:
- P_{ANNUAL} is the annual risk of a particular consequence;
- P_{DAILY} is the daily risk of the same consequence; and
- N is the number of days on which exposure to the hazard occurs within a year.

Thus, swimming-related microbial risks may be characterized for varying season lengths, such as 7, 20 or 200 days per year. Furthermore, the separate impact from bather shedding can also be estimated (Gerba, 2000). For example, Rose et al. (1987) demonstrated that each freshwater bather contributed some 0.045 enteroviruses and 0.67 rotaviruses per 100 litres; the estimated median total virus contribution is 1.4×10^7 per 30-min swim (Gerba, 2000). Other pathogens may be estimated from their relationship to thermotolerant coliforms in stools, given that each bather sheds some 10^5–10^6 thermotolerant coliforms per swim — e.g., 1.4×10^4 parasitic protozoa per swimmer (average of all persons in the water, both shedding and not shedding) (Gerba, 2000). Accidental faecal releases (even at 1 per 1000 persons), particularly from children, may add substantially to these shedding loads of pathogens to bathing waters — i.e., 10^7–10^8 protozoa or 10^4–10^{13} enteric viruses per accident (Gerba, 2000).

Limited comparison between the screening-level QMRA approach and epidemiological investigation exists; for Sydney, both approaches appear to estimate similar levels of illnesses.

The only QMRA studies available (Sydney and Honolulu) (Haas, Rose and Gerba, personal communication; Ashbolt et al., 1997) provide supporting evidence for the results of various epidemiological studies. In particular, they indicated that enteric viruses represented the highest risk, 10- to 100-fold greater than total protozoan risks in Mamala Bay, Honolulu, but similar to *Giardia* risk in Sydney. Nonetheless, risks from *Giardia* cysts may be overestimated, given their likely inactivation within a few days in warm seawater (Johnson & Gerba, 1996). Risks from bacterial pathogens and *Cryptosporidium* were orders of magnitude less in both studies. Nonetheless, estimated entero- or adenovirus infections were low, varying from 10 to 50 per 10 000 people exposed over 7 days in Mamala Bay. The risk from infection was considered equal to or greater than the chance of infection from all other sources during autumn and spring, but 2–10 times lower in the summer, depending on the prevalence of various microbial hazards in the population.

Thus, QMRA can be a useful tool in screening sites where historical data are absent and the acquisition of new data is, for some reason, very difficult. The evidence base from QMRA reinforces epidemiological evidence suggesting that disease transmission is possible at recreational-water areas where water quality would have traditionally passed historical standards.

4.3 Guideline Values

In many fields of environmental health, regulations are set at a level of exposure at which no adverse health effects are expected to occur. This is the case for some chemicals in drinking-water, for example, such as DDT and copper.

For other chemicals in drinking-water, such as genotoxic carcinogens, there is no “safe” level of exposure. In these cases, standards (including WHO Guideline Values) are generally set at the concentration estimated to be associated with a certain (low) excess burden of disease. A frequent point of reference is a 1 in 100 000 excess incidence of cancer over a lifetime of exposure.

Guideline values and standards for microbial quality were originally developed to prevent the occurrence of outbreaks of disease. While they have often been tightened over time, there was limited information available concerning the degree of health protection they provide.

In the case of recreational waters, the quantitative epidemiological studies published in recent years enable the estimation of the degree of health protection (or, conversely, burden of disease) associated with a certain water quality. Further information on this is available in section 4.3.1, which illustrates the association of gastrointestinal illness and AFRI with water quality.

In setting Guidelines for recreational-water quality, it would be logical to ensure that the overall levels of health protection were comparable to those for other water uses. This would require comparison of very different adverse health outcomes, such as cancer, diarrhoea, etc. Significant experience has now been gained in such comparisons, especially using the metric of disability-adjusted life years (DALYs).[1]

When this is done for recreational waters, it becomes clear that typical standards for recreational water would lead to "compliant" recreational waters associated with a health risk very significantly greater than that considered acceptable in other circumstances (such as carcinogens in drinking-water).

Setting recreational-water quality standards at water qualities that would provide for levels of health protection similar to those accepted elsewhere would lead to standards that would be so strict as to be impossible to implement in many parts of the developing and developed world.

The approach adopted here therefore recommends that a range of water quality categories be defined and individual locations be classified according to these (see section 4.4.3). The use of multiple categories provides incentive for progressive improvement throughout the range of qualities in which health effects are believed to occur. The middle cutoff value would normally constitute the regulatory standard, where provision of a specific regulatory standard was wanted.

4.3.1 Selection of key studies

Numerous studies have shown a causal relationship between gastrointestinal symptoms and recreational-water quality as measured by indicator bacteria numbers (Prüss, 1998). Furthermore, a strong and consistent association has been reported with temporal and dose–response relationships, and the studies have biological plausibility and analogy to clinical cases from drinking contaminated water. Nonetheless, various biases occur with epidemiological studies (see Box 4.2).

In 19 of the 22 studies examined in Prüss's (1998) review, the rate of certain symptoms or symptom groups was significantly related to the count of faecal indicator bacteria in recreational water. Hence, there was a consistency across the various studies, and gastrointestinal symptoms were the most frequent health outcome for which significant dose-related associations were reported.

In marine bathing waters, the United Kingdom randomized controlled trials (Kay et al., 1994; Fleisher et al., 1996a) probably contained the least amount of bias. These studies give the most accurate measure of exposure, water quality and illness compared with observational studies where an artificially low threshold and flattened dose–response curve (due to misclassification bias) were likely to have been determined.

The United Kingdom randomized controlled trials therefore form the key studies for derivation of Guideline Values for recreational waters (Box 4.4). However, it should be

[1] A DALY expresses years of life lost to premature death (i.e., a death that occurs before the age to which the dying person could have been expected to survive if s/he were a member of a standardized model population with a life expectancy at birth equal to that of the world's longest-living population — Japan) and years lived with a disability of specific severity and duration. Thus, one DALY is one lost year of healthy life.

emphasized that they are primarily indicative for adult populations in marine waters in temperate climates. Studies that reported higher thresholds and case rate values (for adult populations or populations of countries with higher endemicities) may suggest increased immunity, which is a plausible hypothesis but awaits empirical confirmation. Most studies reviewed by Prüss (1998) suggested that symptom rates were higher in lower age groups, and the United Kingdom studies may therefore systematically underestimate risks to children.

Box 4.4: Key studies for Guideline Value derivation

The United Kingdom randomized trials were designed to overcome significant "misclassification" (i.e., attributing a daily mean water quality to all bathers) and "self-selection" (i.e., the exposed bathers may have been more healthy at the outset) biases in earlier studies. Both effects would have produced an underestimate of the illness rate.

This was done by recruiting healthy adult volunteers in urban centres during the 2 weeks before each of the four studies, conducted from 1998 to 1992 at United Kingdom beaches that passed existing European Union standards. Volunteers reported for an initial interview and medical examination 1–3 days prior to exposure. They reported to a beach on the study day and were informed of their randomization status into the "bather" or "non-bather" group (i.e., avoiding "self-selection" bias). Bathers were taken by a supervisor to a marked section of beach, where they bathed for a minimum period of 10 min and immersed their heads three times during that period. The water in the bathing area was intensively sampled during the bathing period to give a spatial and temporal pattern of water quality, which allowed a unique water quality to be ascribed to each bather derived from a sample collected very close to the time and place of exposure (i.e., minimizing "misclassification" bias). Five candidate bacterial faecal indicators were measured synchronously at three depths during this process. Enumeration of indicators was completed using triplicate filtration to minimize bias caused by the imprecision of indicator measurement in marine waters. All volunteers were interviewed on the day of exposure and at 1 week post-exposure, and they completed a postal questionnaire at 3 weeks post-exposure. These questionnaires collected data on an extensive range of potential confounding factors, which were examined in subsequent analyses. Bathers and all subsequent interviewers were blind to the measure(s) of exposure used in statistical analysis, i.e., faecal indicator organism concentration encountered at the time and place of exposure.

Gastro-enteritis rates in the bather group were predicted by faecal streptococci measured at chest depth. This relationship was observed at three of the four study sites; at the fourth, very low concentrations of this indicator were observed.

Bathers had a statistically significant increase in the occurrence of AFRI at levels at or above 60 faecal streptococci/100 ml.

The faecal indicator concentrations in recreational waters vary greatly. To accommodate this variability, the disease burden attributable to recreational-water exposure is calculated by combining the dose–response relationship with a probability density function (PDF) describing the distribution of indicator bacteria. This allows the health risk assessment to account for the mean and variance of the bacterial distribution encountered by recreational-water users.

The maximum level of faecal streptococci measured in the United Kingdom randomized controlled trials was 158 faecal streptococci/100 ml (Kay et al., 1994). The dose–response curve for gastro-enteritis derived from these studies and used in deriving the Guidelines below is therefore limited to values in the range from where significant effect was first recorded, 30–40 faecal streptococci/100 ml, to the maximum level detected. The probability of gastro-enteritis or AFRI at levels higher than these is unknown. In estimating the risk levels for exposures above 158 faecal streptococci/100 ml, the authors have adopted the assumption that the probability of illness remains constant at the same level as exposure to 158 faecal streptococci/100 ml (i.e., probability of 0.388), rather than continuing to increase. This assumption may be conservative and may need review as studies become available that clarify the risks attributable to exposures above these levels.

As the volunteers in the key studies were all healthy adults, risks to other groups, especially children, are probably underestimated by the results.

For marine waters, only faecal streptococci (enterococci) showed a dose–response relationship for both gastrointestinal illness (Kay et al., 1994) and AFRI (Fleisher et al., 1996a). Considerable discussion has arisen due to the steep dose–response curve reported in these studies, compared with previous studies. The best explanation of the steeper curve simply appears to be that with less misclassification and other biases, a more accurate measure of the association between indicator numbers and illness rates was made. A recent reanalysis of these data (Kay et al., 2001) using a range of contemporary statistical tools has confirmed that the relationships originally reported are robust to alternative statistical approaches. The slopes of the dose–response curves for gastrointestinal illness and AFRI are also broadly consistent with the dose–response models used in QMRA (Ashbolt et al., 1997).

4.3.2 The 95th percentile approach

Many agencies have chosen to base criteria for recreational-water compliance upon either 95% compliance levels (i.e., 95% of the sample measurements taken must lie below a specific value in order to meet the standard) or geometric mean values of water quality data collected in the bathing zone. Both have significant drawbacks. The geometric mean is statistically a more stable measure, but this is because the inherent variability in the distribution of the water quality data is not characterized in the geometric mean. However, it is this variability that produces the high values at the top end of the distribution that are of greatest public health concern.

The 95% compliance system, on the other hand, does reflect much of the top-end variability in the distribution of water quality data and has the merit of being more easily understood. However, it is affected by greater statistical uncertainty and hence is a less reliable measure of water quality, thus requiring careful application to regulation.

Other options include the percentile approach, in which a specified percentile, most commonly the 80th, 90th or 95th, is calculated. A limit can then be set for making judgements about the water quality, depending on whether the specified percentile value exceeds it or not. A simple ranking method by which a specified percentile may be calculated from the sample series being evaluated is given in Bartram & Rees (2000). Other methods for calculating sample series percentiles are given by Ellis (1989). Ninety-fifth percentile values calculated in this manner suffer from some of the same drawbacks described above for the 95% compliance system.

A more appropriate method of calculating the 95th percentile, which makes better use of all the data in the sample set, is to generate a probability density function (PDF) based on the distribution of indicator organisms over a defined bathing area and then to use the properties of this PDF to estimate the 95th percentile value of this distribution. In practice, the full procedure is rarely carried out, and 95th percentiles are calculated using the lognormal distribution method given in Bartram & Rees (2000). This is called a parametric method, since it requires the estimation of the population parameters known as the mean and standard deviation of the lognormal distribution. One limitation of the method is that if the samples are not lognormally distributed, it will yield erroneous estimates of the 95th percentile. Also, if there are data below the limit of detection, these data must be assigned an arbitrary value based on the limit of detection.

4.3.3 Guideline Values for seawater

The Guideline Values for microbiological quality given in Table 4.6 are derived from the key studies described above. The cut-off or bounding Guideline Values (40, 200, 500) are expressed in terms of the 95th percentile of numbers of faecal streptococci per 100 ml and represent readily understood levels of risk based on the exposure conditions of the key studies. The values may need to be adapted to take account of different local conditions and are recommended for use in the recreational-water environment classification scheme discussed in section 4.4.4.

For the purposes of water quality monitoring, the terms faecal streptococci, intestinal enterococci and enterococci are considered to be synonymous (Figueras et al., 2000). Exposure to recreational waters with these measured indicators refers to body contact that is likely to involve head immersion, such as swimming, surfing, white-water canoeing, scuba diving and dinghy boat sailing.

Available evidence suggests that the Guideline Values presented in Table 4.6 provide a lesser degree of health protection than that considered tolerable in other areas of environmental quality regulation. However, the central "200" cut-off or upper bounding value represents a stricter standard than is encountered in many areas at present. As noted above and in section 4.4, measures to discourage water use at times or in locations of greater risk may provide cost-effective means to improve health protection and water quality classification.

Table 4.6: Guideline Values for microbiological quality of recreational waters*

95th percentile value of faecal streptococci/ 100 ml (rounded values)	Basis of derivation	Estimated risk
=40	This value is below the NOAEL in most epidemiological studies.	**<1% GI illness risk** **<0.3% AFRI risk** This relates to an excess illness of less than 1 incidence in every 100 exposures. The AFRI burden would be negligible.

95th percentile value of faecal streptococci/ 100 ml (rounded values)	Basis of derivation	Estimated risk
41–200	The 200/100 ml value is above the threshold of illness transmission reported in most epidemiological studies that have attempted to define a NOAEL or LOAEL for GI illness and AFRI.	**1–5% GI illness risk** **>1.9% AFRI illness risk** The upper 95th percentile value of 200 relates to an average probability of one case of gastro-enteritis in 20 exposures. The AFRI illness rate at this water quality would be 19 per 1000 exposures, or approximately 1 in 50 exposures.
201–500	This level represents a substantial elevation in the probability of all adverse health outcomes for which dose–response data are available.	**5–10% GI illness risk** **1.9–3.9% AFRI illness risk** This range of 95th percentiles represents a probability of 1 in 10 to 1 in 20 of gastro-enteritis for a single exposure. Exposures in this category also suggest a risk of AFRI in the range of 19–39 per 1000 exposures, or a range of approximately 1 in 50 to 1 in 25 exposures.
>500	Above this level, there may be a significant risk of high levels of minor illness transmission.	**>10% GI illness risk** **>3.9% AFRI illness rate** There is a greater than 10% chance of illness per single exposure. The AFRI illness rate at the 95th percentile point of 500 enterococci per 100 ml would be 39 per 1000 exposures, or approximately 1 in 25 exposures.

* Notes:

1. Abbreviations used: AFRI = acute febrile respiratory illness; GI = gastrointestinal; LOAEL = lowest-observed-adverse-effect level; NOAEL = no-observed-adverse-effect level.
2. The “exposure” in the key studies was a minimum of 10 min bathing involving three immersions. It is envisaged that this is equivalent to many immersion activities of similar duration, but it may underestimate risk for longer periods of water contact or for activities involving higher risks of water ingestion (see also note 7).
3. The “estimated risk” refers to the excess risk of illness (relative to a group of non-bathers) among a group of bathers who have been exposed to faecally-contaminated recreational water under conditions similar to those in the key studies.
4. The functional form used in the dose–response curve assumes no excess illness outside the range of the data (i.e., at concentrations above 158 faecal streptococci/100 ml; see Box 4.4). Thus, the estimates of illness rate reported above are likely to be underestimates of the actual disease incidence attributable to recreational-water exposure.
5. This table would produce protection of “healthy adult bathers” exposed to marine waters in temperate north European waters.
6. It does not relate to children, the elderly or immuno-compromised, who would have lower immunity and might require a greater degree of protection. There are no available data with which to quantify this, and no correction factors are therefore applied.
7. Epidemiological data on fresh waters or exposures other than bathing (e.g., high-exposure activities such as surfing, dinghy boat sailing or white-water canoeing) are currently inadequate to present a parallel analysis for defined reference risks. Thus, a single microbiological value is proposed, *at this time*, for all recreational uses of water, because insufficient evidence exists at present to do otherwise. However, it is recommended that the severity and frequency of exposure encountered by special interest

groups (such as bodysurfers, board riders, windsurfers, sub-aqua divers, canoeists and dinghy sailors) be taken into account (chapter 1).

8. Where disinfection is used to reduce the density of indicator bacteria in effluents and discharges, the presumed relationship between faecal streptococci (as indicators of faecal contamination) and pathogen presence may be altered. This alteration is, at present, poorly understood. In water receiving such effluents and discharges, faecal streptococci counts may not provide an accurate estimate of the risk of suffering from mild gastrointestinal symptoms or AFRI.
9. Risk attributable to exposure to recreational water is calculated after the method given by Wyer et al. (1999), in which a $\log_{10}$ standard deviation of 0.8103 was assumed. If the true standard deviation for a beach were less than 0.8103, then reliance on faecal streptococci would tend to overestimate the health risk for people exposed above the threshold level, and vice versa (see Box 4.6).
10. Note that the values presented in this table do not take account of health outcomes other than gastro-enteritis and AFRI. Where other outcomes are of public health concern, then the risks should be assessed and appropriate action taken.
11. Guideline Values should be applied to water used recreationally and at the times of recreational use. This implies care in the design of monitoring programmes to ensure that representative samples are obtained. It also implies that data from periods of high risk may be ignored if effective measures were in place to discourage recreational exposure (see section 4.4.3).

4.3.4 Guideline Values for fresh water

No epidemiological studies from freshwater areas have been reported that have substantively addressed the concerns regarding misclassification bias described above (Box 4.2), and there is therefore inadequate evidence with which to directly derive a water quality Guideline Value for fresh water.

Of all the faecal indicators available, faecal streptococci/enterococci provide the best available match with health outcomes resulting from exposure to recreational waters in seawaters and alongside *E. coli* for fresh waters (Prüss, 1998).

Dufour (1984) discussed the significant differences in swimming-associated gastrointestinal illness rates in seawater and freshwater swimmers. The illness rate in seawater swimmers was about three times greater than that in freshwater swimmers. A similar higher illness rate in seawater swimmers is observed if the epidemiological study data of Kay et al. (1994) and Ferley et al. (1989) are compared. At the same enterococci densities, swimming-associated illness rate was about five times higher in seawater bathers (Kay et al., 1994) than in freshwater swimmers (Ferley et al., 1989). This difference may be due to the more rapid die-off of indicator bacteria than pathogens (especially viruses) in seawater compared with fresh water (Box 4.5). This relationship would result in more pathogens in seawater than in fresh water when indicator densities are identical, which would logically lead to a higher swimming-associated gastrointestinal illness rate in seawater swimmers.

Box 4.5: Differential die-off of indicator bacteria and pathogens in seawater and freshwater

Salinity appears to accelerate the inactivation of sunlight-damaged coliforms in marine environments, such that coliforms are appreciably less persistent than enterococci in seawater. On the other hand, enterococci and thermotolerant coliforms/*E. coli* have fairly similar sunlight inactivation rates in fresh water. Cioglia & Loddo (1962) showed that poliovirus, echovirus and coxsackie virus were inactivated at approximately the same rate in marine and fresh waters (Table 4.7), but it is important to note that other factors, such as water temperature, are more important than salinity for virus inactivation (Gantzer et al., 1998).

Table 4.7: Survival of enteroviruses in seawater and river water[a]

Virus strain	Die-off rates (in days)[b]	
	Seawater	River water
Polio I	8	15
Polio II	8	8
Polio III	8	8
Echo 6	15	8
Coxsackie	2	2

[a] Adapted from Cioglia & Loddo (1962).
[b] Maximum number of days required to reduce the virus population by 3 logs (temperature and sunlight effects not provided, but critical; Gantzer et al., 1998).

Thus, it appears likely that bacterial indicators have different die-off characteristics in marine and fresh waters, while human viruses are inactivated at similar rates in these environments. Hence, higher levels of exposure to viral pathogens may occur in marine waters at similar bacterial indicator levels, and this may require reconsideration in estimating microbial risks in the two environments. Overall, however, the balance of evidence suggests that, under many circumstances, the same levels of faecal indicator bacteria in freshwater and marine environments may imply a greater health risk in marine environments.

Application of the Guideline Values derived above for seawaters (Table 4.6) to fresh waters would be likely to result in a lower illness rate in freshwater swimmers, providing a conservative guideline in the absence of suitable epidemiological data for fresh waters.

Furthermore, in estuaries, the salinity is highly variable, and it would be difficult to decide when or whether a freshwater or marine standard should be applied to any given compliance location assuming separate marine and freshwater Guideline Values were specified.

Studies under way in the early 2000s may, once reported, provide a more adequate basis on which to develop freshwater Guideline Values in due course.

4.3.5 Adaptation of Guideline Values to national/local circumstances

As introduced in chapter 1, there is no universally applicable risk management formula. “Acceptable” or “tolerable” excess disease rates are especially controversial because of the voluntary nature of recreational-water exposure and the generally self-limiting nature of the most studied health outcomes (gastro-enteritis, respiratory illness). Therefore, assessment of recreational-water quality should be interpreted or modified in light of regional and/or local factors. Such factors include the nature and seriousness of local endemic illness, population behaviour, exposure patterns, and socio-cultural, economic, environmental and technical aspects, as well as competing health risk from other diseases that are not associated with recreational water. (From a strictly health perspective, many of the factors that might be taken into account in such an adaptation would often lead to the derivation of stricter standards.) What signifies an acceptable or tolerable risk is not only a regional issue, however, as even within a region children and people from lower socioeconomic areas would be expected to be more at risk (Cabelli et al., 1979; Prüss, 1998).

The Guideline Values given in Table 4.6 were derived from key studies involving healthy adult bathers swimming in marine waters in a temperate climate. The Guidelines do not relate specifically to children, the elderly or immuno-compromised, who would have lower immunity and might require a greater degree of protection. If these are significant water user groups in an area, local authorities may want to adapt the Guidelines accordingly.

In areas with higher carriage rates or prevalence of diseases potentially transmitted through recreational-water contact, risks are likely to be greater (in response to greater numbers or different pathogens), and stricter standards may be judged appropriate by local authorities.

If the region is an international tourist area, other factors that need to be taken into consideration in applying the Guideline Values include the susceptibility of visiting populations to locally endemic disease, such as hepatitis A, as well as the risk of introduction of unfamiliar pathogens by visitors to the resident population.

The Guideline Values were derived from key studies in which the "exposure" was a minimum of 10 min bathing involving three immersions. They may therefore underestimate risk for activities involving higher risks of water ingestion or longer periods of water contact (although recreational-water users spending longer periods in the water may build up higher immunity levels). Recreational-water uses involving lesser degrees of water contact (such as windsurfing and slalom canoeing) will usually result in less water ingestion and thus may require less stringent Guideline Values.

When information on "typical" bathers (e.g., age, number of bathing events per bathing season per bather, average amount of water swallowed per bathing event) is known, local authorities can adapt the Guideline Values to their own circumstances, expressing the health risk in terms of the rate of illness affecting a "typical" bather over a fixed period of time.

Pathogens and faecal indicators are inactivated at different (and time-varying) rates dependent on physicochemical conditions. For example, although enterococci and *E. coli*/thermotolerant coliforms have generally similar rates of inactivation in fresh waters, enterococci are inactivated more quickly than coliforms in fresh waters containing high concentrations of humic compounds. Therefore, any one indicator is, at best, only an approximate index of pathogen removal efficacy in water (Davies-Colley et al., 2000).

This suggests that factors influencing faecal indicator die-off should be taken into consideration when applying the Guideline Values in Table 4.6, depending on local circumstances.

If an epidemiological study cannot be conducted, a QMRA assessment may be most appropriate where conditions differ significantly from those where the United Kingdom randomized controlled trials (i.e., the key studies used for guideline development; see section 4.3.1) were undertaken to support adaptation of Guideline Values to local circumstances. QMRA can be undertaken at significantly less cost than randomized controlled trials. Thus, a screening-level QMRA is recommended where differential persistence of faecal indicators and pathogens compared with the United Kingdom studies may occur. Examples of such circumstances include higher water temperatures, higher sunlight (UV) intensity and possibly different rates of microbial predation, along with different endemic disease(s) or where there is further treatment of sewage effluent prior to discharge.

As a crude reference point, exposure to temperate bathing waters has been considered tolerable when gastrointestinal illness is equivalent to that in the background unexposed population. Background rates have been given as, for example, 1.9–9.7% in marine water studies and 0.1–2.1% in freshwater studies (Cabelli et al., 1982; Kay et al., 1994; van Asperen et al., 1998).

Based on the randomized epidemiological studies of coastal bathers in the United Kingdom that are used as key studies in the development of Guideline Values (Kay et al., 1994), Wyer et al. (1999) provided an example of tolerable risk in terms of faecal indicator bacteria (faecal streptococci) equivalent to "background" or non-water-related gastrointestinal disease. Published or site-specific dose–response curves of the probability of illness over increasing indicator organism exposure can then be used in conjunction with the distribution of faecal indicator bacteria in bathing water to yield prospective microbiological criteria or actual expected disease burden at a particular recreational-water location (Box 4.6).

Box 4.6: Epidemiology-based tolerable bather risk in the United Kingdom and use of microbiological data to set a disease burden of less than 5%

Tolerable bather risk has been illustrated by its relationship to non-water-related (NWR) and person-to-person (PTP) risk factors for gastroenteritis (Wyer et al., 1999). For example, the PTP risk level may be considered unacceptable; that is, bathers might not be expected to encounter water quality with a probability of gastroenteritis greater than or equal to that associated with person-to-person transmission of illness. In other words, an excess probability of gastroenteritis of 0.34, which is equivalent to contact with ill family members, corresponds to a risk of swimming in water with 137 faecal streptococci/100 ml (Figure 4.3). The NWR risk level (excess probability of gastroenteritis of 0.17), which corresponds to a risk of swimming in water with 73 faecal streptococci/100 ml, is of use in defining exposure above and below that associated with "everyday activities" (such as consumption of fast food). Health authorities could therefore aim to limit equivalent exposure to between the NWR and PTP risk levels.

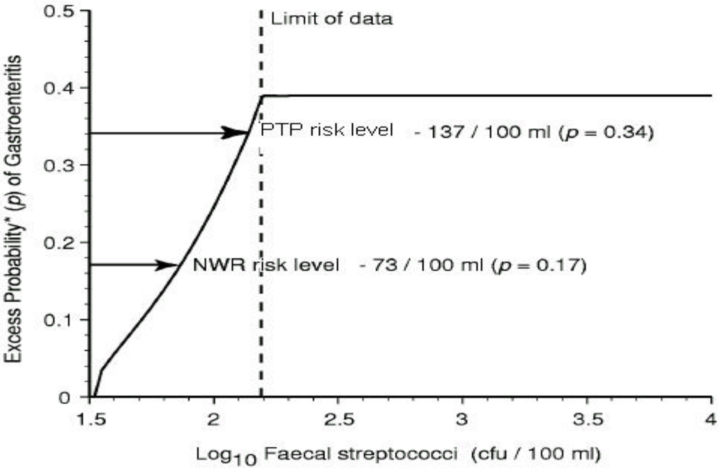


Fig. 4.3: Relationship between exposure to faecal streptococci (cfu/100 ml) at chest depth in marine water and the probability (p) of developing gastroenteritis

Probability of illness is assumed to remain constant at a level of 0.388, above 158 faecal streptococci/100 ml (limit of observational data).

Assume a regulator wishes to set a standard or guideline at a level of a specific indicator organism such that the probability of becoming ill does not exceed 0.05 (1 case/20 exposures).

Explanation of this process requires the introduction of a dose–response relationship illustrated graphically in Figure 4.3 and the PDF in Figure 4.4. This facilitates the disease burden calculation illustrated in Figure 4.4. Here, 1000 persons are assumed to be exposed; of these, 679 experience water quality unlikely to produce any health effect. Of the 321 who experience water quality that might make them ill, 71 become ill with symptoms of gastro-enteritis. Thus, the disease burden (or risk of illness) can be calculated for any recreational-water environment if a suitable dose–response curve is available and the indicator PDF can be drawn. Clearly, if the mean value of the PDF in Figure 4.4 moves to the right, i.e., the beach is more polluted, then the number of individuals exposed to water that might make them ill is increased.

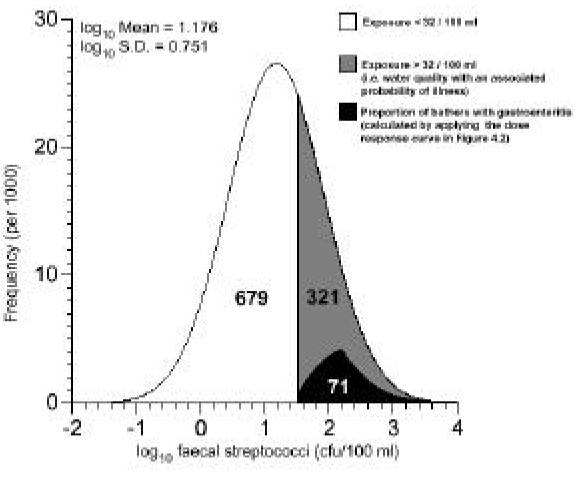


Fig. 4.4: Probability density function for faecal streptococci exposure based on the average United Kingdom bathing water quality data for 1988–1992 (total curve area adjusted to 1000, with the proportions of the area under the curve exceeding threshold risk factors) and integration to calculate total gastroenteritis, for the United Kingdom faecal streptococci exposure distribution

Reference values can be defined by adjusting the mean point of the PDF (i.e., the lognormal distribution of faecal indicator concentrations) until the excess disease burden is 1 case in 20 exposures. This can be repeated using any other desired probabilities of illness (i.e., 1 in 10, which approximates the 95th percentile faecal streptococci of <500/100 ml). The 95th percentile point of the PDF is then reported as the "standard value" associated with the accepted level of illness.

This method requires the regulator to assume constant variance (or spread) in the recreational-water environment PDF. This is required where a single regulatory compliance value is preferred.

The relationship between the 95th percentile value for faecal streptococci (standard deviation 0.8103) and the rate of illness in the exposed population is seen in Figure 4.5.

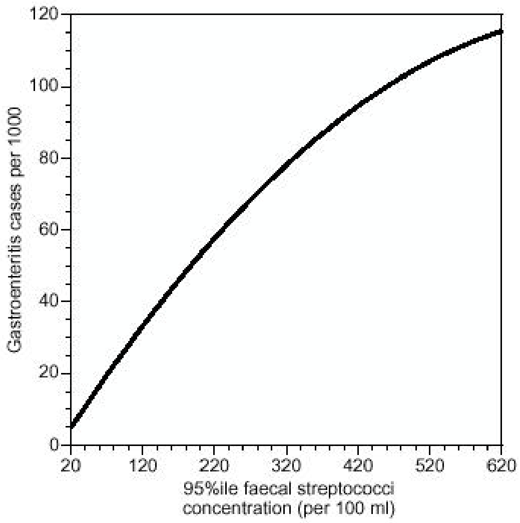


Fig. 4.5: Relationship between 95th percentile value for faecal streptococci and rate of illness in exposed population

The Guideline Values were derived using an average value for the standard deviation of the PDF for faecal streptococci of 0.8103, calculated from a survey of 11 000 European bathing waters (Kay et al., 1996). Local variations in the standard deviation would affect the shape of the PDF; higher standard deviation values would give a broader spread of values, while smaller standard deviation values would produce a more narrow spread of values. The effect of using a fixed standard deviation for all recreational-water environments is variable. If the true standard deviation for a beach were less than 0.8103, then reliance on faecal streptococci would tend to overestimate the health risk for people exposed above the threshold level, and vice versa (Box 4.6).

4.3.6 Regulatory parameters of importance

For any micro-organism to be used as a regulatory parameter of public health significance for recreational waters, it should:

- have a logical health basis;
- have adequate information available with which to derive Guideline Values (e.g., from epidemiological investigations);
- be sufficiently stable in water samples for meaningful results to be obtained from analyses;

- have a standard method for analysis;
- be cost-effective to test;
- make low demands on staff training; and
- require basic equipment that is readily available.

Microorganisms commonly used in regulation include the following:
- **Enterococci/faecal streptococci** meet all of the above.
- ***E. coli*** is intrinsically suitable for fresh waters but not marine water, and, as discussed in section 4.3.4, there are currently insufficient data to develop a Guideline Value with this parameter in fresh water.
- **Thermotolerant coliforms**, although a better indicator than total coliforms, include non-specific faecal indicators (e.g., *Klebsiella* from pulp and paper mill effluents). As there are no adequate studies on which to base guideline values, thermotolerant coliforms are unsuitable as regulatory parameters.
- **Total coliforms** are inadequate for the above criteria, in particular as they are not specific to faecal material.
- **Salmonellae** have been used for regulatory purposes. Their direct health role has not been supported by outbreak data. They are unlikely to contribute significantly to the transmission of disease via the recreational-water route because of their high infective dose and typically relatively low numbers in sewage, which, when combined with their rapid inactivation in waters, particularly seawaters, give limited biological plausibility.
- **Enteroviruses** have been used for regulatory purposes. They are costly to assay and require specialized methods that include a concentration step for their analysis, which are imprecise. Although enteroviruses are always present in sewage and there are standard methods, their numbers are variable and not related to health outcome (Fleisher et al., 1996a,b). Hence, there are insufficient data with which to develop a Guideline Value. Their direct health significance varies from negligible (e.g., vaccine strains) to very high.

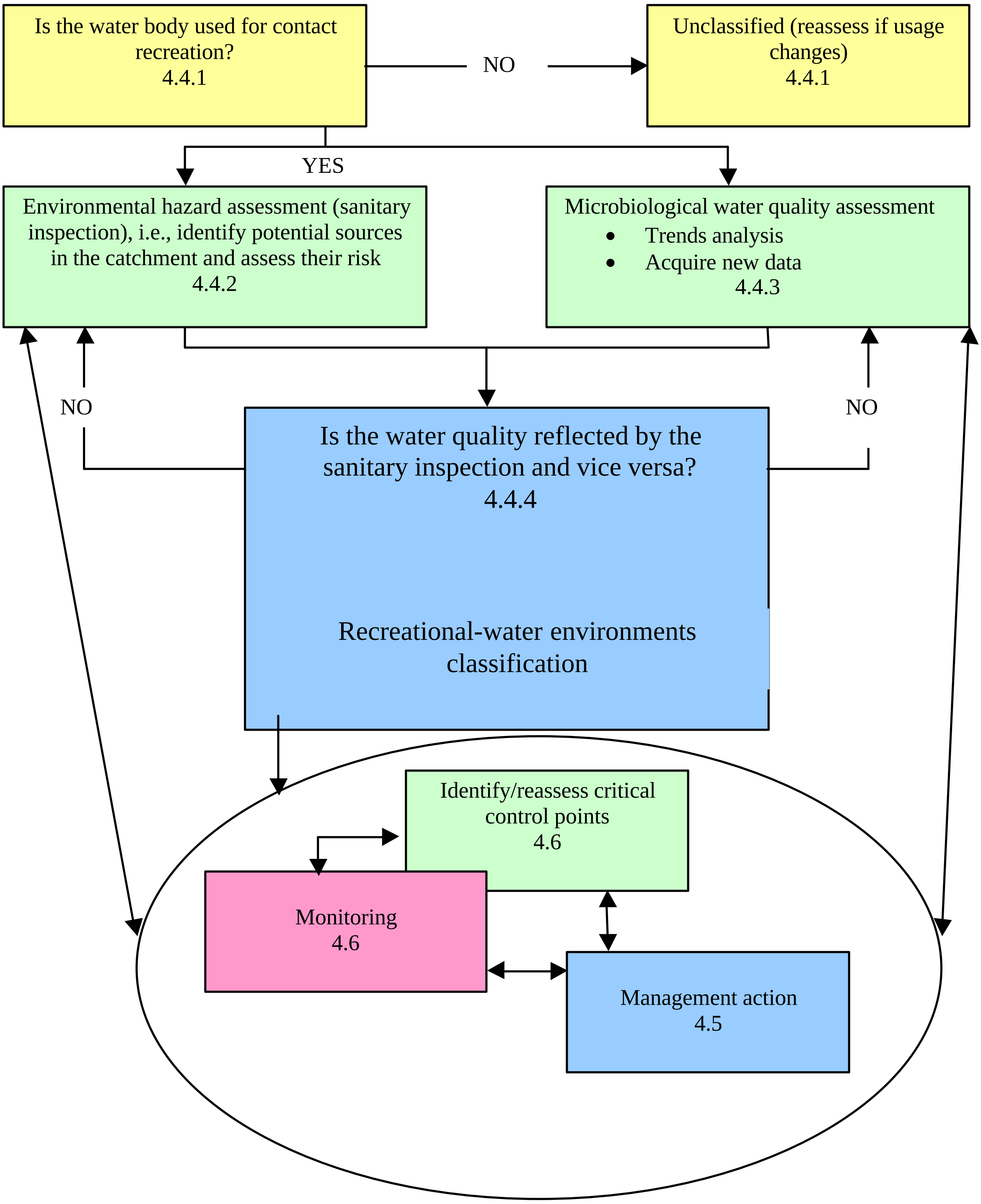
4.4 Assessing faecal contamination of recreational-water environments

In its simplest form, the three principal components required for any assessment are:
- initial classification based upon the combination of evidence for the degree of influence of (human) faecal material (by sanitary inspection of beach and water catchment) alongside counts of suitable faecal indicator bacteria (a microbiological quality assessment);
- identification of factors likely to influence faecal contamination (such as nearby rivers or stormwater outlets that may be influenced by rainfall events or sewer overflows); and
- the possibility of "reclassifying" a recreational-water environment (either better or worse) if a significant change in catchment management reduces or increases human exposure to microbial risk (at times or in places).

The results of this assessment will be twofold: a classification of the recreational-water environment that is based on long-term analysis of data, and immediate actions to reduce exposure over a much shorter time frame (i.e., hours or days).

An outline of the overall approach for recreational waters is presented in Figure 4.6.

Fig. 4.6: Framework for assessing recreational-water environments (numbers refer to sections in chapter)



The approach outlined in Figure 4.6 leads to a classification scheme through which a recreational-water environment would be assigned to a class (i.e., very poor, poor, fair, good or very good) based upon health risk. Classification of a recreational-water environment is based upon both environmental hazard assessment (sanitary survey) and microbiological water quality assessment components. The end result will be a classification matrix of the type shown in Table 4.8.

Table 4.8: Classification matrix for recreational-water environments

<table>
<tr><td colspan="2" rowspan="2"></td><td colspan="4">Microbiological Assessment Category
(indicator counts)</td><td rowspan="7">Exceptional circumstances</td></tr>
<tr><td>A
<40</td><td>B
40–200</td><td>C
201–500</td><td>D
>500</td></tr>
<tr><td rowspan="5">Sanitary Inspection Category (susceptibility to faecal influence)</td><td>Very low</td><td>Very good</td><td>Very good</td><td>Follow up+</td><td>Follow up+</td></tr>
<tr><td>Low</td><td>Very good</td><td>Good</td><td>Fair</td><td>Follow up+</td></tr>
<tr><td>Moderate</td><td>Follow up*</td><td>Good</td><td>Fair</td><td>Poor</td></tr>
<tr><td>High</td><td>Follow up*</td><td>Follow up*</td><td>Poor</td><td>Very poor</td></tr>
<tr><td>Very high</td><td>Follow up*</td><td>Follow up*</td><td>Poor</td><td>Very poor</td></tr>
<tr><td colspan="7">Exceptional circumstances</td></tr>
</table>

Notes:

1. * indicates unexpected results requiring investigation (see sections 4.3.2 and 4.4.4)
   + implies non-sewage sources of faecal indicators (e.g., livestock), and this should be verified
2. In certain circumstances, there may be a risk of transmission of pathogens associated with more severe health effects through recreational-water use. The human health risk depends greatly upon specific (often local) circumstances. Public health authorities should be engaged in the identification and interpretation of such conditions (see section 4.4.4). The need for reassessment of risk may be triggered by various factors. Accounting for such risks may lead to the reclassification of a location, unless recreational-user access can be controlled.
3. Exceptional circumstances (see section 4.4.4) relate to known periods of higher risk, such as during an outbreak with a pathogen that may be waterborne, trunk sewer/combined sewer rupture in the beach catchment, etc. Under such circumstances, the classification matrix would be superseded.

The classification scheme enables local management to respond to sporadic or limited areas of pollution and thereby upgrade a recreational-water environment's classification. This is achieved by discounting of data derived from periods of time in which actions to discourage recreational-water use were deployed and shown to be effective (section 4.4.3). The classification scheme (as opposed to a pass/fail approach) therefore provides incentive to local management actions as well as to pollution abatement. It further provides a generic statement of the level of risk supportive of informed personal choice and indicates the principal management and monitoring actions likely to be appropriate.

The scheme focuses on faecal contamination from humans, with lesser importance placed on faecal contamination from other sources, such as drainage from areas of animal pasture and intensive livestock rearing, the presence of gulls or the use of the beach for dogs or horses. However, in general, due to the "species barrier," the density of pathogens of public health importance is generally assumed to be less in aggregate in animal excreta than in human excreta and may therefore represent a significantly lower risk to human health. As a result, the use of faecal indicator bacteria alone as an index of risk to human health may significantly overestimate risks where the indicators derive from sources other than human excreta. Nevertheless, the human health risk associated with pollution of recreational waters from animal excreta is not zero, and some pathogens, such as *Cryptosporidium parvum* and *E. coli* O157:H7, can be transmitted through this route.

4.4.1 Identification of recreational-water environments

Recreational uses of inland and marine waters are increasing in many countries world-wide. These uses range from total-immersion sports, such as swimming, surfing and slalom canoeing, to non-contact sports, such as fishing, walking, bird-watching and picnicking. The water recreation categories of non-contact, incidental contact and whole-body contact are described in more detail in chapter 1.

This chapter relates to recreational-water activities where whole-body contact takes place (i.e., those in which there is a meaningful risk of swallowing water). If the water body is not used for whole-body contact recreation, then the recreational-water environment would remain unclassified until such time as its usage changes (or credible epidemiological data are available for low-contact recreational activities). Where such contact does occur, an environmental hazard assessment and microbiological water quality assessment should be performed.

4.4.2 Environmental hazard assessment

The three most important sources of human faecal contamination of recreational-water environments for public health purposes are sewage; riverine discharges, where the river is a receiving water for sewage discharges and either is used directly for recreation or discharges near a coastal or lake area used for recreation; and bather contamination, including excreta. Information to be collected during sanitary inspections should at least cover the following:

- Sewage outfalls, combined sewer overflows, stormwater outfalls
  - Presence/absence (each is considered to be of equal human faecal load unless otherwise determined)
  - Type of sewage treatment
  - Effectiveness of outfall type

- Riverine discharges
  - Presence/absence
  - Type of sewage treatment
  - Population size from which sewage originates
  - River flow in dry season

- Bather shedding
  - Bather density
  - Dilution (mixing of water in bathing area)

Information available will vary based on the current regulatory criteria and compliance status of each recreational-water environment.

Additional information that may assist in assessing the safety of recreational waters and in controlling associated risks is often readily available and may concern, for example:

- rainfall (duration and quantity);
- wind (speed and direction);
- tides and currents; and
- coastal physiography.

Rainfall has a very important influence on indicator densities in recreational waters. Indicator densities in recreational waters can be increased to high levels because treatment plants are overwhelmed, causing sewage to bypass treatment, or because of animal wastes washed from forest land, pasture land and urban settings. Re-suspension of sediment-trapped pathogens is another factor influenced by rainfall, particularly in freshwater river catchments. In all these cases, the effect of rainfall on recreational-water quality can be highly variable, yet characteristic for each recreational-water area.

The relative risks to human health through direct sewage discharge, riverine discharge contaminated with sewage and bather contamination have been ranked in this chapter (see below). In doing so, account is taken of the likelihood of human exposure and the degree of treatment of sewage. In taking sewage and riverine discharges to recreational areas into consideration, account is also taken of the pollutant load, using population as an index. Information on local circumstances needs to be taken into account and may lead to variation. For example, sewage being discharged in an estuary with small tidal interchanges may have an effect different from that of the same quantity of sewage discharged in an estuary with large tidal interchanges. Similarly, a river discharging in an enclosed bay can be considered of a higher risk than one discharging directly into the open sea.

While in many circumstances several contamination sources would be significant at a single location, a recreational-water environment may be best categorized according to the single most significant source of pollution.

The following sections assist in the placement of recreational-water environments in an appropriate sanitary inspection category indicative of susceptibility to faecal influence, but cannot totally remove regional subjectivity in this classification.

1) Sewage discharges (including combined sewer overflows and stormwater discharges)

Sewage-related risk is a combination of the likelihood of pollution and, where pollution occurs, the degree of inactivation through treatment.

Sewage discharges, or outfalls, may be readily classified into three principal types:

- those where the discharge is directly onto the beach (above low water level in tidal areas);
- those where discharge is through “short” outfalls, where discharge is into the water but sewage-polluted water is likely to contaminate the recreational-water area; and

- those where discharge is through "long" sea outfalls, where the sewage is diluted and dispersed and the design criteria for the outfall should ensure that sewage does not pollute recreational-water areas.

While the terms "short" and "long" are often used, length is generally less important than proper location and effective diffusion, which will ensure that pollution is unlikely to reach recreational areas.

Direct discharge of crude, untreated sewage (for instance, through short outfalls or combined sewer overflows, which contain a mixture of raw sewage and stormwater) into recreational areas presents a serious risk to public health. Public health authorities should take measures to protect public health where this occurs and co-operate with appropriate authorities to encourage cessation of this practice. Recreational-water users, other than swimmers, may venture into areas adjacent to effluent discharges where water quality has not traditionally been monitored but where health risks may be significantly elevated. These exposures also present a potentially significant risk for the population concerned and should be managed through appropriate measures.

For short outfalls, the relative risk is increased based upon the size of the contributing population.

An effective outfall is assumed to be properly designed, with sufficient length and diffuser discharge depth to ensure low probability of the sewage plume reaching the designated beach zone.

In public health terms, it is generally assumed that the processes of dispersion, dilution, sedimentation and inactivation (through isolation, predation, natural die-off, etc.) following discharge into the marine (or freshwater) environment from a piped outfall will lead to a certain degree of safety regarding public health. A number of confounding factors reduce the efficiency of this in practice. Most important among these are those that lead to the rapid movement of sewage into recreational areas. For example, where sewage is relatively warm and of low salinity when compared with the receiving water, it may mix poorly and form a floating "slick." Such slicks should not form where properly designed and operated diffusers are in place on the outfall. Where they form, they will be readily influenced by wind and may therefore pollute (even distant) recreational-water environments severely. While not providing long-term security for public health, periods of high risk (such as during onshore winds) may be recognized on such beaches and action (advisory notices, zoning or banning of bathing) taken as appropriate. Coastal currents and tides may also give rise to similar problems and may also be recognized and dealt with in a similar manner.

Control of sewage pollution by holding sewage in storage for varying periods of time is practised in some countries. Where sewage is retained throughout the bathing season, water users are effectively isolated from the source of pollution. Such an approach is of limited applicability for practical reasons and will be fully effective only where there is a strict cut-off in recreational activity at the end of the bathing season. The efficacy of shorter-term retention — such as retention during the day and discharge at night — is less certain and is strongly influenced by the nature of the discharge and environmental factors as discussed above. The health risks associated with beach sand and littoral zone sediments remain poorly understood.

The degree of treatment applied to sewage varies widely and includes:

- no treatment (discharge of raw, untreated sewage);
- “preliminary” treatment (screening with milli- or micro-screens to remove large solids);
- conventional primary treatment (physical sedimentation or settling);
- conventional secondary treatment (primary sedimentation plus high-rate biological processes, such as trickling filter/activated sludge);
- secondary treatment plus disinfection (chlorination, ultraviolet or ozone);
- conventional tertiary treatment (advanced wastewater treatment, including primary sedimentation, secondary treatment plus, for example, coagulation-sand filtration, ultraviolet, micro-filtration);
- tertiary treatment plus disinfection; and
- lagooning (low-rate biological treatment).

Of these, lagooning, conventional primary plus secondary treatment, tertiary treatment and disinfection will effect a significant reduction in indicator and/or pathogen contamination. It should, however, be noted that some treatments, notably disinfection (in particular, chlorination), may affect the validity of the assessment of risk due to possible differential attenuation between indicator and pathogenic organisms within the treatment systems, leading to underestimates of risk, particularly with disinfection-resistant enteric viruses and *Cryptosporidium.*

Urban stormwater runoff and outputs from combined sewer overflows are included within the scheme under the category of direct beach outfalls. Septic systems and stormwater/combined sewers are assumed to be equivalent to primary treatment. The classification is based upon a qualitative assessment of risk of contact/exposure under “normal” conditions with respect to the operation of sewage treatment works, hydro-meteorological and oceanographic conditions. The potential risk to human health through exposure to sewage through outfalls can be categorized as shown in Table 4.9.

Table 4.9: Relative risk potential to human health through exposure to sewage through outfalls

Treatment	Discharge type		
	Directly on beach	Short outfall[a]	Effective outfall[b]
None[c]	Very high	High	NA
Preliminary	Very high	High	Low
Primary (including septic tanks)	Very high	High	Low
Secondary	High	High	Low
Secondary plus disinfection	Moderate	Moderate	Very low
Tertiary	Moderate	Moderate	Very low
Tertiary plus disinfection	Very low	Very low	Very low
Lagoons	High	High	Low

[a] The relative risk is modified by population size. Relative risk is increased for discharges from large populations and decreased for discharges from small populations.

[b] This assumes that the design capacity has not been exceeded and that climatic and oceanic extreme conditions are considered in the design objective (i.e., no sewage on the beach zone).

[c] Includes combined sewer overflows.

2) Riverine discharges

Rivers discharging in coastal areas may carry a heavy load of microorganisms from diverse sources, including municipal sewage (treated or otherwise) and animal husbandry. Following rainfall, microbial loads may be significantly increased due to surface runoff, urban stormwater overflows and resuspension of sediments. Coastal pollution levels may therefore be elevated following rainfall, and periods of high risk in some coastal areas may be found to correlate with such climatological data. Once the hazard is recognized and characterized, simple advisory measures may be taken prospectively to alert water users of such risks and/or prevent recreational use during such periods.

Recreational areas on rivers will be subject to influences similar to those indicated above. In addition, where water flow is managed, either for recreation (such as where water is impounded before discharge) or for other purposes, the act of impoundment and discharge may itself lead to elevated microbial levels through re-suspension of sediment. Rivers are commonly receiving environments for sewage effluents following secondary or biological treatment in some countries. Much lower levels of effluent dilution may occur in riverine environments than in their coastal equivalents, and differential pathogen–indicator organism relationships may exist between saline and non-saline waters. The balance of evidence suggests that, under many circumstances, the same level of faecal indicator bacteria in freshwater and marine environments may imply a greater health risk in marine environments (see section 4.3.4).

Riverine discharges may be categorized with respect to the sewage effluent load and the degree of dilution in a manner similar to that described in Table 4.10.

Table 4.10: Relative risk potential to human health through exposure to sewage through riverine flow and discharge

Dilution effect[a,b]	Treatment level				
	None	Primary	Secondary	Secondary plus disinfection	Lagoon
High population with low river flow	Very high	Very high	High	Low	Moderate
Low population with low river flow	Very high	High	Moderate	Very low	Moderate
Medium population with medium river flow	High	Moderate	Low	Very low	Low
High population with high river flow	High	Moderate	Low	Very low	Low
Low population with high river flow	High	Moderate	Very low	Very low	Very low

[a] The population factor includes all the population upstream from the recreational-water environment to be classified and assumes no in-stream reduction in hazard factor used to classify the recreational-water environment.

[b] Stream flow is the 10% flow during the period of active beach use. Stream flow assumes no dispersion plug flow conditions to the beach.

Sheltered coastal areas such as closed bays often attract recreational-water users and may present special problems. Small volume, low circulation and low water exchange rates often occur in such bodies of water. The indicator and pathogen concentrations in the water may be strongly influenced by slow exchange rates, effectively "trapping" sewage effluents for relatively long periods of time. These small bodies of water behave similarly to swimming pools and may be managed as described in Volume 2 of the *Guidelines for Safe Recreational-water Environments.*

3) Bathers

Bathers themselves can influence water quality directly. This is most commonly seen as microbial build-up during the day, such that peak levels are reached by the afternoon. In circumstances of limited dispersion, bather-derived faecal pollution may present a significant health risk, as evidenced by several outbreaks of disease (see section 4.1) and by analogy to swimming pools and spas (see Volume 2 of the Guidelines). There is insufficient evidence to judge the contribution that bather-derived pollution makes in other circumstances.

The two principal factors of importance in relation to bathers are bather density and degree of dilution (Table 4.11). Low dilution is assumed to represent no water movement (e.g., lakes, lagoons, coastal embayments). The likelihood of bathers defecating or urinating into the water is substantially increased if toilet facilities are not readily available. Under high bather density, the classification should therefore be increased to the next higher class if no sanitary facilities are available at the beach.

Table 4.11: Relative risk potential to human health through exposure to sewage from bathers

Bather shedding	Category
High bather density, high dilution[a]	Low
Low bather density, high dilution	Very low
High bather density, low dilution[a,b]	Moderate
Low bather density, low dilution[b]	Low

[a] Move to next higher category if no sanitary facilities available at beach site.
[b] If no water movement.

Sheltered coastal areas and shallow lakes may also be subject to accumulation of sediments, which may be associated with high microbial loads that may be re-suspended by water users and/or rainfall events. The health risks associated with re-suspended sediments remain poorly understood, but should be noted as a potential risk during sanitary surveys.

4) Follow-up investigation: source identification

Follow-up analyses are recommended when the enterococci counts are high but the environmental hazard assessment suggests low sanitary impact, or vice versa (Table 4.8). The possibility of simple error in either analysis or sanitary inspection should always be considered and investigated. A primary role of the follow-up is to help identify the source of the faecal pollution, thereby assisting in the assessment and management of faecal contamination in recreational-water environments.

A range of parameters is available that suit investigations associated with sanitary surveys and follow-up investigations (Bartram & Rees, 2000). For example, *Clostridium* *perfringens* is used in Hawaii for regulatory use, where *E. coli* and enterococci have been suggested to grow in tropical soils and not directly indicate faecal pollution (Fujioka et al., 1999). *C. perfringens* is most appropriately used as a secondary parameter indicating the likely presence of human sewage when enterococci suggest (warm-blooded animal) faecal pollution. Care is needed, however, as these spore-forming bacteria are very persistent and may not indicate faecal pollution where pathogens are still infectious. A range of mammals also excrete *C. perfringens*, although generally at significantly lower numbers than humans, dogs excepted (Leeming et al., 1998). Sulfite-reducing clostridia have been assayed as a surrogate for *C. perfringens*, but lack specificity as a faecal indicator. A second, although more expensive, means of differentiating faecal sources may be provided by the analysis of faecal sterols (Bartram & Rees, 2000).

High counts of enterococci in the apparent absence of sanitary problems could imply environmental growth of enterococci, but is much more likely to result from hidden sewer/septic or stormwater leaks (checked by the presence of *C. perfringens*). Detection of relatively high concentrations of nutrients (nitrogen or phosphorus) in the recreational/catchment waters may also support the presence of wastewater input. Furthermore, aeromonads may provide information on the state of eutrophication of waters (Bahlaoui et al., 1997), but their role as primary pathogens causing gastrointestinal or respiratory illness via recreational exposures is unclear.

Specific pathogens or other microorganisms may be genotyped by various molecular methods (e.g., ribotyping, polymerase chain reaction-pulse field gel electrophoresis) to provide subspecies resolution of strains, which may provide assistance in faecal source identification and molecular epidemiology. Such methods are currently being applied to identify source(s) of waterborne cryptosporidiosis (Spano et al., 1998) and swimmer-acquired *E. coli* O157:H7 (Ackman et al., 1997).

4.4.3 Microbiological water quality assessment

The various stages involved in an assessment of the microbiological quality of a recreational-water environment are briefly summarized as follows:

- **Stage 1:** Initial sampling to determine whether significant spatial variation exists. Sampling at spatially separated sampling sites should be carried out at the start of the bathing season on different days. Timing of samples should take into account the likely period of maximum contamination from local sewage discharges and maximum bather shedding (e.g., the afternoon or day after peak bather numbers).
- **Stage 2:** Assessment of spatial variation based on data from the above.
- **Stage 3:** Intensive sampling (if no significant spatial variation) and assessment of results. If there is no evidence of spatial variation, the initial classification is determined from results of the sanitary survey and microbiological quality assessment. Microbiological quality for all recreational waters is classified into four categories by the 95th percentile of the enterococci distribution as per Table 4.6.
- **Stage 4:** Definition, separate assessment and management of impacted areas if spatial variation evident at Stage 2.
- **Stage 5:** Confirmatory monitoring in the following year, using a reduced sampling regime and repeat of the sanitary inspection. If the classification is very good or very poor, less frequent monitoring can be justified if appropriate management actions are in place.

The sampling programme should be representative of the range of conditions in the recreational-water environment while it is being used. When determining recreational-water classification, all results from that water, on days when the recreational-water area was open to the public, must be used. For example, it is not acceptable to resample should an unexpectedly high result be obtained and use the resample, but not the original sample, for classification purposes. On the other hand, reactive samples that are taken following an adverse event to investigate the full impact of that event on the beach need not be included within the analysis. Providing bathing is demonstrated to have been effectively discouraged, the microbiological results on such days also need not be included amongst the data used to classify the recreational-water environment.

A most important issue in assessing the microbiological quality of waters is that of collecting sufficient numbers of samples so as to make an appropriate estimation of the likely densities to which recreational-water users are exposed. Previous recommendations based on 20 or fewer samples are considered not statistically representative, given the usual variation in microbiological faecal indicators. When 20 or so samples are analysed, the precision of the estimate of the 95^{th} percentile is low. Increasing sample numbers, for instance towards 100 samples, would increase precision.

The number of results available can be increased significantly — and with no additional cost — by pooling data from multiple years. This practice is justified unless there is reason to believe that local (pollution) conditions have changed, causing the results to deviate from established behaviour. For practical purposes, it is suggested that data on at least 100 samples from a 5-year period and a rolling 5-year data set be used for microbiological assessment purposes. In many situations, a much shorter period will be required, where, for example, more extensive sampling is undertaken by the competent authorities. As well, fewer samples may be required — for instance, where the water quality is very poor (section 4.4.4).

Various indicator bacteria, including *E. coli*, thermotolerant coliforms and enterococci/faecal streptococci, are used for the monitoring of recreational waters. Several methods are available for estimating bacteriological numbers, in particular faecal indicator numbers, at recreational-water areas (e.g., ISO, 1975, 1990, 1995, 1998a, 1998a; APHA, 1989).

Where a change between indicators is made, limited amounts of data may be available in the initial years of implementation. In order to overcome this, correction factors appropriate to local conditions can be applied to historical records. Such conversion factors would normally be driven by the results of local analyses.

For many locations, there will be a large amount of historical data available that can be used for preliminary recreational-water environment classification. If these historic data include analysis for enterococci, there will be no problem in using this data. However, many recreational-water environment managers will have data based only on coliform and faecal coliform counts. There is no exact relationship between faecal streptococci and *E. coli* counts (Figure 4.7). Nevertheless, a relationship can be expressed by the below equation that may assist in interpreting historical data:

$$\text{log faecal coliform count} = 1.028 + 0.601 * \text{log faecal streptococcal count}$$

Consequently, counts of =100 faecal coliforms/100 ml can equate to =40 faecal streptococci/100 ml, =250 faecal coliforms/100 ml to =200 faecal streptococci/100 ml and =450 faecal coliforms/100 ml to =500 faecal streptococci/100 ml. However, this equivalence is not exact, and, if possible, local recreational-water environment managers should define the relationship that exists in their own waters.

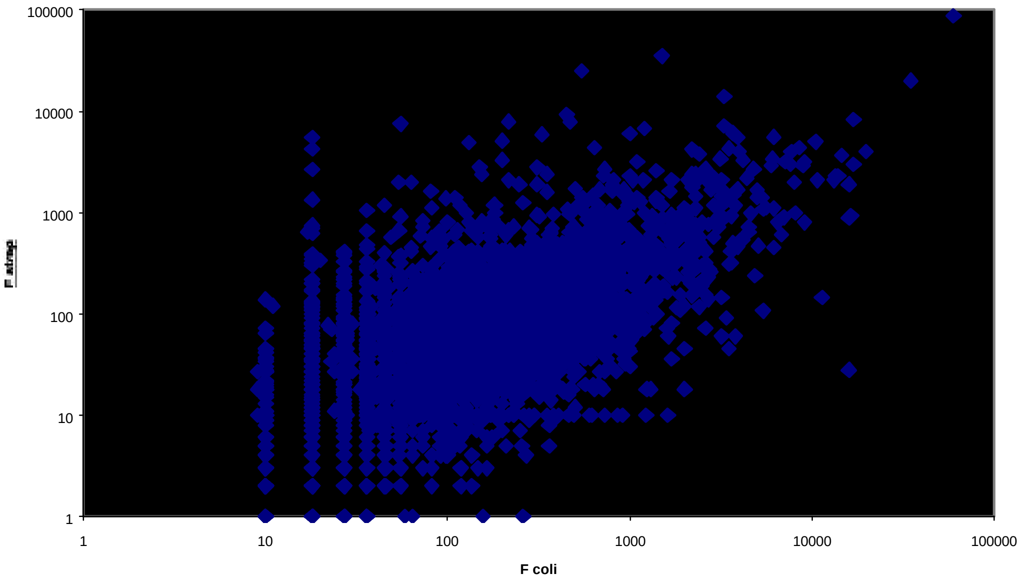


Figure 4.7. Relationship between faecal coliform and faecal streptococcal counts in United Kingdom bathing waters after censoring zero values

4.4.4 Classification of recreational-water environments

1) Initial classification

The outcome of the environmental hazard assessment and the microbiological water quality assessment is a five-level classification for recreational-water environments — very good, good, fair, poor and very poor (see Table 4.8 above). If the assessment of spatial variation shows that higher microbial contamination levels are limited to only part of a recreational-water environment, separate assessment and management are required.

In cases where multiple sources of contamination exist, the single most significant source is used to determine the susceptibility to faecal influence. Relative contributions from riverine discharges and bather densities need to be scaled based on local knowledge of hydrological conditions.

Should the results obtained identify events (such as rainfall) that predict when bathing water quality will deteriorate, it may be possible to post the beach with an advisory notice discouraging bathing following such events. Samples taken while such bathing waters are posted should not be included in the classification. However, the management must show that they have actively discouraged bathing and their efforts in this regard have been largely successful (see “Reclassification” below).

A case study is provided in Box 4.7 to illustrate the approach.

Box 4.7: Case study

The following is an example of how to apply the guideline approach to a real seawater recreational-water area. Historical data for the recreational water were available; therefore, the last 5 years of data (in this case, more than 20 samples per year) were used to provide the microbiological assessment.

NAME OF RECREATIONAL-WATER AREA: ________________________________

1 SANITARY INSPECTION

Sewage discharges

Outfalls	Present?	If present:		
	Y / N	Type of sewage treatment (see note 1)	Type of outfall (see note 2)	Category
Sewage outfalls	Y	primary	effective	low
Combined sewer overflows	N			-
Stormwater	Y		direct	very high, but only during events; as signage successfully warns users not to swim during rainfall and up to 2 days after heavy rainfall, classified as low

Note 1: Type of sewage treatment
none = raw sewage
preliminary = filtration with milli- or micro-screens
primary = physical sedimentation (includes septic tank systems)
secondary = primary + trickling filter/activated sludge
secondary + disinfection
tertiary = secondary + coagulation-sand filtration
tertiary + disinfection
lagoons = low-rate biological treatment

Note 2: Type of outfall
direct = on beach
short = within inter-tidal zone, significant probability of sewage plume reaching beach
effective = sufficient length and depth to ensure low probability of sewage plume reaching beach

Riverine discharges
Riverine discharges on beach (where river receives sewage discharge)

Present? Y / N	If present: Size of population from which sewage effluent originates	Type of sewage treatment	River flow during dry season (high, medium, low)
N			-

Bather shedding

Bather density in swimming season (high, low)	Dilution (low if beach has restricted water flow — lakes, lagoons, enclosed inlets — otherwise high)
High	high

Are there toilet facilities on the beach (Y/N)? Y

Overall category	**Low, but very high during rainfall-driven events**

2 PHYSICAL CHARACTERISTICS OF THE BEACH AND DATA ON MICROBIOLOGICAL WATER QUALITY

a) Provide a scale sketch map of the beach showing location of sampling points and swimming areas.

The beach is 800 m long. There are several stormwater drains discharging to the beach.

b) Describe the current monitoring programme for microbiological quality:
- sample volume and micro-organisms tested
- sampling schedule:
  months of the year when sampling occurs
  frequency of sampling during this period
- number and location of sampling points

Current routine monitoring programme for microbiological quality:

Sample volume = 100 ml
Tested for thermotolerant coliforms and enterococci
Sampling schedule: approx every 6 days
Sampling points: 1

c) Summarize data file(s) covering at least 1 year of monitoring (or 100 samples) for faecal indicator organisms — 100 raw numbers are needed in order to calculate 95th percentiles. Preferably these should be the most recent data available.

Microbiological assessment under new WHO Guidelines:
5-year data set

95th percentile = 276 enterococci/100 ml
Microbiological Assessment Category = C

d) Describe the current regulations governing water quality at this beach — on what basis is “acceptable” microbiological quality for recreational-water use defined.

Current regulations governing water quality at this beach:
Primary indicator

Median ≤150 thermo-tolerant coliforms/100 ml for 5 samples at regular intervals not exceeding 1 month and second highest sample ≤600/100 ml

Secondary indicator

median ≤35 enterococci/100 ml for 5 samples at regular intervals not exceeding 1 month and second highest sample <100 enterococci/100 ml

Compliance is currently determined from the rolling 30-day average median and the rolling 30-day average of the second highest samples.

Compliance with previous microbiological criteria:
1998/99 summer season - compliance was 100% for thermo-tolerant (faecal) coliforms and 48% for enterococci.
1999/00 summer season - compliance was 100% for both indicators.

3 IDENTIFICATION OF FACTORS LIKELY TO YIELD FAECAL CONTAMINATION EVENTS

Recreational-water areas with successful management control of event periods should be separated from “normal” periods in recreational-water environment classification. Examples of management controls include signage or closure following events, such as rainfall-induced faecal contamination via river, stormwater and sewer overflows.

Management control present? Y / N	Evidence for beach management during faecal event periods (signage, media release)
Y	Signage successfully warns users not to swim during rainfall and up to 2 days after heavy rainfall

4 COMBINED SANITARY AND MICROBIOLOGICAL ASSESSMENT UNDER NEW WHO GUIDELINES

This beach is rated as “fair”:
Sanitary Inspection Category - Low
Microbiological Assessment Category - C

		Microbiological Assessment Category (indicator counts)				**Exceptional circumstances**
		A =40	**B** 41–200	**C** 201–500	**D** >500	
Sanitary Inspection Category (susceptibility to faecal influence)	**Very low**	Very good	Very good	Follow up^{+}	Follow up^{+}	
	Low	Very good	Good	Fair	Follow up^{+}	
	Moderate	Follow up*	Good	Fair	Poor	
	High	Follow up*	Follow up*	Poor	Very poor	
	Very high	Follow up*	Follow up*	Poor	Very poor	
Exceptional circumstances						

This ocean bathing area is classified as “fair,” despite management interventions to overcome the effects of rainfall events. If other periods contribute significant stormwater outfall pollution, which could be reduced or eliminated, that could result in reclassification to “good” or even “very good.” Hence, investigations need to explore if there is chronic stormwater flow and, indeed, if the stormwaters contain a high concentration of sewage. Risk reduction management may then be best directed to rectifying the faecal load to stormwater flow in the medium term.

2) *Exceptional circumstances*

In certain circumstances there may be a risk of transmission of pathogens associated with more severe health effects (such as infectious hepatitis or typhoid fever) through recreational-water use, or circumstances may indicate that there is a greater health risk.

The pathogens of concern typically have a low infectious dose. The risks associated with such pathogens may not be adequately reflected in the water quality categories described in Table 4.6, nor may the general control measures described here be adequate. Very limited evidence exists regarding safety, and public health authorities should be alert to such hazards where exposure may occur.

These human health risks depend greatly upon specific (often local) circumstances. Such circumstances include, for example, the susceptibility of visiting populations to locally endemic disease and the risk of introduction of unfamiliar pathogens by visitors to the resident population, as well as the nature and effectiveness of control measures, such as sewage treatment.

Population groups that may be at higher risk of disease may include the young, the elderly and the immuno-compromised. If such groups are significant water users, then this should be taken into account in risk assessment.

While no general guidance concerning these risks is provided here (for instance, as Guideline Values), there is a need to make provisions to enable their identification and management. Public health authorities should be engaged in the definition of standards and of water qualities relevant to specific circumstances. This will normally require provision for responsibility and authority to act in response to such risks/circumstances.

While interpretation of the public health significance of specific conditions will generally require the participation of the public health authority, initial identification of a potential problem may arise from (human) disease surveillance, authorities responsible for wastewater treatment and management or veterinary authorities. Furthermore, while the public health authorities bear responsibility for assessing public health risk, determining and implementing appropriate actions will require inter-sectoral action and will often include also local government, facility operators, user groups and so on.

Examples of relevance may include:

- ***E. coli* O157**. This pathogen arises primarily from livestock rearing. It has a low infectious dose, causes a severe dysentery-like illness and may be associated with haemolytic uraemic syndrome. The disease has a significant mortality and morbidity. To date there has been one documented report of transmission of *E. coli* O157 through recreational waters (Ackman et al., 1997). In catchment areas impacted by livestock excreta, there is a potential risk of transmission to humans. The carriage rate among cattle varies from 1 to 15% in the United Kingdom, and higher rates have been reported in the USA (Jones, 1999). Where effluent from dairies or intensive grazing is a significant proportion of the faecal load in recreational waters, public health authorities should be informed.
- **Enteric hepatitis viruses** (HAV, HEV). Infection with HAV is typically mild when first acquired early in life but is severe when first acquired in adulthood. It is a recognized problem among susceptible travellers to areas of high endemicity, and, although there are no documented cases of transmission through bathing, such transmission is biologically plausible.
- **Typhoid and paratyphoid (enteric) fevers**. *Salmonella typhi* and *S. paratyphi*, the causative agents of typhoid and paratyphoid fevers, respectively, can be transmitted by the waterborne route. *S. typhi* has a low infectious dose. There has been a

documented association of *S. paratyphi* transmission with recreational-water use (Public Health Laboratory Service, 1959). The only source of the agents is human excreta; therefore, in areas with outbreaks or high endemicity of the diseases, a risk of transmission exists. The one documented study found no transmission in water containing less than 10 000 total coliforms (approximately equivalent to 1000 faecal streptococci/enterococci).

- **Outbreaks of disease among human populations**. When there is an outbreak of certain diseases among a population, there may be a significant increase in the occurrence of the causative agent in the faeces of the affected person and in turn among sewage and sewage-polluted bathing waters. However, in many circumstances, the overall public health risk is modest because the number of infected/excreting persons is a small proportion of the total.
- **Cholera**. While the infectious dose for cholera is generally considered high, it is variable, and the causative agent may be excreted in large numbers when an outbreak occurs. The causative bacteria, *Vibrio cholerae*, may also establish itself in local ecosystems in some conditions, and the significance of this for human health is poorly understood. Where *V. cholerae* occurs, the significance of this for human health should be specifically assessed.

Exceptional circumstances requiring re-evaluation of risk also include those circumstances leading to increased pollution and, by inference, increased risk to bathers. Thus, failure in sewage treatment or fracture of a long sea outfall would imply the need to immediately reassess safety. Results of microbiological testing should be monitored on a "control chart," and deviation from established behaviour should be one trigger for investigation and assessment of public health risk.

3) Resolution of anomalous results

The flow chart in Figure 4.6 presents a schema in which the sanitary inspection and water quality data inspection are conducted, where possible, in parallel. Where these two risk assessments result in incongruent categorization according to the classification matrix (as indicated by "follow up" in Table 4.8), further assessment will be required. This could include reassessing the sanitary inspection (i.e., identifying further potential sources in the catchment and assessing their risk; see section 4.4.2) and additional analysis of water quality, with specific consideration given to the sampling protocol and analytical methodology.

Examples of situations contributing to divergent assessments include the following:

- analytical errors;
- where the importance of non-point sources is not appreciated in the initial survey;
- where historical data are of insufficient quality to identify the source of pollution, giving rise to episodic peaks; and
- where extreme events, whether anthropogenic or natural in origin, arise from damaged infrastructure and/or inappropriate sewage disposal practices, e.g., shipping damage to marine outfalls or historical connections to surface water systems of foul drains from domestic and other properties.

Where sanitary inspection indicates low risk but water quality data inspection indicates water of low quality, this may indicate previously unidentified sources of diffuse pollution. In this case, specific studies demonstrating the relative levels of human and non-human contamination (e.g., analysis of faecal sterols, surveys of mammal and bird numbers) are recommended. Confirmation that contamination is primarily from non-human sources may allow reclassification (see below) to a more favourable grading.

Similarly, where microbiological assessment would indicate a very low risk that is not supported by the sanitary inspection, consideration should be given to the sampling design and the analytical methodology used.

4) Reclassification

Only where the picture revealed by the sanitary inspection is stable and favourable or unfavourable will it be appropriate to stop at the initial classification step. This situation is most likely to occur when the classification is very good or very poor. In other cases, authorities should continue on to the reclassification loop shown in Figure 4.6.

In recognition of the fact that water contamination may be triggered by specific and predictable conditions (e.g., rainfall), it is also proposed that local management actions can be employed to reduce or prevent exposure at such times. Provided the effectiveness of such actions can be demonstrated, the recreational-water environment may be reclassified to a more favourable level. A reclassification should therefore initially be provisional (see below); it may be confirmed if the efficacy of management interventions is verified during the initial season of provisional reclassification.

Some of the events triggering water contamination can be measured by simple means, such as rainfall gauges, detectors on stormwater overflows, etc. More sophisticated approaches involving modelling may be appropriate under some circumstances. Real-time prediction of faecal indicator concentrations at recreational compliance points has been achieved using two principal approaches. The first, a simple stochastic method, uses antecedent conditions to calibrate a statistical model based on multivariate statistical methods such as multiple regression. Predictor variables that can be used are:
- degree and type of treatment applied to sewage;
- preceding rainfall;
- wind direction;
- tides and currents;
- visible/modelled plume location;
- solar irradiance (and turbidity of water); and
- physicochemical parameters of water quality (salinity/conductivity [inexpensive and reliable], ammonium/phosphate [more expensive and less reliable]).

The alternative approach is the construction of a near-shore hydrodynamic model linked to a water quality model predicting faecal indicator concentrations (Falconer et al., 1998).

Both approaches offer potential for real-time prediction of bacterial concentrations at beach compliance points for protection of public health through timely management interventions.

5) Provisional classification

There will be occasions when there is a pressing need to issue advice on the classification of a recreational-water environment, even though the full information required in Figure 4.6 for moving to the classification (or reclassification) step is missing. Three scenarios may be envisaged:

- where there are no data of any kind available as to the microbiological quality of the water body or its susceptibility to faecal influence;
- where the data available are incomplete, in respect of either the microbiological assessment category or the sanitary inspection or both; and
- where there is reason to believe that the existing classification no longer accords with changed circumstances, but the data required for completing the reclassification loop in Figure 4.6 are insufficient.

In these circumstances, it may be necessary to issue a provisional classification (Box 4.8). When such a step is taken, it should be made clear that the advice is provisional and subject to change, and there should be a commitment to obtaining the necessary data for the purposes of Figure 4.6 as soon as possible.

Box 4.8: Actions for provisional classification

A. No historical data or assessment

Examples of recreational-water environments for which no sanitary inspection information and no water quality data are available include a newly used beach or a part of a long beach that becomes “popular.”

The first step is to identify the extent of the water body or beachfront requiring classification. Urgent microbiological assessment will be required; if the immediate sampling capacity is insufficient for the size of the water body, select the most intensively used recreational-water area as the transect for initial study.

At the first opportunity and in any event while water use is occurring, take a minimum of 8–12 samples across the selected transect, ideally at about 50-m intervals, but in any case not more than 200 m apart.

At the time of initial sampling, conduct a limited sanitary inspection, for the purpose of identifying possible pollution sources in the immediate vicinity of the area that will require further evaluation. Over the next day or two, while the presumptive laboratory results are awaited, the sanitary inspection should be completed as far as possible and arrangements made for the receipt of maps, plans and other information that may be needed for a proper interpretation of the findings.

Review the initial laboratory results as soon as they become available. If these results are extremely good or extremely bad, it may already be obvious that the water body may be provisionally placed in microbiological assessment category A or D. For example, if almost all the samples have values over 500 enterococci/100 ml, then the 95th percentile will clearly exceed 500, thus provisionally placing the water in category D. Consequently, if at any time during the collection of classification data it becomes obvious that, once all 100 samples have been collected, the 95th percentile will exceed a particular classification boundary, then the bathing water should be provisionally classified at the appropriate level.

If the results are not so clear-cut, a second round of sampling will be needed. It is not necessary, or even desirable, to wait to re-sample, as long as bathing is occurring at the time of re-sampling.

On the basis of the sanitary and microbiological data available after the second round of sampling, an early assessment should be made, and, if judged necessary, a provisional classification of the recreational-water environment should be made and acted upon at that point. At the same time, a commitment should be made to proceed with all necessary steps to permit full classification of the area in accordance with Figure 4.6 as soon as possible.

B. Incomplete data

Where the data available are insufficient, in respect of either the microbiological assessment or the sanitary inspection or both, the first step is to review the data carefully to see whether it is possible to reach any provisional conclusions. It may turn out that this is relatively easy to do at the extreme ends of the classification spectrum. For example, a major sewage discharge point in the immediate vicinity of the bathing area or a set of sampling results with a strong trend to very high or very low values may enable a provisional classification to be made. If not, the review may make it apparent where the key deficiencies in the data lie and so point the way to what additional information is most critically needed.

It is recognized that the absence of past enterococcal indicator data may be a special case, where use needs to be made of historical records relating to another indicator, such as *E. coli.* The issue of conversion factors that may be applied for that purpose is dealt with elsewhere in this chapter (section 4.4.3).

If, upon review, it is concluded that the data are insufficient to allow any conclusion to be drawn as to the appropriate classification of the recreational-water environment, a complete or virtually complete application of the data-gathering process in Figure 4.6 may need to be embarked upon. If it is necessary for this to be done on an urgent basis, the procedure outlined above for a recreational-water environment for which there are no data may be adapted accordingly.

C. Inappropriate existing classification

Where there is reason to believe that the existing classification no longer accords with changed circumstances, but the data required for completing the reclassification loop in Figure 4.6 are insufficient, it will be necessary, as in the previous scenario relating to the problem of incomplete data, to carry out a careful review of the existing data to see whether it is possible to reach any provisional conclusions.

If this review shows an incongruity between the sanitary inspection data and the microbiological water quality data, steps should be taken, as set out elsewhere in this chapter, to resolve that incongruity. Where both the sanitary inspection data and the microbiological water quality data point to a similar inappropriateness in the existing classification, it may well be that the strength of their association does enable a provisional conclusion to be drawn. If not, however, the steps set out above for overcoming the problem of incomplete data should be followed.

4.5 Management action

Management action in response to a recreational-water environment classification indicating unacceptable faecal contamination can be both immediate, such as public health advisories, and long term, such as pollution abatement.

4.5.1 Informed personal choice

Good quality information in near-real time about the recreational-water environment is of utmost importance. Such information is needed by the public to make informed choices about if and where to use recreational-water areas; it is also needed by competent authorities to make long-term decisions about water quality management. Comprehensive information should be actively provided by those who collect it.

In some regions, such a strategy may induce competition between resorts/destinations based upon relative safety. Eventually, economic competition may prove effective in facilitating improvement in recreational-water quality.

Classification of beaches into various categories of health thereby provides users with the ability to make informed choices in selecting between potentially competing recreational locations. Informed consumer choice can reduce both overall exposure and adverse health outcomes and produce an economic incentive for water quality improvement. It is thereby important that the information does not cover only the *microbiological quality category*, such as categories A to D, but that that information is *linked to the sanitary inspection category* leading to an *overall quality classification* — from very poor to very good — as presented in Table 4.8. Such a classification provides the incentive to improve poor locations through public awareness of both beach class and risk to health.

4.5.2 Public health advisories and warnings

Recreational-water managers may take steps to identify periods when water quality is poor, issue advisory notices warning the public of increased risk and audit the impact of those advisories in discouraging bathing. This approach has the benefit of protecting public health and, in many circumstances, provides potential both to improve the class of a location through low-cost measures and to enable safe use of areas that might otherwise be considered inappropriate for recreational use.

Some locations will consistently have very poor water quality due to the proximity of effluent discharges; others will have intermittently poor water quality due to accidental pollution that may be rare or impossible to predict. Still more sites will have episodic, but possibly predictable, deterioration in water quality, such as that driven by meteorological conditions, particularly rainfall. In any of these circumstances, local public health agencies may wish to issue an advisory notice or other form of public notification. The level at which an advisory might be issued depends on local circumstances, which include levels and type of endemic illness prevalent in the population and the existence of any “outbreaks” or endemic occurrence of potentially serious illness that may be spread by bathing water exposure.

4.5.3 Long-term remedial action

Recreational waters, especially near urban areas, are often subject to pollution due to sewage and industrial discharges, combined sewer overflows and urban runoff. Pollution abatement measures for sewage may be grouped into three major wastewater disposal alternatives: treatment, dispersion through sea outfalls and discharge to non-surface waters (i.e., reuse, groundwater injection).

1) Direct point source pollution abatement

Effective "long" outfalls with sufficient length and diffuser discharge depth are designed to ensure a low probability of the sewage plume reaching the designated recreational-water environment. Therefore, the premise is to separate the bather from contact with diluted sewage, and, as such, long outfalls are a reliable alternative to protect public health. Pre-treatment with milli-screens is considered to be the minimum treatment level.

For nearshore discharges of large urban communities, at least secondary or tertiary sewage treatment plants with disinfection will reduce risks. Public health risks will vary depending on the operation of the plant (avoiding "upsets") and the effectiveness of disinfection. Some viruses will be resistant to disinfection (Reynolds et al., 2000), and *Cryptosporidium* may not be killed (Carpenter et al., 1999). UV disinfection and to a certain extent ozonation are more effective than chlorination at removing both indicator bacteria and viruses. Smaller communities with lesser population density can apply treatment via septic tank systems, latrines, etc., where adequate subsoil conditions permit.

Reuse of wastewater for agricultural or other purposes essentially eliminates health risks for recreational-water areas.

2) Non-point source pollution abatement

Runoff via drainage ditches, combined sewer overflows, etc. are event-driven pollution sources. Combined sewer overflows may present the greater health risk, in that diluted untreated sewage may come into contact with bathers. Sanitary sewer overflows present a direct health risk that can usually be avoided.

Pollution abatement alternatives are: a) holding tanks that discharge during non-recreational water use periods, which are costly and often impractical for large urban areas; b) transport to locations distant from designated recreational areas via piped collection systems or effective outfalls; and c) disinfection (ozone, chlorine or UV). Although treatment is an option, often the treatment plant cannot cope with the quantity of the sewage or the effectiveness of the treatment is lowered due to the lesser "quality" of the sewage (too much dilution).

The above pollution abatement alternatives usually require major capital expenditures for event-driven pollution episodes and, as such, may not be readily justifiable, especially in developing countries. The alternative to be usually adopted is the development and application of management programmes that minimize recreational use during event-driven pollution episodes.

In general terms, combined sewer systems present a greater hazard to recreational waters than separate sewer systems for event-driven pollution episodes. The best option is to have separate collection systems for sewage and rain/stormwater.

3) Drainage basin pollution abatement

Upstream point and non-point source discharges and riverine discharges to coastal recreational areas may represent potentially significant pollution sources that present a broad challenge to the manager of recreational-water areas for pollution abatement. Major sources of pollution must be identified and a basin-wide pollution abatement programme developed inclusive of waste load allocation analyses. Cross-agency and interdisciplinary cooperation is required among health and environmental control agencies, users, polluters, etc. The role of the agricultural sector in generation and remediation of pollution loadings is crucial here.

4.5.4 Enforcement of regulatory compliance

Problems exist in the application of regulatory compliance as a principal tool for the protection and improvement of microbiological quality of recreational waters. The two principal problems concern responsibility for cause of failure and the nature of intervention.

Where a recreational-water use location fails a regulatory standard, major problems may exist in defining responsibility. In many locations, a number of diverse sources will contribute pollution loading to the overall pollution outcome, and the relative importance of different sources may vary greatly with time. Rivers often function as major sources of microbial loads and will in turn be affected greatly by, for instance, rainfall; also, they may themselves be recipients of multiple pollution loads. Approaches to regulatory compliance enforcement that depend upon identifying and requiring change of a discharge/pollution source “responsible” for failure are therefore problematic. Integrated catchment management and control of such pollution loadings are essential.

4.6 Monitoring and auditing

Monitoring and auditing include visual inspection of potential sources of contamination in a catchment, water sampling and verification of critical control points. Critical control points are those points that can be monitored to provide information to management so that management actions can have an impact on the exposure risk (section 4.2.3). Examples include rainfall in the catchment, municipal discharge points, treatment works operation, combined sewer overflows and illegal connections to combined sewers.

Following initial classification, it is proposed that all categories of recreational-water environment would be subject to an annual sanitary inspection to determine whether pollution sources have changed.

For recreational-water areas where no change to the sanitary survey has occurred and the sanitary inspection category was “Very low” or “Low,” it is proposed that microbiological assessment be repeated only once every 5 years. For beaches where the sanitary inspection resulted in a “Very high” categorization for susceptibility to faecal contamination (where swimming would be strongly discouraged), a similar situation applies. For intermediate quality recreational-water environments ("Moderate" and "High"), an annual microbiological sampling programme is recommended (Table 4.14).

Table 4.14: Recommended monitoring schedule

Sanitary inspection category	Microbiological inspection	Sanitary inspection
Very low	Every 5 years	Annual
Low	Every 5 years	Annual
Moderate	Annual low-level sampling 4 samples × 5 occasions during bathing season Annual verification of management effectiveness Additional sampling if abnormal results obtained	Annual
High	Annual low-level sampling 4 samples × 5 occasions during bathing season Annual verification of management effectiveness Additional sampling if abnormal results obtained	Annual
Very high	Every 5 years	Annual

4.7 References

Ackman D, Marks S, Mack P, Caldwell M, Root T, Birkhead G (1997) Swimming-associated haemorrhagic colitis due to *Escherichia coli* O157:H7 infection: evidence of prolonged contamination of a fresh water lake. *Epidemiology and infections*, 119: 1–8.

APHA (1989) *Standard methods for the examination of water and wastewater,* 17th ed. Washington, DC, American Public Health Association.

Ashbolt NJ, Reidy C, Haas CN (1997) Microbial health risk at Sydney's coastal bathing beaches. In: *Proceedings of the 17th Australian Water and Wastewater Association Meeting, 16–21 March 1997, Melbourne, Vol. 2.* Melbourne, Australian Water and Wastewater Association, pp. 104–111.

Bahlaoui MA, Baleuz B, Troussellier M (1997) Dynamics of pollution-indicator and pathogenic bacteria in high-rate oxidation wastewater treatment ponds. *Water research*, 31: 630–638.

Bartram J, Rees G, ed. (2000) *Monitoring bathing waters: a practical guide to the design and implementation of assessments and monitoring programmes*. London, E & FN Spons. Published on behalf of the World Health Organization, Commission of the European Communities and US Environmental Protection Agency.

Bartram J, Fewtrell L, Strenström T-A (2001) Harmonised assessment of risk and risk management for water-related infectious disease. An overview. In: Fewtrell L, Bartram J, ed. *Water quality: Guidelines, standards and health. Risk assessment and management for water-related infectious disease*. London, IWA Publishing (in press).

Barwick RS, Levy DA, Craun GF, Beach MJ, Calderon RL (2000) Surveillance for waterborne-disease outbreaks — United States, 1997–1998. *Morbidity and mortality weekly report*, 49: 1–37.

Cabelli VJ, Dufour AP, Levin MA, McCabe LJ, Haberman PW (1979) Relationship of microbial indicators to health effects at marine bathing beaches. *American journal of public health,* 69: 690–696.

Cabelli VJ, Dufour AP, McCabe LJ, Levin MA (1982) Swimming-associated gastroenteritis and water quality. *American journal of epidemiology*, 115(4): 606–616.

Carpenter C, Fayer R, Trout J, Beach MJ (1999) Chlorine disinfection of recreational water for *Cryptosporidium parvum. Emerging information on diseases*, 5 (http://www.cdc.gov/ncidod/EID/vol5no4/carpenter.htm).

Cioglia L, Loddo B (1962) The process of self-purification in the marine environment III. Resistance of some enteroviruses. *Nouvi Annali di Igiene E, Microbiologia,* 13: 11.

Cransberg K, van den Kerkhof JH, Banffer JR, Stijnen C, Wernars K, van de Kar NC, Nautal J, Wolff ED (1996) Four cases of hemolytic uremic syndrome — source contaminated swimming water? *Clinical nephrology*, 46: 45–49.

Craun GF, Berger PS, Calderon RL (1997) Coliform bacteria and waterborne disease outbreaks. *Journal of the American Water Works Association*, 9: 96–104.

Davies-Colley RJ, Donnison AM, Speed DJ (2000) Towards a mechanistic understanding of pond disinfection. *Water science and technology*, 42(10–11): 149–158.

Deere D, Stevens M, Davison A, Helm G, Dufour A (2001) Management strategies. In Fewtrell L, Bartram J, ed. (2001) *Water quality: Guidelines, standards and health. Risk assessment and management for water-related infectious disease*. London, IWA Publishing (in press).

Dufour AP (1984) *Health effects criteria for fresh recreational waters*. Cincinnati, OH, US Environmental Protection Agency (EPA 600/1-84-004).

EEC (European Economic Community) (1976) Council Directive 70/160/EEC of 8 December 1975 concerning the quality of bathing water. *Official journal of the European Union*, 19: L 31.

Ellis JC (1989) *Handbook on the design and interpretation of monitoring programmes*. Medmenham, Water Research Centre (Report NS 29).

El Sharkawi F, Hassan MNER (1982) The relation between the state of pollution in Alexandria swimming beaches and the occurrence of typhoid among bathers. *Bulletin of the High Institute of Public Health of Alexandria*, 12: 337–351.

Esrey SA, Faechem RG, Hughes JM (1985) Interventions for the control of diarrhoeal disease among young children: improving water supplies and excreta disposal facilities. *Bulletin of the World Health Organization*, 63(4): 757–772.

Falconer RA, Lin B, Kay D, Stapleton C (1998) *Predicting coastal health risks using combined water quality and epidemiological models*. In: Babovic V, Larsen LC, ed. *Hydroinformatics '98*. Rotterdam, A.A Balkema, pp. 1213–1218.

Ferley JP, Zmirou D, Balducci F, Fera P, Larbaigt G, Jacq E, Moissonnier B, Blineau A, Boudot J (1989) Epidemiological significance of microbiological pollution criteria for river recreational waters. *International journal of epidemiology*, 18(1): 198–205.

Fewtrell L, Jones F, Kay D, Godfree AF, Merrett H (1993) *Pathogens in surface temperate waters*. United Kingdom, Samara Press, 262 pp.

Figueras MJ, Robertson W, Pike EB, Ashbolt NJ, Borrego JJ (2000) Sanitary inspection and microbiological water quality. In: Bartram J, Rees G, ed. *Monitoring bathing waters: a practical guide to the design and implementation of assessments and monitoring programmes*. London, E & FN Spons, pp. 113–167. Published on behalf of the World Health Organization, Commission of the European Communities and US Environmental Protection Agency.

Fleisher JM (1990) The effects of measurement error on previously reported mathematical relationship between indicator organism density and swimming-associated illness: a quantitative estimate of the resulting bias. *International journal of epidemiology*, 19(4): 1100–1106.

Fleisher J, Kay D, Jones F, Morano R (1983) Setting recreational water criteria. In: Kay D, Hanbury R, ed. *Recreational water quality management. Vol. 2. Freshwaters.* Chichester, Ellis Horwood, pp. 123–135.

Fleisher JM, Kay D, Salmon RL, Jones F, Wyer MD, Godfree AF (1996a) Marine waters contaminated with domestic sewage: nonenteric illnesses associated with bather exposure in the United Kingdom. *American journal of public health*, 86(9): 1228–1234.

Fleisher JM, Kay D, Wyer M, Merrett H (1996b) The enterovirus test in the assessment of recreational water-associated gastroenteritis. *Water research*, 30: 2341–2346.

Fleisher JM, Kay D, Wyer MD, Godfree AF (1998) Estimates of the severity of illnesses associated with bathing in marine recreational waters contaminated with domestic sewage. *International journal of epidemiology*, 27: 722–726.

Fujioka R, Sian-Denton C, Borja M, Castro J, Morphew K (1999) Soil: the environmental source of *Escherichia coli* and enterococci in Guam's streams. *Journal of applied microbiology, symposium supplement*, 85: 83S–89S.

Gantzer C, Dubois E, Crance JM, Billaudel S, Kopecka H, Schwartzbrod L, Pommepuy M, Le Guyader F (1998) [Influence of environmental factors on the survival of enteric viruses in seawater.] *Oceanologica acta*, 21: 983–992 (in French).

Gerba CP (2000) Assessment of enteric pathogen shedding by bathers during recreational activity and its impact on water quality. *Quantitative microbiology*, 2: 55–68.

Haas CN (1983) Estimation of risk due to low doses of microorganisms: a comparison of alternative methodologies. *American journal of epidemiology*, 118: 573–582.

Haas CN, Rose JB, Gerba C, Regli S (1993) Risk assessment of virus in drinking water. *Risk analysis*, 13: 545–552.

Haas CN, Rose JB, Gerba CP (1999) Quantitative microbial risk assessment. New York, John Wiley & Sons, Inc., 449 pp.

ISO (1975) *Draft report of SC4/WGS meeting on sulfite-reducing spore-forming anaerobes (Clostridia), 16 January 1975*. Berlin, International Organization for Standardization.

ISO (1990) *Water quality — Detection and enumeration of coliform organisms, thermotolerant coliform organisms and presumptive Escherichia coli — Part 1: Membrane filtration method*. Geneva, International Organization for Standardization (ISO 9308-1).

ISO (1995) *Water quality — Detection and enumeration of bacteriophages — Part 1: Enumeration of F-specific RNA bacteriophages*. Geneva, International Organization for Standardization, 15 pp. (ISO 10705-1:1995).

ISO (1998a) *Water quality — Detection and enumeration of intestinal enterococci in surface and waste water — Part 1: Miniaturized method (most probable number) by inoculation in liquid medium.* Geneva, International Organization for Standardization (ISO/FDIS 7899-1).

ISO (1998b) *Water quality — Detection and enumeration of Escherichia coli and coliform bacteria in surface and waste water — Part 3: Miniaturized method (most probable number) by inoculation in liquid medium.* Geneva, International Organization for Standardization (ISO/FDIS 9308-3).

Johnson DC, Gerba CP (1996) *Giardia muris* in marine waters. In: *Proceedings of IAWQ Health-Related Water Microbiology, 6–10 October 1996, Mallorca, Spain.* Sant Pere, International Association for Water Quality, p. 51.

Jones DL (1999) Potential health risks associated with the persistence of *Escherichia coli* O157 in agricultural environments. *Soil use and management*, 15: 76–83.

Kay D, Fleisher JM, Salmon RL, Wyer MD, Godfree AF, Zelenauch-Jacquotte Z, Shore R (1994) Predicting likelihood of gastroenteritis from sea bathing; results from randomized exposure. *Lancet*, 344(8927): 905–909.

Kay D, Jones F, Fleisher J, Wyer M, Salmon RL, Lightfoot N, Godfree A, Pike E, Figueras MJ, Masterson B (1996) *Relevance of faecal streptococci as indicator of pollution. Report to DG XI of the Commission of the European Communities.* Leeds, University of Leeds, Centre for Research into Environmental Health, 30 pp.

Kay D, Wyer MD, Crowther J, Fewtrell L (1999) Faecal indicator impacts on recreational waters: budget studies and diffuse source modelling. *Journal of applied microbiology, symposium supplement*, 85: 70S–82S.

Kay D, Fleisher J, Wyer MD, Salmon RL (2001) *Re-analysis of the seabathing data from the UK randomised trials. A report to DETR.* Aberystwyth, University of Wales, Centre for Research into Environment and Health, 17 pp.

Keene WE, McAnulty JM, Hoesly FC, Williams LP, Hedber K, Oxman GL, Barrett TJ, Pfaller MA, Fleming DW (1994) A swimming-associated outbreak of hemorrhagic colitis caused by *Escherichia coli* O157:H7 and *Shigella sonnei*. *New England journal of medicine*, 331: 579–584.

Kramer MH, Herwaldt BL, Craun GF, Calderon RL, Juranek DD (1996) Surveillance for waterborne-disease outbreaks — United States, 1993–1994. *Morbidity and mortality weekly report*, 45: 1–33.

Kramer MH, Sorhage FE, Goldstein ST, Dalley E, Wahlquist SP, Herwaldt BL (1998) First reported outbreak in the United States of cryptosporidiosis associated with a recreational lake. *Clinical infectious disease,* 26: 27–33.

Leeming R, Nichols PD, Ashbolt NJ (1998) *Distinguishing sources of faecal pollution in Australian inland and coastal waters using sterol biomarkers and microbial faecal indicators.* Melbourne, Water Services Association of Australia, 46 pp. (Research Report No. 204).

Levy DA, Bens MS, Craun GF, Calderon RL, Herwaldt BL (1998) Surveillance for waterborne-disease outbreaks — United States, 1995–1996. *Morbidity and mortality weekly report*, 47: 1–34.

Long J, Ashbolt NJ (1994) *Microbiological quality of sewage treatment plant effluents.* Sydney, Sydney Water Corporation, 26 pp. (AWT Science & Environment Report No. 94/123).

Okhuysen PC, Chappell CL, Crabb JH, Sterling CR, DuPont HL (1999) Virulence of three distinct *Cryptosporidium parvum* isolates for healthy adults. *Journal of infectious diseases*, 180: 1275–1281.

Philipp R, Waitkins S, Caul O, Roome A, McMahon S, Enticott R (1989) Leptospiral and hepatitis A antibodies amongst windsurfers and waterskiers at Bristol City Docks. *Public health,* 103: 123–129.

Prüss A (1998) A review of epidemiological studies from exposure to recreational water. *International journal of epidemiology,* 27: 1–9.

Public Health Laboratory Service (1959) Sewage contamination of coastal bathing waters in England and Wales: a bacteriological and epidemiological study. *Journal of hygiene*, 43: 435–472.

Reynolds KA, Roll K, Fujioka RS, Gerba CP, Pepper IL (1998) Incidence of enteroviruses in Mamala Bay, Hawaii using cell culture and direct polymerase chain reaction methodologies. *Canadian journal of microbiology,* 44: 598–604.

Reynolds KA, Gerba CP, Pepper IL (2000) Use of integrated cell culture-PCR to evaluate the effectiveness of poliovirus inactivation by chlorine. *Applied environmental microbiology*, 66: 2267–2268.

Rose JB, Mullinax RL, Singh SN, Yates MV, Gerba CP (1987) Occurrence of rotaviruses and enteroviruses in recreational waters of Oak Creek, Arizona. *Water research*, 21: 1375–1381.

Spano F, Putignani L, Crisanti A, Sallicandro P, Morgan UM, Le Blancq SM, Tchack L, Tzipori S, Widmer G (1998) Multilocus genotypic analysis of *Cryptosporidium parvum* isolates from different hosts and geographical origins. *Journal of clinical microbiology*, 36: 3255–3259.

US EPA (1998) *Bacterial water quality standards for recreational waters (freshwater and marine waters). Status report. May 1998.* Washington, DC, US Environmental Protection Agency, Office of Water (http://www.epa.gov/OST/beaches/local/sum2.html) (EPA-823-R-98-003).

van Asperen IA, Medema G, Borgdorff MW, Sprenger MJ, Havelaar AH (1998) Risk of gastroenteritis among triathletes in relation to faecal pollution of fresh waters. *International journal of epidemiology*, 27: 309–315.

Voelker R (1996) Lake-associated outbreak of *Escherichia coli* O157:H7 Illinois, 1995. *Journal of the American Medical Association*, 275: 1872–1873.

WHO (1999) *Health-based monitoring of recreational waters: the feasibility of a new approach (the "Annapolis Protocol").* Geneva, World Health Organization (http://www.who.int/environmentalinformation/Informationresources/documents/Recreationalwaters/Annapolis.pdf) (Protection of the Human Environment Water, Sanitation and Health Series, WHO/SDE/WSH/99.1.)

Wyer MD, Kay D, Fleisher JM, Salmon RL, Jones F, Godfree AF, Jackson G, Rogers A (1999) An experimental health-related classification for marine waters. *Water research*, 33: 715–722 [correction, addition: *Water research*, 33(10): 2467, 1999].

Wyn-Jones AP, Pallin R, Deboussis C, Shore J, Sellwood J (2000) The detection of small round-structured viruses in water and environmental materials. *Journal of virological methods*, 87: 99–107.

Yates MV, Gerba CP (1998) Microbial considerations in wastewater reclamation and reuse. In: Asano T, ed. *Wastewater reclamation and reuse, Vol. 10.* Lancaster, PA, Technomic Publishing Co. Inc., pp. 437–488 (Water Quality Management Library).

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